

ECOSYSTEM-BASED EVALUATION OF FISHERY POLICIES AND TRADEOFFS ON
THE WEST FLORIDA SHELF

By

DAVID DEMETRE CHAGARIS JR.

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To Lauren and Hannon

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Abstract of Dissertation Presented to the Graduate School
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David Demetre Chagaris Jr.

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Ecosystem-based fishery management inherently involves confronting tradeoffs and requires taking a holistic approach to evaluating and developing fisheries management strategies that look beyond single-species harvest management. The goal of this study was to identify ecological tradeoffs that arise through predator-prey interactions, tradeoffs between conflicting management objectives, and tradeoffs associated with marine protected areas. To accomplish this, I developed a trophic-dynamic ecosystem model of the West Florida Shelf with particular emphasis on the reef fish component (groupers and snappers) of the fish community, using Ecopath with Ecosim and Ecospace.

The dynamic Ecosim model was used to project forward a suite of management prescriptions and predict ecosystem impacts that occur due to predation and competition. Overall, I found that there were winners and losers in all policy options and that Vermilion Snapper (*Rhomboplites aurorubens*) and Black Seabass (*Centropristis striata*) were predicted to be at a competitive disadvantage to other groupers and snappers. Mostly modest impacts were predicted for policies oriented towards single

species, such as rebuilding plans, and more widespread impacts were predicted for scenarios that affect a suite of species or species in the middle of the food web.

The tradeoff relationship between reef fish biomass and fishery profits was revealed through a policy optimization search. My results suggested that the WFS is suboptimal in terms of both biomass and profits and that a set of balanced policy options exist where profits and reef fish biomass both increase by about 20% and 50% respectively. Lastly, marine protected areas were evaluated using a spatially explicit model, Ecospace. The model predicted little ecological or economic benefit from the small existing closures on the WFS and suggested that MPAs between 15-30% of the area can increase population sizes without causing economic collapse of the fishery. MPAs that were larger in size and with less edge were more beneficial than many smaller MPAs.

CHAPTER 1 GENERAL INTRODUCTION

Introduction

Fisheries function as part of the larger ecosystem that includes the relationships between species, the physical environment, the social communities and economic markets that rely on fisheries, and the institutions that regulate them. Historically, fisheries managers focused on achieving the maximum sustainable catch of individual target species without consideration to critical processes such as predator-prey interactions, habitat alterations, and changes in system productivity. Further, single species fishery objectives usually had little regard for the objectives of diverse user groups (e.g. conservation stakeholders, resource managers, the scientific community, and the general public) when setting regulations. Recently, regulatory agencies have been encouraged, and in some cases mandated to take a more holistic approach to managing fisheries. This management process has come to be known as ecosystem based fisheries management (EBFM) or the ecosystem approach to fisheries (EAF).

Because EBFM is, by nature, multi-objective and interdisciplinary, it often generates conflicts within and among user groups who have different objectives (e.g., maximizing yield versus maintaining ecological diversity) and management strategies to achieve those objectives. Confronting such conflicts often involves making a tradeoff - giving up one thing in return for another, or more desirably, a balancing of conflicted objectives that are not attainable at the same time. Tradeoffs are a ubiquitous and central problem to managing fisheries, and EBFM approaches are well suited to address the full range of issues spanning all user groups (Cheung and Sumaila 2008; Christensen and Walters 2004; Heymans et al. 2009). In most cases the main source

of conflict occurs between the need for stock conservation and the well-being of the fishing community today versus into the future. Some common tradeoffs that are outside the context of single-species management include: harvest of one species versus abundance of its predators, biological diversity versus fishery production, abundance of unproductive stocks versus harvest of more productive stocks, and economic efficiency versus diversified employment (Walters and Martell 2004). These tradeoffs have so far been rarely explored.

Exposing the unintended consequences of fisheries and the tradeoffs they impose is critical to sustaining fisheries. When tradeoffs are not identified and acknowledged, policy choices are made based on the interests of one stakeholder, which can often be costly to others (Walters and Martell 2004). Fisheries managers are rarely (if ever) provided the comprehensive information required to make decisions that avoid negative impacts to both the ecosystem and society while balancing tradeoffs among several interest groups. My dissertation aims to provide such information using a quantitative multi-species and multi-fleet dynamic model of the West Florida Shelf reef fish fishery and is intended to be useful for EBFM in the Gulf of Mexico.

Ecosystem Based Fishery Management: Definition and History

Ecosystem-based fishery management (EBFM) essentially means taking a more holistic approach to marine resource management (Larkin 1996). Link (2010) described EBFM as managing fisheries in a coordinated and integrated fashion that starts with an ecosystem perspective on fisheries production and addresses specific interactions among species and with the environment. Similarly, the ecosystem approach to fisheries (EAF) can be thought of as starting with a single stock and building up to include multiple trophic levels and environmental drivers (Link 2010). Within the

literature and management arena, these terms are often used interchangeably and I will not attempt to distinguish between them in this thesis. Ultimately, EAF and EBFM both emphasize a broader context when managing fisheries with specific consideration to how fisheries impact, and are impacted by, other components of the ecosystem and other user groups.

Concepts germane to EBFM can be traced back over 100 years. Spencer Baird, the founder of the U.S. Fisheries Service, recognized during the late 19th century that excessive overfishing, food web interactions, changing environmental conditions, and pollution were all possible causes for declines in New England fish stocks (Link 2010). Debates over the relative importance of fishing versus the environment on fluctuating fish stocks, most notably the Thompson-Burkenroad debate (Skud 1975), arose during the first half of the twentieth century and are still argued today when discussing EBFM (Rose 2000; Walters and Collie 1988). Other advancements that occurred around the same time period influenced the quantitative and applied ecosystem models of today, these include Lindeman's trophic-dynamic aspect of ecology (Lindeman 1942) and the systems ecology perspective of the Odum brothers (Odum 1956; Odum and Odum 1955).

The global sustainable development initiative of the 1970s and 1980s acknowledged that human use was impacting Earth's ecosystems. This initiative was a precursor to EBFM and called for a precautionary approach and the need to consider a broader range of factors effecting natural resources. Soon thereafter, The Food and Agricultural Organization (FAO) and United Nations (UN) recognized EAF as a general management and research need for fisheries (FAO 1995; UN 2002). In the United

States, the 2006 reauthorization of the Magnuson-Stevens Fisheries Conservation and Management Act of 2006 explicitly emphasized the need for ecosystem approaches to fisheries management and mandated that councils advance the concept of integrating ecosystem considerations into regional fisheries management.

The Gulf of Mexico Fisheries Management Council (GMFMC) is the body responsible for managing fisheries from west Florida to Texas. In 2005 The GMFMC participated in a pilot project to develop EAF and as a result created an Ecosystem Scientific and Statistical committee (ESSC) whose goals are to identify and investigate potential ecosystem effects of fishing, foster development of ecosystem level indicators and ecosystem models, and communicate their findings to the GMFMC for consideration when drafting harvest policies (Atran et al. 2010). Each of the eight regional fishery management councils in the U.S. has taken a slightly different approach to meeting this mandate, but the common theme is that there is a strong need for multi-disciplinary approaches to data collection and management, collaboration among scientists at different levels of government and academia, development of ecosystem scale models, and national guidance and endowment of EBFM activities (NMFS 2009).

Food Webs and Fisheries

There are unintended and unforeseen consequences of fishing that are not being addressed through single-species approaches. These indirect effects of fishing usually arise through disruptions to the food web, whether they are “top-down” (predator dominated) or “bottom-up” (nutrient driven). For example, removing predators can cause an increase in abundance of their prey and a decline of organisms two trophic levels below them, a phenomenon known as a trophic cascade (Carpenter *et al.* 1985). Frank et al. (2005) provide evidence of a trophic cascade on the eastern Scotian Shelf

off Nova Scotia, Canada where over exploitation of large-bodied predators (cod, haddock, pollock) during the 1980s resulted in increased abundance of pelagic fishes, crabs, and shrimp followed by subsequent declines in zooplankton. Consequently, consumption on the larval stage of the predator by small pelagic fishes may now be suppressing the recovery of the benthic predators (Walters and Kitchell 2001). A similar situation involving cod in the Baltic Sea provides even more evidence for top-down control and trophic cascades in large marine ecosystems (Steneck 2012).

Fishing on lower trophic level species, planktivorous “forage” fishes for example, can also have effects on other components of the system. When the net productivity of a prey species is diverted to harvest, predator populations will ultimately decline (Walters and Martell 2004). Ecosystem models have shown that even sustainable harvest of forage fish could negatively impact its predators (Pikitch et al. 2012; Smith et al. 2011; Walters et al. 2005). However, these effects could be mediated by the increased availability of alternative prey in response to fishing. In systems with a “wasp-waist” trophic structure (i.e. upwelling systems), where forage fishes exert top-down control on zooplankton and bottom-up control on predators, the overall ecosystem responds more strongly to changes in stock size of the forage fish than to changes in predator biomass (Cury et al. 2000).

Competition for food occurs when there are two or more species that overlap in time and space and utilize the same limited resource (Link 2010). Competition is a process assumed to be closely tied to carrying capacity and population growth rate but has not been emphasized nearly as much as direct predator-prey interactions. An early simulation model of species interactions on Georges Bank suggested that competitive

interactions were occurring during earlier years when predator abundances were still high (Overholtz and Tyler 1986). Following this logic, rebuilding of depleted species can be expected to have consequences on other members of the community through increased competition (Andersen and Rice 2010; Hartman 2003). Therefore, competition can have implications for management because simultaneously rebuilding two competing species may require more conservative harvest restrictions.

While it may be obvious that fishing can have widespread impacts on an ecosystem, changes in primary production can also have noticeable effects throughout. These “bottom-up” processes are largely driven by changes in climate or physical oceanography, particularly those that drive patterns of precipitation or upwelling and therefore nutrient input. Fish stock dynamics are influenced by climate variability on multiple time scales as demonstrated in the Northeast Pacific Ocean (Hollowed *et al.* 2001). Nutrient loading due to runoff and high outflow from rivers increases primary productivity, especially in estuaries and coastal waters, thereby increasing the productivity of fishery resources (Houde and Rutherford 1993; Wilber 1994). However, over enrichment of nutrients can lead to hypoxia, poor water quality, and loss of aquatic vegetation (Breitburg 2002; Greening and Janicki 2006). While dynamics of lower trophic level species are more strongly tied to environmental forcing, for most species it's the combination of both fishing and environmental forcing that drive changes in population size (Chagaris and Mahmoudi 2009; Mackinson *et al.* 2009).

A key tenet of EBFM is to consider trophic-dynamic processes in management, whether they are bottom-up or top-down driven. Food web models, particularly those that can simulate changes over time, are needed to quantify the impacts of these

processes and the tradeoffs that they create. Models can be used to screen policy options for unintended consequences on the system, evaluate their effectiveness in an ever changing environment, and confront the tradeoffs among stakeholder groups that they impose. Additionally, food web models can serve to inform single species assessment and management and are capable of generating reference points (Walters *et al.* 2005) and ecosystem-level indicators (Coll *et al.* 2006; Fulton *et al.* 2005). My goal in this dissertation was to develop such a tool for use in the Eastern Gulf of Mexico.

Study Area

Physical Description

The West Florida Shelf (WFS) is a broad continental shelf with an area of about 170,000 km² that extends up to 220 km from shore across a very gradual slope to approximately the 100 m isobath where the shelf break begins (Figure 1-1). The topography is low relief ridges that run parallel to the shore. Local carbonate beds (“hardbottom”) are especially present on mid- to outer-shelf ridges (Roberts *et al.* 1999). Habitat areas that are of particular concern, such as the Florida Middle Grounds located 150 km south of the Panhandle and 160 km northwest of Tampa Bay, have relatively high relief reef-like structures covered with living mollusks, algae, and octocoral. The sediment grades from carbonate oozes at the shelf edge to carbonate sands on the outer shelf and mollusk-rich sands midshelf with quartz sand on the inner shelf. Sediments are transported north to south and driven by the Loop Current (Roberts *et al.* 1999).

Circulation along the WFS is driven by tides, winds, buoyancy fluxes and the Loop Current, which makes it a rather energetic environment. The Loop Current enters the Gulf through the Yucatan Channel, turns clockwise and exits through the Straits of

Florida. Upwelling along the Loop Current boundary is a major source of nutrients to the euphotic zone (Walsh *et al.* 1989). Deep water is upwelled at the shelf break where the Loop Current contacts the shelf and can sometimes be transported to much shallower depths (He and Weisberg 2003). Upper layer circulation on the mid- to outer-shelf is southward and there is little freshwater input south of Cedar Key, FL (Wiseman and Sturges 1999). Chlorophyll plumes have been observed during spring that extend southward along the middle shelf from about Cape San Blas and are thought to be related to river discharge from the Mississippi and NW Florida rivers, Loop Current circulation and upwelling in the DeSoto Canyon, and the pressure gradient between shelf and deeper waters (Gilbes *et al.* 1996).

The El Nino-La Nino Southern Oscillation (ENSO) drives rainfall patterns for the Southeast U.S. and Gulf of Mexico so as to result in roughly periodic variation over the order of 3-5 years, with El Nino years producing cooler and wetter winters. During fall and winter cold fronts bring north or northeasterly winds and often become stationary along the northern Gulf coast. In the spring, winds are out of the south and southeast and during the summer a high pressure system generally prevails over the Gulf (Hsu 1999). Hurricanes and low pressure tropical weather systems are capable of developing throughout the summer and into early fall, some of which can be extremely severe and change the distribution of dissolved organic matter and underwater light properties (Conmy *et al.* 2009). Sea surface temperature during the winter is between 16°C and 25°C while summer temperature is greater than 28°C (Liu *et al.* 2006). Salinity along the WFS is 35 ppt, varies seasonally and with precipitation, and usually lower around river outflows (NOAA 2012).

Biological Description

The WFS exhibits high biological diversity and contains a mix of both warm-temperate and tropical species. There are at least 1,000 different species of arthropods and mollusks, 500 species of annelid worms, 300 species of cnidarians and echinoderms, 700 species of bony fish, 48 species of sharks and rays, 28 marine mammals, and 232 species of birds (Moretzsohn 2012). Primary producers include phytoplankton consisting of diatoms and dinoflagellates, benthic microscopic algae, more than 300 species of red, brown, and green macroalgae, and three main types of seagrass (< 40 m) (Mackinson *et al.* 2001). Zooplankton is patchily distributed and consists of carnivorous chaetognaths, small herbivorous copepods less than 1.5 mm long (e.g. *Oithona sp.*, *Eucalanus sp.*), larger herbivorous copepods, and other larval meroplankton and ichthyoplankton. The meiobenthos, invertebrates that live interstitially and are less than 0.5 mm, is dominated by nematode worms and harpacticoid copepods. The benthic macrofauna (sediment organisms larger than 0.5 mm) of the WFS is dominated by polychaete worms, amphipod crustaceans, bivalves, and bryozoans with community structure varying by seasons (especially nearshore), sediment type, and bathymetry (Rabalais *et al.* 1999). These are preyed upon by larger mobile penaeid and sicyonid shrimps, portunid crabs, mud crabs, stomatopods, and various benthic fishes.

Along the WFS, there are rather distinct fish communities that include species associated with estuaries, open bottom habitat, hard bottom habitat, and the pelagic habitat. These habitats are overlapping and not mutually exclusive. Species are shared among the communities and some transition through these habitats during different life-history stages or seasonally. For example, Spanish Mackerel

(*Scomberomorus maculatus*) are both estuarine and pelagic and Gag Grouper (*Mycteroperca microlepis*) inhabit estuaries as juveniles then migrate through open bottom habitat as young adults to deeper hard bottom as full grown adults.

Over 100 species of fish from 38 families have been observed on central West Florida Shelf reefs between 13-42 m depth, with the families Serranidae (groupers), Lutjanidae (snappers), Pomacentridae (damselfish), and Gobiidae (gobies) being the most species-rich (Smith 1976). Of these, 42 are managed by the GMFMC (GMFMC 2012b), including 17 species of grouper, 14 species of snapper, 5 species of tilefish, 4 species of jack, 1 triggerfish, and 1 wrasse (GMFMC 2011). In addition to the managed species, other species commonly captured in the fishery include a variety of grunts, porgies, and sand perches. For most of the harvested species, juveniles are found inshore with older and larger adults located in deeper water out to the shelf edge. Community structure between warm-temperate species and tropical fish fauna appears to be better defined by bathymetry than latitude with little seasonal differences in the resident adult fish populations (Smith 1976). The recolonization pattern on reefs following a severe summer red tide in 1971 suggest that competitive interactions played an important role in determining community structure (Smith 1979).

The pelagic fish community is made up of several species of predatory fish including three species of mackerel, four species of tuna, Cobia (*Rachycentron canadum*), Tarpon (*Megalops atlanticus*), jacks, and few species of billfish. King Mackerel (*Scomberomorus cavalla*), Spanish Mackerel, and Cobia comprise the coastal migratory pelagic (CMP) management unit while tunas and billfishes are managed at the international level. Most of the pelagic predators are highly migratory and follow the

patterns of several planktivorous “baitfish” species. The pelagic baitfish component includes several species of clupeids (e.g. Spanish Sardine (*Sardinella aurita*), Thread Herring (*Opisthonema oglinum*)) and small carangids (e.g. Round Scad (*Decapterus punctatus*)) that often form dense schools. These schools are concentrated in the south during the winter and distributed more widely and northerly during warmer months. The baitfishes are a major prey item of both reef fishes and migratory pelagics.

Reef Fish Fisheries of the West Florida Shelf

Combined, reef fishes make up an important multi-species, multi-gear, and multi-user fishery in the Gulf of Mexico. Groupers and snappers in particular support some of the most valuable recreational and commercial fisheries in the southeast U.S. and Gulf of Mexico. In 2009, the commercial fishery landed over 6.4 million kilograms of reef fishes on the west coast of Florida for a value of nearly US\$32 million. This accounts for approximately 37% of the total weight and 67% of the total value of commercially landed finfish in west Florida. Of the reef fishes landed by the commercial fishery in west Florida, grouper and snapper made up 87% of the total weight and 92% of the value in 2009 with Red Grouper (*Epinephelus morio*) dominating the catch (31%) and value (32%). During the same year, recreational anglers captured an estimated 3.4 million kilograms of reef fish, with 82% of those being snapper or grouper (Personal communication from the National Marine Fisheries Service, Fisheries Statistics Division Website, Silver Spring, MD).

The fishery is characterized by a regulatory environment that includes several management tools with the potential for a variety of others. Grouper fishing is currently regulated through a combination of recreational bag limits, minimum size limits, commercial trip limits, gear restrictions, annual catch limits, seasonal closures, area

closures, and an individual fishing quota (IFQ). Rule changes proposed in Reef Fish Amendment 30B aim to end overfishing of one species (i.e., Gag) and respond to an improved status of another (i.e., Red Grouper) (GMFMC 2008), while a more recent rule change is intended to reduce the bycatch mortality of loggerhead turtles (*Caretta caretta*) in the longline grouper fishery (GMFMC 2009). Previous stock assessment models for groupers have been specific in estimating cohort dynamics through time but have not been adequate in quantifying the spatial variability observed in population dynamics and fishing exploitation patterns, ecological interactions, and associated socio-economic impacts. Management regulations that are planned or are currently being proposed for several reef fish species on the West Florida Shelf (e.g., IFQs, seasonal/spatial closures, and annual catch limits) present challenges and tradeoffs (both ecological and economical) that have not been fully evaluated in an interdisciplinary manner.

Stock assessments indicate that several species are exposed to unsustainably high exploitation rates, especially in nearshore areas. Development of reliable stock assessment models and effective management plans for reef fishes, groupers in particular, are difficult due to complex life histories and ecological interactions that expose them to varying sources of mortality. Recently, the allocation of landings between user groups has come under increasingly intense debate (Carter et al. 2008; Gentner 2009). Current single-species stock assessment models make very particular assumptions about how natural mortality rates remain stable despite changes in ecological circumstances (e.g., changes in predator/prey abundances and primary production). The potential for these very restrictive and unrealistic assumptions to

mislead fishery managers has led to the recent development of models that account explicitly for at least some major trophic interaction effects.

Study Objectives

One way to address the need for ecosystem based fisheries management is through models. Mathematical trophic-dynamic models are particularly useful because they can assist in determining the tradeoffs associated with harvesting fish from different parts of the food web while also allowing for examination of impacts resulting from changes in primary production and other bottom-up processes. I have developed an Ecopath with Ecosim model of the West Florida Shelf reef-fish fishery and used it to reveal system-wide impacts and tradeoffs of single species policies and to develop new policies that balance tradeoffs without disrupting the trophic structure. This is a multi-species, multi-fleet trophic-dynamic model and is intended to serve the management agency (i.e., GMFMC) in meeting its mandate for ecosystem based management while also providing a framework for hypothesis testing and prioritizing of future research.

The overall goal of this project was to develop a practical fisheries management tool with predictive capabilities that aims to provide a better understanding of the trophic interactions in the West Florida Shelf reef fish community as they relate to fisheries management. I used Ecosim to simulate existing and proposed regulations as well as some hypothetical “what if” scenarios to quantify the potential impacts that these management options have on all species in the system (Chapter 2). I then used the Ecosim policy optimization capability to search for optimal harvest policies that balance the tradeoff between conservation and socio-economic objectives (Chapter 3). To assess the efficacy and impact of marine protected areas I conducted spatial simulations using Ecospace (Chapter 4). Lastly, I identified key data gaps and made

recommendations for research, model development, and management application
(Chapter 5).

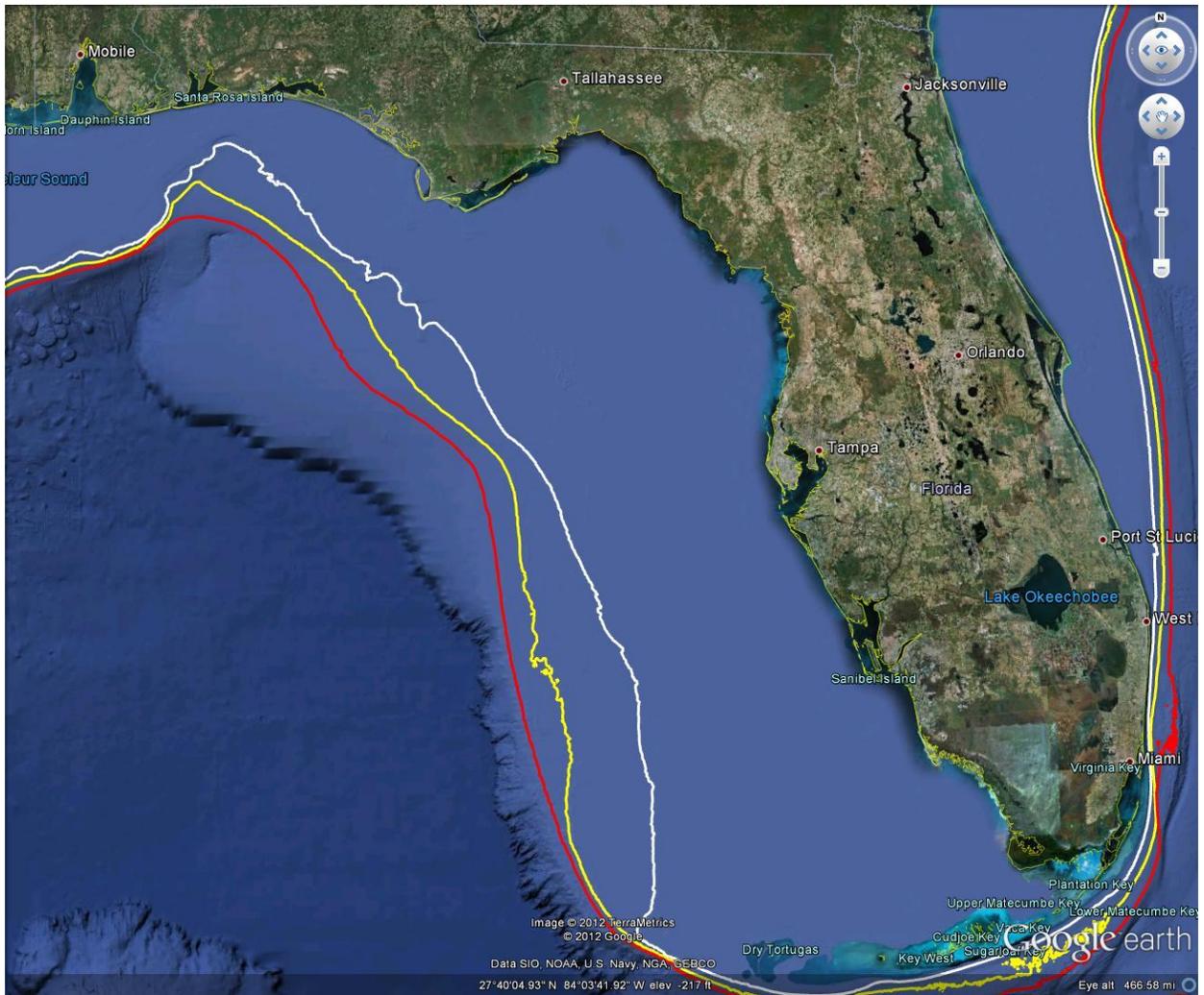


Figure 1-1. Map of the West Florida Shelf. The white line is the 100m isobath, yellow line is the 200m isobath, and red line is the 300m isobath.

CHAPTER 2 TROPHIC INTERACTIONS AND FISHERY HARVEST POLICIES ON THE WEST FLORIDA SHELF

Introduction

Context

A basic tenet of ecosystem-based fisheries management is that species are interconnected and that fishing, along with other human and natural perturbations, has the potential to impact entire ecosystems. The impacts, whether they be induced by fishing or environmental change, can be caused by direct effects such as increased mortality due to fishing or indirect effects that arise from predator-prey interactions and changes to the food web. Fisheries management has relied on single species approaches that typically do not consider trophic interactions or environmental processes. One way to improve fisheries management is to take a more holistic approach that depends upon a better understanding of ecological interactions and the ability to forecast the impacts caused by direct human activities such as harvest, indirect effects on other species, and natural perturbations. Such ecological forecasting has become a common goal for ecosystem-based fisheries management (EBFM) and allows resource managers to make better informed decisions about a diverse set of policy options.

On the west coast of Florida, reef fish species such as groupers and snappers support valuable commercial and recreational fisheries. Over the last 50 years several of these species have been severely depleted and are now under rebuilding plans. Additionally, major reductions in fishing effort for the bottom longline fishery have been imposed in order to limit the bycatch of endangered sea turtles, thereby reducing fishing mortality for a suite of top predators. An increase in the abundance of these predators

is likely to impact other target and non-target species, either directly through higher predation mortality rates or indirectly through competition for food. While these changes are taking place at the upper trophic levels, climate change and warming sea temperatures are altering the productivity at lower trophic levels. To guard against unintended consequences of management actions and failed policy in an ever changing environment, the future impacts of these and similar changes to the ecosystem need to be evaluated.

Ecosystem models are increasingly being utilized as ecological prediction tools because they provide the capability to simulate the entire ecosystem from primary producers to top predators and fisheries. They have been used around the world for various fisheries applications, and all of the National Marine Fisheries Service's (NMFS) regional science centers have adopted some level of ecosystem modeling to address management issues (Plaganyi 2007; Townsend *et al.* 2008). These range from multispecies stock assessment models (Garrison *et al.* 2010; Mueter and Megrey 2006) to coupled climate-population models (Hare *et al.* 2010), trophic-dynamic models (Walters *et al.* 2008), and spatially explicit whole ecosystem models (Kaplan *et al.* 2012b; Link *et al.* 2010). My objective was to develop a trophic dynamic model to simulate the direct and indirect effects of potential management options and environmental change on valuable commercial and recreational species along the West Florida Shelf.

Ecopath with Ecosim Model

Ecopath with Ecosim is a modeling package that facilitates management of basic biomass and trophic interaction data for whole ecosystems and has been widely used for analysis of aquatic resources (Christensen and Walters 2004). The Ecopath

component of EwE is a static mass-balance representation of the food web that essentially provides an initial state for dynamic modeling, and the Ecosim component simulates ecosystem dynamics over time by changing fishing mortality, fishing effort, and environmental forcing functions. The EwE model allows for age structure representation, and it can be fitted to time series of abundance and catch in the same way that single species models are “calibrated” using historical data. The basic data requirements for Ecopath are biomass, total mortality or production rate, consumption rate, diet composition, landings, and discards. Ecopath assumes mass balance between groups based on how production is allocated to fishing, predation, other mortality, and migration.

In Ecosim, biomass dynamics are typically modeled on a monthly time step as a series of differential equations where change in biomass is predicted as consumption minus losses to predation, fishing, and migration (Walters *et al.* 1997). Consumption is predicted based on foraging arena theory which states that prey are divided into states that make them vulnerable or invulnerable to predators and that predation is dependent upon the exchange rate between these two states (Ahrens *et al.* 2012). Partitioning of prey into these two states may arise out of spatial and temporal risk-sensitive behavioral patterns of predators and prey (e.g. hiding when not feeding). The assumption made by foraging arena theory that the bulk of prey are located in refuges at any moment and that feeding times are density dependent results in more stable time dynamics than classical type I functional response assumptions, essentially in the form of type II or III functional response patterns. It is possible in Ecosim to include mediation effects where a third party organism either facilitates or protects against a predator-prey interaction.

Furthermore, environmental forcing functions can be included to drive long term and seasonal patterns of primary production.

I constructed the model with the intention of using it for fishery management applications. My goal was to predict the impact and performance of a suite of management scenarios proposed or recently implemented in the eastern Gulf of Mexico. The forecasts generated by this model evaluate strategic management objectives and reveal whether or not proposed policy options will lead to any unintended consequences on other species. The objective was to evaluate the impact of strategic management changes (reductions in landings, effort, etc.), and I did not consider tactical regulatory options such as bag limits, size limits, seasonal closures, or gear modifications that might be needed in order to achieve desired strategic changes. I focused on the direct effects of fishing and the indirect effects that occur due to trophic interactions (predation and competition). While the model is capable of producing biomass estimates with some precision relative to those estimated by single species stock assessment models, the emphasis of these simulations is on the direction and magnitude of change. To date, few attempts have been made to evaluate the impacts of harvest policies on the fisheries and ecosystems in the Gulf of Mexico (Okey *et al.* 2004; Walters *et al.* 2008), and I intend for this model to provide quantitative, predictive capabilities to fisheries managers and stakeholders in this region.

Materials and Methods

Model Description

I developed the model around regulated species in the eastern Gulf of Mexico including reef fishes (groupers and snappers), coastal migratory pelagics (mackerels), and highly migratory pelagics as defined by the GMFMC and NMFS. The area modeled

extended from the Florida Panhandle south to a boundary that excludes the Florida Keys. Particular emphasis was given to select groupers and snappers that inhabit reefs on the West Florida Shelf and support valuable commercial and recreational fisheries. These were represented in the model by 2 or 3 age stanzas to capture basic ontogenetic changes in diet, habitat, and fishery selectivity. Because several reef fishes were further divided into age-stanzas, coastal and inshore species were included because they interact with reef fish juveniles that have not yet migrated to offshore reefs. Aggregate groups of non-target fishes, invertebrates, zooplankton, and primary producers are necessary for a complete food web and were also included. The resulting model consisted of 70 biomass pools including one each for dolphins and seabirds, 43 fish groups (of which 11 are non-adult life stages), 18 invertebrate groups, 4 primary producers, and 3 detritus groups (Table 2-1).

Biomass (B , mt/km²) values were taken from single species stock assessments, estimated by dividing observed catch by assumed fishing mortality ($B = C/F$), or derived from survey data. Production rate (P/B) or instantaneous total mortality (Z) was calculated by adding an assumed natural mortality to the fishing mortality from stock assessments or using empirical equations for mortality (Pauly 1980; Ralston 1987). Estimates of consumption (Q) were derived empirically using equations that incorporate data on morphometrics, ambient water temperature, and diet (Palomares and Pauly 1998; Pauly 1989). Diet compositions of fish were estimated by combining data from the Florida FWC Fisheries-Independent Monitoring program's trophic database with information available in the literature. Stomach contents data from field collections and diet descriptions provided in the literature were combined for each species using a

weighted average, where the weighting scheme included number of stomachs, geographic location, and quality of the data (e.g. quantitative vs. qualitative). Much detail was put into the derivation of parameters for invertebrates in an earlier WFS model (Okey and Mahmoudi 2002; Okey *et al.* 2004), therefore I used those values as initial input for this reef fish centric model.

The fishery included four recreational (shore based, private boat, charter boat, and headboat) and nine commercial fishing 'fleets' (vertical line, bottom longline, pelagic longline, pelagic troll, gill/trammel nets, cast nets, purse seines, trawls, fish traps, and crab traps). Commercial landings were obtained from the Florida Trip Ticket program and NMFS and commercial discard information was obtained from a national bycatch report (NMFS 2011) and other observer programs (Passerotti and Carlson 2009; NMFS-SEFSC, personal communication; FWC-FWRI, personal communication; Pierce *et al.* 1998). Recreational landings and discards were made available by the Marine Recreational Fisheries Statistics Survey, and headboat landings were obtained from the NMFS Beaufort Headboat Survey. After entering the required input data, the Ecopath model was "mass-balanced" by making small adjustments to diet, mortality, and biomass so that fishing and predation mortality rates did not exceed assumed or estimated total mortality.

Model Calibration

Before making future predictions, it was important to use parameter estimation methods to insure that the Ecosim model can at least reasonably represent past trajectories of biomass for fish species in the model. This was done by fitting the model to time-series of observed trends in abundance and catch over the time period 1950-2009. Because the calibration simulation began in 1950, biomass, catch, and total

mortality parameters from the present day Ecopath model were rescaled to represent a historic (1950s) condition (Table 2-2). This involved increasing biomass, reducing catch, and reducing total mortality to a level closer to natural mortality. In most cases, the stock assessment or time series data provided the information necessary to make such adjustments. Diet compositions remained mostly the same in the present and historic models.

Time series were obtained directly from stock assessments or taken from fisheries independent and other various survey data. I used nutrient loads from the Mississippi River to drive phytoplankton production on the West Florida Shelf over the entire time period (Goolsby and Battaglin 2000; Aulenbach *et al.* 2007). Chlorophyll-a production along the West Florida Shelf is dependent upon a variety of factors including outflow from the Mississippi and smaller rivers in northwest Florida along with upwelling events and transport mechanisms related to wind, loop current excursions, steric differences between shelf and offshore waters, and the Coriolis effect (Castillo *et al.* 2001; Gilbes *et al.* 2002; Gilbes *et al.* 1996). It is impossible to construct a historical time series of chlorophyll-a that incorporates all of these processes, thus the Mississippi River nutrient flux served a proxy that was assumed to capture the major changes that have taken place in primary productivity over the last 60 years (Figure 2-1).

The most important parameters when calibrating Ecosim models are the vulnerability exchange rates (v_{ij}). The vulnerability parameters represent the rate that prey move from invulnerable to vulnerable states, and there is one parameter for each predator-prey interaction. Vulnerability parameters are essentially multipliers on predation mortality rates. For example, a vulnerability of 2 implies that a predator could,

at most, double the predation mortality exerted on its prey when predator biomass increases. Very low vulnerability exchange rates imply 'bottom-up' control of predation rates because changes in predator abundances do not result in drastic changes to prey mortality rates. High vulnerability exchange rates imply 'top-down' control where consumption increases dramatically with predator abundance and therefore produces more dynamic predator-prey interactions. Other parameters are available to modify how quickly organisms adjust the time they spend foraging, the sensitivity of other mortality to changes in feeding time, the effect that predators have on feeding time, and prey switching. Overall, the model is less sensitive to these parameters but they do provide the ability to stabilize consumption rates, cause density dependent natural mortality, represent varying degrees of recruitment compensation, simulate risk sensitive foraging behavior, and adjust the responsiveness of predators to prey abundances. The vulnerability parameters thus play the same basic role in Ecosim as parameters for the steepness of stock-recruitment relationships in single species assessment models.

To fit the model to time series, manual adjustments were made to the foraging arena parameters, especially the vulnerability exchange rates, to correct for any gross divergence from the data. For example, groups that were having biomass drop to zero required that v_{ij} 's be reduced or feeding time varied to cause stronger compensatory improvements in survival at low stock sizes. After correcting for any obvious errors, an automated search was executed that adjusts the v_{ij} 's to minimize the sum of squared deviation between predicted and observed data. This process was repeated iteratively, focusing on the group with the poorest fit, until the model was able to at least reproduce

the major patterns in biomass and catch over the entire time period for all groups. As a further diagnostic, I compared the F_{msy} values from Ecosim to those estimated by single species stock assessment models. F_{msy} was estimated by incrementing fishing mortality rates and running Ecosim to equilibrium while holding other groups stationary. This was done to correct for spurious parameter estimates obtained for some groups during the calibration process, which led to unrealistic estimates of biological reference points for reasons such as lack of adequate contrast in historical biomass trend data.

Policy Screening

To conduct forward projecting policy simulations with the present day model, I rescaled the vulnerabilities from the calibrated historical model so that maximum possible predation mortalities were the same in both the 'historic' and 'present day' versions. This was done by multiplying each vulnerability exchange rate V_{ij} (for vulnerability of prey i to predator j) from the historic model by the ratio of historic to present day predation mortality rates for the same predator prey interaction

$$\hat{V}_{ij} = V_{ij} * \frac{M2_{ij}}{\widehat{M2}_{ij}}$$

Where \hat{V}_{ij} is the rescaled vulnerability for the present day model, $M2_{ij}$ is the predation mortality from the historic model and $\widehat{M2}_{ij}$ is the predation mortality from the present day model. Biomass accumulation (BA) was also added to the model to account for initial rate of change of biomass in the current (initial for forecasting) year. BA was calculated from the change in biomass over the last year of the historical simulation. All parameter values estimated for the historic (calibrated) and present day models were saved in access databases, which can be examined and changed using

the EwE 6.4 user interface. These models are available at the dropbox link https://dl.dropboxusercontent.com/u/31499559/WFS_EwE_files.zip.

I evaluated a set of policy options over a 20-year projection period using the present day model with vulnerability exchange rates rescaled as described above. In some cases I simulated actual harvest policies that are currently proposed or were recently implemented, while in others I prescribed some relevant, but hypothetical, “what-if” scenarios. This was done by varying the fleet fishing efforts, fishing mortality rates of species, or primary production forcing functions. For policy optimization tests, fishing effort was held constant, at either prescribed test values or Ecopath base level throughout the duration of each scenario and did not respond to changes in abundance of target species. Using the ‘present day’ Ecosim model, I simulated the impact of each of these scenarios on biomass of all species in the model and focused mainly on valuable recreationally and commercially harvested species.

For each scenario, I conducted a deterministic run over 20 years using the base parameterization representing 2009 conditions, and then carried out 100 Monte-Carlo simulations to establish sensitivity of the model predictions to uncertainty about Ecopath parameter values. The Monte-Carlo simulation proceeds by randomly selecting a biomass value for each species from a uniform distribution where the mean is the 2009 base value and the upper and lower limits were defined based on knowledge about uncertainty in the source data and the biomass estimates derived from them. In each Monte-Carlo trial, a random draw was made and the model was checked for mass balance. If the parameter combination produced a model that was not mass-balanced then it was discarded and another draw was made. Only the trials that did not violate

mass-balance were used as starting points for a Monte-Carlo trial in Ecosim. For each 20 year simulation, I calculated the percent change in terminal year biomass relative to the biomass at the end of the status quo scenario along with standard errors calculated from the 100 Monte-Carlo simulations.

Status quo simulation

The status quo scenario held fishing mortality rates and fishing effort constant at 2009 levels for 20 years. In 2009 Red Snapper (*Lutjanus campechanus*), Greater Amberjack (*Seriola dumerili*), and Gag (*Mycteroperca microlepis*) were all determined to be overfished and undergoing overfishing (NMFS 2011). Red snapper had been overfished and undergoing overfishing since the late 1980s, and a rebuilding plan was first implemented in 1990. By 2009, fishing mortality on Red Snapper had been reduced and the stock was showing signs of recovery (SEDAR 2009c). Greater Amberjack have been under a rebuilding plan since 2003, but both the 2006 and 2010 stock assessments concluded that the stock remained overfished and undergoing overfishing (SEDAR 2006b; SEDAR 2010a). In 2009, fishing mortality rates of Greater Amberjack were determined to be low enough to allow the stock to recover but not within the 10-year time frame mandated by the rebuilding plan (GMFMC 2012a). Gag grouper were determined to be overfished and undergoing overfishing in 2009 necessitating a plan to rebuild the stock to a sustainable level within 10 years (SEDAR 2009a). In contrast, Red Grouper (*Epinephelus morio*) were neither overfished nor undergoing overfishing in 2009 and stock status has been improving (SEDAR 2009b). Thus, the status quo scenario simulated the continued increase in biomass for species that were recovering as of 2009 (e.g. Amberjack, Red Grouper, Red Snapper) and

declining stock sizes for species whose fishing mortality rates had not yet been reduced below the overfishing limit (i.e. Gag).

Rebuilding Gag stocks

The 2009 stock assessment for Gulf of Mexico Gag determined the stock to be overfished and undergoing overfishing (SEDAR 2009a). The Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA) required overfishing to end within 2 years and the stock rebuilt to sustainable levels within 10 years. Reef fish amendment 32 was developed to address the overfished status and develop a stock rebuilding plan for Gag (GMFMC 2011). The rebuilding plan called for reducing fishing mortality rates to a third of 2009 levels which would be achieved through a combination of larger size limits, smaller bag limits, and/or shorter seasons. The baseline fishing mortality of Gag was 0.52 in 2009 therefore fishing mortality on Gag was set at $F_{rebuild}$ of 0.16 for the duration of the simulation.

Longline effort reduction

Amendment 31 to the Gulf of Mexico Reef Fish Fishery Management Plan aims to reduce the bycatch of sea turtles in the bottom longline component of the commercial reef fish fishery (GMFMC 2009). In 2005, the allowed take of the endangered loggerhead sea turtle by bottom longlines was set at 85 turtles over a three year period (NMFS 2005). A 2008 NMFS report indicated that bottom longline gear took between 339 and 1,884 loggerheads over an 18 month period (NMFS 2008). To reduce the frequency of interactions between bottom longlines and sea turtles, reef fish amendment 31 prohibited the use of bottom longline gear east of Cape San Blas at approximately the 35-fathom depth contour from June through August, reduced the number of longline vessels to those with an annual average landings of 40,000 lbs

during 1999-2007, and restricted the number of hooks per vessel to 1,000 of which only 750 may be fished at a time. The overall reduction in effective effort based on these regulations is expected to be between 48 and 67%. Although sea turtles are valued as charismatic species and often trigger management actions as mandated by the Endangered Species Act, I did not include them in the model because they do not play a key role in the food web dynamics of groupers and snappers. I assessed the impact of a 60% effort reduction in the bottom longline fishery on other species in the system.

Increase exploitation of baitfishes

On the West Florida Shelf, Scaled Sardine (*Harengula jaguana*), Spanish Sardine (*Sardinella aurita*), Atlantic Threadfin Herring (*Opisthonema oglinum*), Round Scad (*Decapterus punctatus*), and other less dominant clupeids and small carangids are commonly referred to as the 'baitfish' complex. These small pelagic planktivores make up an important forage base on the West Florida Shelf for pelagic predators as well as large-bodied reef fish. When compared to the industrial Menhaden (*Brevoortia patronus*) fishery in the northern Gulf, the 'baitfish' fishery in Florida is relatively small. Commercial landings of baitfish peaked in 1989 at about 30 million pounds, whereas Gulf Menhaden landings often exceed 1 billion pounds per year. While current and historic stock sizes and fishing mortality rates of baitfish are unknown on the West Florida Shelf, I assumed that a fishing mortality rate of 0.02 is reasonable given the size and scale of the fishery. During 1989, effort and catch in the baitfish fishery were both almost 20 times higher than they are currently. To evaluate the impact of harvesting baitfish at historically high levels, I simulated a 20-fold increase of effort in the purse seine fishery, which generated a fishing mortality rate of about 0.40 for the baitfish complex. For comparison, gulf menhaden have recently been fished at a level of about

0.5 but historically experienced fishing mortality rates higher than 1.5 (Vaughan *et al.* 2007).

Changes in primary production

How complex marine ecosystems will respond to climate change is uncertain and there are plausible hypothesis for both lower and higher overall productivity under a warmer environment. Warming sea temperatures have been shown to decrease phytoplankton productivity by reducing mixing throughout the water column and lowering nutrient supply (Behrenfeld *et al.* 2006; Doney 2006). Severe and prolonged droughts will reduce delivery of nutrients from freshwater sources and lower the productivity in coastal and estuarine areas (Wetz *et al.* 2011; Wiseman *et al.* 1999). Alternatively, it has been hypothesized that increased productivity may occur in warmer, more stratified waters due to enhanced atmospheric nitrogen fixation at the surface (Karl *et al.* 1997). To investigate how the ecosystem might respond to broad changes in productivity, likely driven by global warming and climate change, I considered two simple production change scenarios. In one scenario I allowed phytoplankton productivity (P/B) to increase 1% each year for 20 years, in the other productivity decreased 1% each year over the duration of the simulation. Seasonal and inter-annual variability in phytoplankton production is sure to impact marine ecosystems in a more complex manner, but I used linear forcing functions to simplify the analysis and interpretation of results.

Sensitivity scenarios

To provide some perspective and contrast to the policy simulations, I explored some simple scenarios in which fishing effort for all fleets in 2009 was multiplied by 0, 0.5, 1.5, 2 and 5. This was meant to provide context for other simulations and to

demonstrate model behavior under extremely low and high fishing mortalities. It is also useful as a diagnostic to identify whether the predicted unfished biomass of any group is unrealistic or if the group is overly sensitive to fishing and crash when exposed to modest increases in fishing pressure.

Results

Calibration

The model was capable of reproducing historical trends in abundance and catch quite well for the period 1950 to 2010 (Figure 2-2 and Figure 2-3). The patterns of biomass and catch were simulated forward from 1950 using only input values of known fishing effort, fishing mortality, and primary productivity. Because there were no year-specific parameters that can be adjusted to provide better fits to the data, a poorly parameterized model or one that is missing, or misrepresenting, some key functional relationship could easily diverge from observed data. While it was possible to obtain reasonable fits under multiple parameter combinations, incorrect parameterizations would usually lead to unreasonable predictions of unfished biomass and equilibrium F_{msy} , which were also used as diagnostics when determining the base model configuration. I attempted to calibrate the model to the entire available time series, but was less concerned about poor fits during earlier years when there was little information available on catch and biomass trends. Rather, the historical period provided a useful perspective about the gross changes that have taken place over the last 60 years and the model did not diverge from those.

Policy Screening

Status quo

The non-stationary behavior in the status quo simulation (solid black line in Figure 2-4) was a result of biomass accumulation rates calculated from the change in biomass that occurred over the last year of the historic, calibrated simulation. That is, groups that were declining (or increasing) through 2009 of the historic model continued to decline (or increase) through the projection period. Biomass accumulation rates for most species were positive, leading to increasing stock sizes reflective of generally more conservative management in recent years due to stock rebuilding plans.

Gag Grouper, which was overfished and undergoing overfishing in 2009, was predicted to continue declining to approximately 50% of current stock sizes to a biomass of 8.32 million pounds (Table 2-3). Biomass of Red Grouper was predicted to be about 20 million pounds higher in year 20, a 28% increase from 2009. Under status quo, Red Snapper (27% increase) continue to recover because of reduced F achieved by their rebuilding plans. Biomass was predicted to increase for Black Grouper (*Myceteroperca bonaci*) (16%), Greater Amberjack (8%), Yellowedge Grouper (*Epinephelus flavolimbatus*) (14%), Goliath Grouper (*Epinephelus itajara*) (40%), Vermilion Snapper (*Rhomboplites aurorubens*) (7%), and Triggerfish (*Balistes capriscus*) (14%) while Tilefish (*Lopholatilus chamaeleonticeps*) were predicted to decline by -5%. The other shallow water grouper (SWG) and deep water grouper (DWG) species also increased over the 20 year simulation, benefitting from stricter regulation on weaker stocks. King Mackerel (*Scomberomorus cavalla*), Spanish Mackerel (*Scomberomorus maculatus*), and Cobia (*Rachycentron canadum*) make up the coastal migratory pelagic (CMP) management unit. All of the species in the CMP

increased in the status quo simulation with King Mackerel increasing by 22%, Spanish Mackerel by 17%, and Cobia by 13%. Dolphins and seabirds showed only little change in biomass while the sardine-herring-scad baitfish complex declined by -9% as predators recovered.

Rebuilding Gag stocks

Under the fishing mortality rate of 0.16 Gag increased over its 2009 biomass by 68% to 28 million pounds (mp) in year 10 which was 3.4 times larger than biomass predicted under the overfishing condition of the status quo simulation. One hundred Monte Carlo trials produced stock biomass estimates ranging from 20-38 million pounds (Figure 2-5). For reference, the single species stock assessment projected spawning stock biomass of Gag to be 22 and 30 million pounds after 10 years under $F_{rebuild}$ of 0.19 and 0.14 respectively (SEDAR 2009a). The impacts on other species were modest with most biomasses remaining within +/- 5% of the status quo (Table 2-3). Black Grouper, other SWG, Vermilion Snapper, King Mackerel and Black Seabass (*Centropristis striata*) all were impacted negatively by the Gag recovery, by at least 5%. Black seabass biomass when Gag are rebuilding was predicted to be 20% lower than the status quo. The contribution of Black Seabass to the diet of Gag was just 1% and the baseline predation mortality rate was 0.15/yr. The vulnerability exchange rate setting of 10 for this interaction would allow predation mortality of Gag on Black Seabass to increase by up to 10 times the baseline rate. Total predation mortality rates increased from 0.55/yr in 2009 to 0.58/yr in the status quo simulation and to 0.64/yr in this scenario, an increase of about 10%. Greater Amberjack showed a potential 5% decline in biomass from the status quo as a result of this policy. This arises from predicted

competition with groupers for sardine herring, small coastal fishes, and small reef fishes.

Longline effort reduction

The impact of a 60% reduction in bottom longline effort had direct positive effects on biomass of several reef fish species including Red Grouper, Tilefish, Yellowedge Grouper, and Gag. The deepwater species (Tilefish and Yellowedge Grouper) are caught almost exclusively by longliners because other fleets and recreational anglers rarely operate in such deep water. Therefore their biomass increased by over 65% following the reduction. Longlines account for the largest portion of fishing mortality on Red Grouper and this policy scenario resulted in an additional 20% increase in biomass from status quo. In contrast, Gag biomass was only 5% higher because most of their mortality is caused by the recreational sector. Vermilion snapper and Black Seabass biomass were lower than the status quo by -7% and -13% respectively. This was driven by an increase in predation mortality from other groupers as well as competition for food, especially with the sub-adults. The biomass of the baitfish complex was -6% lower than status quo and stochastic sensitivity runs (Monte Carlo trials with randomly varying Ecopath parameters) indicate that it could be as low as -11%. Consequently, the impact on pelagic species was negative but within -5% for the base scenario. The 95% confidence intervals from stochastic runs were centered on zero for Spanish Mackerel and Cobia indicating the impact, though small, could be in either direction. The diets of Spanish Mackerel and Cobia have less overlap with adult groupers than King Mackerel and therefore exhibited little to no impact.

Increase exploitation of baitfish

Increasing effort in the purse seine fishery by 20 fold, to historic highs, reduced the biomass of baitfish by 23% from status quo (Table 2-3). The biomass of other shallow water grouper (SWG), Red Snapper, Vermilion Snapper, Amberjack, King Mackerel, dolphins and seabirds were between 5-12% lower after 20 years of harvesting baitfish at $F = 0.40$ than under status quo conditions. Monte-Carlo simulations showed the impact on predators to be no more than about 15%. The base model predicted a greater than 5% increase in biomass for other deep water groupers (DWG), and Triggerfish and mostly positive MCMC values for Goliath Grouper, Tilefish and Cobia. These species all have diets largely composed of prey items other than sardines, herring, or scad and therefore benefit from lower predator abundances without experiencing a decline in their main food resource.

Changes in primary production

As expected, the model predicted widespread reductions in fish biomass as productivity declined, and higher biomass when it increased. Under a lower production regime, biomass of Red Grouper, Red Snapper, Vermilion Snapper, Amberjack, and Triggerfish were all at least 5% less than the status quo with Red Snapper and Vermilion Snapper biomass reduced by as much as -19% and -16% respectively. The biomass of the coastal migratory pelagics was predicted to be -17% lower than status quo for King Mackerel and -12% for Spanish Mackerel, a stronger impact than predicted for the reef fish component. Pelagic species such as the sardine-herring complex and anchovies are expected to be more tightly coupled to changes in phytoplankton abundance than those associated with the benthos. The impacts of reduced productivity on pelagic associated prey groups were -15-17% whereas the impacts on

benthic associated prey species such as grunts, pinfish, shrimp, and crabs were between 0 and -12%. Of the fish listed in Table 2-3 only Goliath Grouper, Tilefish, Black Seabass, and Cobia had Monte Carlo simulation trials where biomass improved under lower primary production relative to status quo. They benefitted because they all have a large benthic component to their diet, and especially in the case of Black Seabass, would also benefit from reduced predation and competition.

Sensitivity scenarios

Model predictions under extreme hypothetical fishing pressures were consistent with expectations based on single-species models. In the no fishing scenario, biomass of most harvested species was higher by 2 to 8 fold over the status quo (Table 2-3) with the magnitude determined by how heavily the species is exploited as well as impacts due to predation and competition. Prey species and those at a competitive disadvantage declined in biomass. Sardine-herring declined by about -20% as predation mortality increased while biomass of dolphins and seabirds was predicted to be -11% and -13% lower respectively, due to increased competition. In contrast, under the high fishing scenario prey species, birds, and dolphins increase in biomass while most of the predators were fished to very low abundances. With most predators removed, the forage species could easily withstand a five-fold increase in fishing mortality and, assuming no negative interactions with fishing gear, allow for unfished predators such as birds and mammals to flourish.

Discussion

The overall conclusion from these Ecosim simulation tests is that there are winners and losers in all policy options. Management options oriented towards a single species, such as rebuilding an overfished stock, had less widespread and more modest

impacts (+/- 5%) than policies that impacted a suite of species. Simulations which involved perturbations to the middle of the food web or changes to primary production had more drastic impacts over a broader set of species. Excluding the sensitivity runs, none of the harvest policies or environmental conditions I considered was predicted to cause any species to collapse.

The model illustrated the direct and indirect impacts that occur between and within predator and prey groups. Reducing fishing mortality on predators caused a decline in baitfish due to increased predation that then led to a decline in unfished predators such as dolphins and birds. Alternatively, scenarios in which fishing mortality was increased had the opposite effect with reduced predator abundance leaving more food available (baitfish) for birds and dolphins. Similarly, a higher abundance of reef fish, resulting from either rebuilding plans or reduced effort in the longline fishery, tended to cause slight declines in coastal migratory pelagics and Amberjacks.

Competitive interactions are believed to strongly influence reef fish community structure in the eastern Gulf of Mexico (Smith 1979). Two managed reef fish species that appeared to be at a competitive disadvantage were Vermilion Snapper and Black Seabass. Their biomasses were consistently lower under policies that allowed for the increase of one or more upper trophic level reef fish species. This is consistent with observations made on artificial and experimental reefs in the northeastern Gulf of Mexico. For instance, Black Seabass were observed in higher densities on experimental reefs where Gag were excluded (Lindberg *et al.* 2006) and Vermilion Snapper became more abundant after a decrease in resident piscivores (groupers) on artificial reefs (Dance *et al.* 2011). Ecosim only predicts the impacts due to trophic

interactions and ignores any competition for shelter. There are opposing hypothesis about the role of competition for habitat in structuring reef fish communities (Sale and Williams 1982) and there is conflicting evidence for habitat limitation in the Gulf of Mexico (Bohnsack 1989; Grossman *et al.* 1997; Shipp and Bortone 2009). Assuming that Vermilion Snapper and Black Seabass do compete with other higher trophic level species for habitat as well as food, and that they are at a disadvantage in those interactions, I would expect the impacts to be greater than those predicted from trophic interactions alone. Other species for which the model predicted little to no impact because of little overlap in diet or low predation mortalities, could in fact be affected by competition for space not accounted for in the Ecosim model.

Divers in the northeastern Gulf of Mexico have observed a school of Amberjack driving prey downward towards the reef where Gag were waiting to feed (Stallings and Dingeldein 2012). It was possible to simulate this facilitative feeding interaction using a mediation function that allowed for sardines to become more vulnerable to Gag when Amberjack biomass increases. Under this configuration, the biomass of Gag nearly doubled after 20 years of fishing at $F_{rebuild}$, (compared to a 70% increase without mediation), because the recovering Amberjack stocks enhanced Gag consumption of baitfish through facilitative feeding. The observed feeding event also illustrates that separate foraging arenas can exist for multiple predators over a single prey resource, water column and reef in this case. This provides support for the assumption in Ecosim that foraging arenas are unique for each predator-prey interaction, but also support for the possibility that vulnerability exchange rates can be influenced by multiple species that pursue the same prey in different microhabitats (e.g. at different depths).

Fish abundance trajectories were expected to vary substantially among species but were similar to stock sizes predicted in the stock assessments for most species (SEDAR assessments available at www.sefsc.noaa.gov/sedar/). Status quo biomasses were similar in magnitude and direction to projections made with single species models for Gag, Red Grouper, Red Snapper, Greater Amberjack, Yellowedge Grouper, Goliath Grouper, and Vermillion Snapper. The similarity between status quo forecasts made by Ecosim and stock assessment models should not come as a surprise because many groups were calibrated using historical data generated by the stock assessment and are therefore expected to have similar biomass dynamics. While this does not validate the model, it does facilitate direct comparison between predictions made by Ecosim and single species models and allows us to characterize environmental uncertainty not captured by the single species model. Divergence between predictions made by Ecosim and single species models (i.e. Cobia and King Mackerel) could be due to incorrect parameter estimates, especially the vulnerability exchange and biomass accumulation rates, or failure of the single species model to capture an important environmental process.

The predicted response of predators to forage fish depletion are consistent with, and perhaps slightly more conservative, than those made by other ecosystem models. When half of a predator's diet is composed of forage fish, ecosystem models tend to predict a 20-40% loss of predator biomass when forage fishes are reduced to between 80-40% of their virgin stock sizes (Pikitch *et al.* 2012). The model described here predicted predator biomass to decline by at most 15% when baitfishes were harvested at the historic high rates of the 1980s. A diverse prey resource, such as that available

in the Gulf of Mexico, would likely lessen the impact by allowing predators to switch to prey that are more abundant and opportunistic species to replace niches left behind by species targeted in the fishery.

Changes in primary production were predicted to have rather large effects on the biomass of fish along the WFS. The magnitude of the impacts predicted in this scenario are consistent with those predicted in a suite of Ecosim models from Australia (Brown *et al.* 2010). They showed that predicted consumer biomass was proportional to primary production rates and that a 20% change in primary production would lead to a similar change in biomass. The analysis presented here demonstrates that changes in primary productivity will cause predictable changes in the biomass of most marine organisms. However it is highly unlikely that future change will be without variability and that the responses will be so predictable. For example, an increase in phytoplankton biomass could lead to declines in submerged aquatic vegetation (Okey *et al.* 2004) or cause widespread hypoxic zones (Breitburg 2002; Diaz and Rosenberg 2008; Justic *et al.* 2002) that subsequently have negative and non-linear effects on marine organisms. Other changes to the Gulf of Mexico ecosystem brought on by climate change such as ocean acidification, lower dissolved oxygen, warming sea temperatures, and shifts in species ranges are likely to interact with changes in primary productivity. Ainsworth *et al.* (2011) demonstrated that the cumulative impact of these stressors in the Northeast Pacific was greater than the impact of each considered in isolation. Nevertheless, these simulations offer some insight into how bottom-up processes impact the entire ecosystem and provide a framework for forecasting more comprehensive and detailed climate change scenarios.

This model, like all ecosystem models, is a simplified representation of a far more complex system. In order for the model to be useful in management I attempted to strike a balance between simplicity and complexity that captured what I believed were the major ecological processes while still being flexible, functional, and interpretable. There are several caveats and limitations associated with the EwE approach and those are described elsewhere (Christensen and Walters 2004). Here, I list a few of the more important caveats to consider when interpreting the results from the simulations.

I considered each scenario in isolation and did not evaluate whether impacts of multiple stressors would simply be additive or if synergistic processes would compound the impacts. This was by design as my intention was to demonstrate how specific policy options have impacts on other components of the system. In practice, policies are typically drafted and evaluated individually and therefore my results fit into that management model. Ecosystem models provide a platform to simultaneously evaluate a suite of policy options across multiple species and fleets while also accounting for key environmental drivers (Ainsworth *et al.* 2011; Kaplan *et al.* 2012a). Evaluating the cumulative and synergistic impacts of the entire management portfolio is the next logical extension of this model.

This model gave no consideration to the spatial distribution of fish and fisheries and therefore no representation of competition for habitat or any spatial regulations that are part of the policies considered. For example, effort reduction in the longline fishery is achieved through a combination of regulations, one of them being spatial closures and depth restrictions. Prohibiting longlines in depths less than 35 fathoms, as is the rule in Amendment 31, will therefore not benefit the deep water species (Yellowedge

Grouper, Tilefish, other DWG) as drastically as was predicted in this study. Spatially explicit models should provide a more realistic forecast for this regulation. In the Gulf of Mexico there is concern that recovery and range expansion of some species (i.e. Red Snapper) could have negative impacts on others through competition for food and habitat (GMFMC 2013 and associated public hearing comments). The absence of spatial structure in Ecosim does not allow us to account for those effects. Furthermore, it would be difficult to separate the impacts arising from trophic interactions from those arising out of competition for habitat because habitat selection and food availability are so intertwined. This is an area that needs to be addressed through both experimental field work and further simulation modeling.

There was no simulated response of fishing effort to changing stock sizes and no simulated policy adjustments made to fishing mortality rates when stock sizes met or fell below some target level. In reality, as stock sizes increase more people will likely be attracted to at least the recreational fishery and as stocks decline they will fish less, switch to other species, or stop fishing altogether as profitability declines. The responsiveness of anglers to increasing and decreasing stock sizes has been shown to influence the probability of overfishing (Allen 2012; Hunt *et al.* 2011). Ignoring economic forces and fisher behavior can have biological consequences and alter the results of policy projections (Wilens *et al.* 2002) which would surely have ecological implications. Walters and Martell (2004) offer several options for modeling complex effort dynamics in fisheries over short and long time scales, one of which is available in Ecosim. I did not implement the effort dynamics model here because it does not enable prescription of specific policy options to individual fleets. Additionally, harvest control

rules specify management actions (F, effort, or catch) based upon current stock status. For example, when a rebuilding plan achieves its goal managers will be pressured to ease-up on restrictions, and vice versa. This feedback in management can be modeled in Ecosim through “closed-loop” policy simulations where the fishing mortality at each time step is based on the stock size estimated at the previous time step; such simulations were not used in this initial part of the modeling study.

There are two basic sources of uncertainty in ecosystem models, uncertainty associated with the data used to derive input parameters and uncertainty associated with model structure. Structural uncertainty manifests itself in the definition of biomass pools, functional relationships, and choice of environmental drivers (Pauly *et al.* 2000). For example, over-aggregation of groups could lead to stronger competitive interactions and ignoring any important environmental or mediation effects could impact model calibration and prediction. This type of uncertainty was addressed early on during model development through a series of internal reviews, iterative improvements, and a biological workshop where the model was reviewed by a group of scientists familiar with the WFS and reef fish species. Ultimately, I sought to strike a balance in model complexity while capturing the major drivers on the WFS.

Regarding data, the most critical sources of uncertainty probably occur for the diet compositions of large bodied, offshore predators and biomass estimates of the forage species. The diet matrix was constructed using data from more than 40,000 stomachs from over 200 species either collected by the Florida FWC or available in literature. Despite these efforts, data gaps do exist, particularly for large predators that reside on structured habitats in deep water. Quality stomach contents from deep water reef

species are difficult to obtain due to barotrauma which can lead to stomach eversion and the inability to sample with active, non-baited gear. While collection of stomach contents in concert with routine fisheries independent sampling is ongoing, targeted studies may be necessary to increase sample sizes across all sizes, seasons, and geographic regions.

One of the most ecologically important groups in this system is the sardine-herring-scad “baifish” complex, yet there is considerable uncertainty about their biomass estimates. Houde (1976) estimated the biomass of baitfish (Thread Herring, Scaled Sardine, Round Herring, and Spanish Sardine) from egg and larval surveys in the eastern Gulf of Mexico to be nearly 1 million tons during the early 1970s. Recent estimates based on fishery independent trawl and acoustic surveys conducted offshore of west-central Florida are around 50,000 and 800,000 tons respectively. I used an estimate of about 630,000 tons, which is close to the biomass derived from acoustic surveys and falls somewhere between the trawl-based and egg/larvae-based estimates. Nearly every commercially and recreationally important species utilizes the baitfish resource to some extent, therefore it is important to gain a better understanding of their abundance and productivity.

Monte Carlo simulations were conducted to characterize the uncertainty in projections associated with Ecopath biomass input values. I recognize that uncertainty does exist in other Ecopath parameters such as production, consumption, and biomass accumulation rates. However, there is little information available to characterize the uncertainty with those parameters and I assumed that these bioenergetic rates were less uncertain than biomass measurements. Ideally, the uncertainty associated with

diet compositions and vulnerability settings should also be incorporated into stochastic model projections. The former could be accomplished using the Monte Carlo approach following a statistical evaluation of diet data in which confidence intervals are generated for each predator-prey interaction (Ainsworth *et al.* 2010). Performing model simulations while including uncertainty in vulnerability exchange rates would be less straightforward because there are no data for which to define confidence bounds.

Even with the caveats and uncertainties, there is great utility in EwE and other food web models because understanding the impacts of trophic interactions and environmental change can only improve upon single species management. In general, the model does indicate that trophic effects in the food web are potentially strong and can lead to ecological tradeoffs. Impacts that appear to be only modest or trivial could be important for species that are currently near a threshold or under a rebuilding plan. Because large-scale, long-term ecological experiments are impractical, if not impossible, scientists will continue to rely on simulation models to predict the impacts of complex ecosystem processes. While the field of ecosystem-based fisheries science has grown greatly over the last decade or so, agencies have had difficulties incorporating it into the management process. Ecosystem models such as this are intended to complement, not replace, existing tools used for assessment and management of fish stocks. The work presented here offers one example of how they can be useful and hopefully will lead to a more focused and coherent strategy for EBFM in the Gulf of Mexico.

Table 2-1. Biomass, catch, and mortality estimates for the present day, 2009 WFS Reef Fish Ecopath Model.

Group name	Biomass (mil lbs)	Total Fishery Removals (mil lbs)	Inst. Total Mortality (Z)	Inst. Fishing Mortality (F)
dolphins	6.37	0.00	0.16	0.00
seabirds	4.50	0.00	0.30	0.00
large coastal sharks	15.18	3.64	0.41	0.24
small coastal sharks	25.07	1.36	0.54	0.05
rays skates	89.57	2.97	0.85	0.03
billfish, tuna	12.13	1.29	0.68	0.11
oceanic sm. pelagics	149.91	0.92	1.36	0.01
Cobia	4.66	0.93	0.70	0.20
King Mackerel juv	0.72	0.03	2.00	0.04
King Mackerel adult	19.36	4.07	0.80	0.21
Sp. Mackerel juv	2.88	0.06	2.00	0.02
Sp. Mackerel adult	29.01	4.64	1.08	0.16
jacks/dolphin/tunnies	47.47	6.92	0.72	0.15
Red Snapper juv	0.51	0.02	1.50	0.03
Red Snapper adult	17.35	3.47	0.74	0.20
Vermilion Snapper	3.37	1.55	0.86	0.46
other snapper	72.45	21.96	0.63	0.30
Tilefish	2.08	0.35	0.50	0.17
YEG 0-1 yr	0.05	0.00	2.50	0.00
YEG 1-3 yr	0.73	0.00	0.80	0.00
YEG 3+ yr	8.88	0.62	0.40	0.07
other DWG	3.91	0.32	0.40	0.08
Gag 0-1 yr	0.38	0.02	1.48	0.04
Gag 1-3 yr	5.87	1.38	1.06	0.23
Gag 3+ yr	16.64	8.49	0.80	0.51
Red Grouper 0-1 yr	1.10	0.00	2.00	0.00
Red Grouper 1-3 yr	14.12	0.28	0.80	0.02
Red Grouper 3+ yr	72.00	11.45	0.40	0.16
Black Grouper 0-1 yr	0.08	0.00	2.00	0.02
Black Grouper 1-3 yr	1.10	0.09	0.80	0.08
Black Grouper 3+ yr	6.90	0.74	0.40	0.11
other SWG	7.50	0.37	0.40	0.05
Goliath Grouper	1.44	0.08	0.40	0.05
Triggerfish	2.07	0.87	0.89	0.42
Amberjacks	2.40	1.37	1.06	0.57
Black Seabass	4.49	0.34	1.12	0.08
reef carnivores	337.31	3.50	1.32	0.01

Table 2-1. Continued

Group name	Biomass (mil lbs)	Total Fishery Removals (mil lbs)	Inst. Total Mortality (Z)	Inst. Fishing Mortality (F)
reef omnivores	149.91	0.01	1.66	0.00
coastal piscivores	39.03	8.07	0.71	0.21
lg. coastal carniv.	159.66	11.24	0.92	0.07
sm. coastal carniv.	215.88	1.74	1.76	0.01
coastal omnivores	262.35	2.75	1.98	0.01
sardine/herring/scad	637.14	16.61	2.20	0.03
anchovies/silversides	222.06	0.04	2.67	0.00
mulletts	57.45	14.68	1.42	0.26
squid	120.08	0.04	2.67	0.00
shrimp	341.09	7.76	3.66	0.02
lobsters	13.12	1.49	0.90	0.11
large crabs	191.80	7.71	1.69	0.04
octopods	38.27	0.01	3.10	0.00
stomatopods	372.54	0.00	1.34	0.00
echino/lg. g-pods	7213.13	1.22	2.60	0.00
bivalves	18213.84	0.00	5.35	0.00
sessile epibenthos	8207.81	0.00	1.62	0.00
small infauna	7132.92	0.00	4.02	0.00
small mobile epifauna	4649.97	0.56	4.76	0.00
meiofauna	4872.22	0.00	6.20	0.00
small copepods	3110.72	0.00	10.60	0.00
mesozooplankton	2511.07	0.00	10.60	0.00
carnivzooplank	4047.69	0.00	8.70	0.00
ichthyoplankton	71.21	0.00	50.45	0.00
carnivjellyfish	82.83	0.54	20.08	0.01
microbes	22487.15	0.00	100.00	0.00
macroalgae	13511.03	0.00	4.00	0.00
microphytobenthos	11161.12	0.00	23.73	0.00
phytoplankton	9369.65	0.00	182.13	0.00
sea grasses	65819.89	0.00	9.00	0.00
water column detritus	46848.23	0.00	0.00	0.00
sediment detritus	146166.48	0.00	0.00	0.00
dead discards	0.37	0.00	0.00	0.00

Table 2-2. Biomass, catch, and mortality estimates for the historic, 1950s WFS Reef Fish Ecopath Model.

Group name	Biomass (mil lbs)	Total Fishery Removals (mil lbs)	Inst. Total Mortality (Z)	Inst. Fishing Mortality (F)
dolphins	6.37	0.00	0.16	0.00
seabirds	1.87	0.00	0.30	0.00
large coastal sharks	33.73	1.82	0.15	0.05
small coastal sharks	40.43	0.29	0.30	0.01
rays skates	89.57	2.97	0.50	0.03
billfish, tuna	17.41	0.64	0.34	0.04
oceanic sm. pelagics	149.91	0.92	1.74	0.01
Cobia	13.65	0.14	0.50	0.01
King Mackerel juv	0.17	0.00	2.00	0.00
King Mackerel adult	29.88	2.08	0.40	0.07
Sp. Mackerel juv	0.78	0.00	2.00	0.00
Sp. Mackerel adult	64.85	2.43	0.40	0.04
jacks/dolphin/tunnies	67.82	0.68	0.50	0.01
Red Snapper juv	0.08	0.00	2.00	0.01
Red Snapper adult	38.75	1.48	0.25	0.04
Vermilion Snapper	6.25	0.06	0.40	0.01
other snapper	90.55	0.91	0.60	0.01
Tilefish	2.81	0.01	0.30	0.01
YEG 0-1 yr	0.01	0.00	6.00	0.00
YEG 1-3 yr	0.06	0.00	0.80	0.00
YEG 3+ yr	26.91	0.26	0.08	0.01
other DWG	11.83	0.32	0.20	0.03
Gag 0-1 yr	0.15	0.00	3.00	0.01
Gag 1-3 yr	1.36	0.03	0.90	0.02
Gag 3+ yr	50.45	1.32	0.15	0.03
Red Grouper 0-1 yr	0.64	0.00	2.00	0.00
Red Grouper 1-3 yr	8.65	0.10	0.80	0.01
Red Grouper 3+ yr	121.99	12.34	0.25	0.10
Black Grouper 0-1 yr	0.12	0.00	2.00	0.00
Black Grouper 1-3 yr	1.75	0.00	0.80	0.00
Black Grouper 3+ yr	21.39	0.52	0.30	0.02
other SWG	19.56	0.37	0.25	0.02
Goliath Grouper	2.75	0.08	0.13	0.03
Triggerfish	6.68	0.01	0.50	0.00
Amberjacks	9.45	0.09	0.25	0.01
Black Seabass	10.91	0.17	1.00	0.02
reef omnivores	222.62	0.01	1.98	0.00

Table 2-2. Continued

Group name	Biomass (mil lbs)	Total Fishery Removals (mil lbs)	Inst. Total Mortality (Z)	Inst. Fishing Mortality (F)
coastal piscivores	77.96	8.07	0.71	0.10
lg. coastal carniv.	287.39	11.25	0.92	0.04
sm. coastal carniv.	287.84	1.75	1.76	0.01
coastal omnivores	349.80	2.75	1.98	0.01
sardine/herring/scad	637.14	12.71	2.31	0.02
anchovies/silversides	291.45	0.04	2.67	0.00
mulletts	69.97	29.65	1.20	0.42
squid	120.08	0.04	2.67	0.00
shrimp	255.98	7.77	3.66	0.03
lobsters	26.24	1.49	0.90	0.06
large crabs	333.56	7.70	1.69	0.02
octopods	47.93	0.01	3.10	0.00
stomatopods	372.54	0.00	1.50	0.00
echino/lg. g-pods	7213.13	1.22	2.60	0.00
bivalves	18213.84	0.00	5.35	0.00
sessile epibenthos	8207.81	0.00	1.62	0.00
small infauna	7132.92	0.00	4.02	0.00
small mobile epifauna	4649.97	0.56	4.76	0.00
meiofauna	4872.22	0.00	6.20	0.00
small copepods	3110.72	0.00	10.60	0.00
mesozooplankton	2511.07	0.00	10.60	0.00
carnivzooplank	4047.69	0.00	8.70	0.00
ichthyoplankton	71.21	0.00	50.45	0.00
carnivjellyfish	82.83	0.54	20.08	0.01
microbes	22487.15	0.00	100.00	0.00
macroalgae	13511.03	0.00	4.00	0.00
microphytobenthos	11161.12	0.00	23.73	0.00
phytoplankton	4920.94	0.00	182.13	0.00
sea grasses	65819.89	0.00	9.00	0.00
water column detritus	46848.23	0.00	0.00	0.00
sediment detritus	131175.04	0.00	0.00	0.00
dead discards	0.37	0.00	0.00	0.00

Table 2-3. Impact of current management options, possible primary productivity changes, and sensitivity scenarios on select groups. Baseline Ecopath and status quo biomass estimated expressed in million pounds while other scenarios expressed as percent change from status quo. Values in parenthesis represent the 95% confidence intervals estimated from 100 Monte-Carlo simulations.

Group	Ecopath Base (mil lbs)	Status Quo (mil lbs)	Gag Rebuild (%)	Longline Effort (%)	Baitfish F = 0.4 (%)	Low PP (%)	High PP (%)
gag	16.65	8.32	240 (230,246)	5 (3,6)	-2 (-3,0)	-1 (-4,-1)	5 (3,6)
red grouper	72.03	92.81	-1 (-2,0)	20 (20,23)	-3 (-3,-1)	-6 (-7,-5)	6 (5,7)
black grouper	6.90	8.05	-5 (-5,-3)	1 (0,3)	0 (-1,2)	-1 (-4,-1)	2 (0,2)
yellowedge grouper	8.89	10.14	-1 (-1,1)	65 (62,65)	-1 (-2,0)	-2 (-4,-2)	2 (1,2)
goliath grouper	1.44	2.01	0 (-2,13)	-2 (-6,12)	2 (5,22)	7 (-2,15)	13 (4,22)
other SWG	7.50	9.86	-7 (-11,-2)	6 (1,13)	-9 (-16,-6)	-2 (-8,2)	4 (3,14)
other DWG	3.91	4.45	-2 (-5,12)	22 (28,49)	7 (-4,12)	-7 (-15,0)	14 (5,23)
red snapper	17.36	22.00	0 (-2,7)	-4 (-3,7)	-7 (-11,-3)	-19 (-22,-15)	19 (16,27)
vermillion snapper	3.37	3.61	-6 (-9,-2)	-7 (-10,-3)	-12 (-15,-9)	-16 (-24,-18)	17 (12,21)
tilefish	2.08	1.97	4 (-3,10)	74 (69,89)	1 (-1,13)	0 (-6,6)	7 (0,14)
greater amberjack	2.40	2.59	-3 (-5,-1)	-2 (-4,0)	-5 (-7,-2)	-7 (-9,-5)	8 (4,9)
triggerfish	2.07	2.35	2 (-1,4)	1 (-1,3)	14 (13,18)	-5 (-7,-2)	7 (8,13)
black sea bass	4.50	4.78	-20 (-22,-18)	-13 (-18,-13)	-1 (-4,2)	0 (-3,3)	4 (4,11)
king mackerel	19.37	23.73	-6 (-8,-4)	-3 (-5,-1)	-10 (-12,-8)	-17 (-20,-17)	16 (11,15)
Spanish mackerel	29.02	34.17	-2 (-4,0)	-1 (-3,2)	-4 (-6,-2)	-12 (-15,-11)	17 (14,20)
cobia	4.66	5.26	0 (-2,4)	-2 (-3,3)	4 (1,7)	-1 (-3,2)	2 (2,9)
sardine-herring-scad	637.42	579.11	-2 (-11,-2)	-6 (-11,-2)	-23 (-32,-23)	-18 (-25,-18)	16 (14,25)
dolphins	6.37	6.25	-2 (1,13)	-2 (7,19)	-6 (1,12)	-6 (-6,6)	7 (10,23)
seabirds	4.50	4.61	-2 (16,26)	-1 (14,23)	-11 (3,11)	-7 (6,14)	7 (26,37)

Table 2-3. Continued

Group	F = 0 (%)	F x 0.5 (%)	F x 1.5 (%)	F x 2 (%)	F x 5 (%)
gag	868 (869,959)	192 (182,190)	-77 (-78,-75)	-98 (-98,-98)	-100 (-100,-100)
red grouper	119 (114,124)	44 (42,46)	-30 (-30,-28)	-51 (-52,-50)	-96 (-96,-95)
black grouper	24 (23,28)	18 (17,20)	-22 (-24,-21)	-45 (-45,-43)	-99 (-99,-99)
yellowedge grouper	113 (112,119)	42 (41,44)	-30 (-32,-30)	-51 (-51,-49)	-91 (-92,-91)
goliath grouper	21 (20,45)	14 (12,33)	-16 (-14,0)	-30 (-35,-21)	-84 (-83,-76)
other SWG	-2 (-4,8)	1 (-4,6)	-5 (-10,-1)	-12 (-14,-4)	-53 (-56,-49)
other DWG	36 (34,56)	18 (10,30)	-16 (-18,-4)	-32 (-30,-16)	-91 (-91,-89)
red snapper	20 (15,24)	16 (10,21)	-15 (-18,-12)	-37 (-39,-34)	-100 (-100,-100)
vermilion snapper	94 (102,118)	65 (58,71)	-66 (-71,-67)	-97 (-97,-96)	-100 (-100,-100)
tilefish	125 (118,142)	57 (52,70)	-45 (-49,-41)	-75 (-75,-69)	-100 (-100,-100)
greater amberjack	82 (78,89)	33 (31,35)	-45 (-47,-43)	-76 (-77,-74)	-100 (-100,-100)
triggerfish	121 (115,124)	50 (48,54)	-36 (-36,-32)	-62 (-62,-59)	-100 (-100,-100)
black sea bass	-80 (-81,-79)	-44 (-47,-43)	58 (55,64)	115 (108,120)	158 (147,163)
king mackerel	65 (58,64)	29 (23,27)	-25 (-29,-26)	-46 (-50,-47)	-100 (-100,-99)
Spanish mackerel	38 (30,37)	18 (13,18)	-15 (-18,-14)	-28 (-31,-28)	-87 (-87,-85)
cobia	89 (90,107)	33 (33,44)	-21 (-23,-18)	-34 (-34,-30)	-71 (-72,-70)
sardine-herring-scad	-19 (-26,-18)	-8 (-17,-8)	8 (-4,5)	15 (4,14)	47 (39,47)
dolphins	-11 (-8,4)	-5 (-5,8)	5 (10,23)	10 (19,32)	51 (50,71)
seabirds	-13 (-2,6)	-5 (11,20)	4 (18,29)	8 (23,33)	37 (55,68)

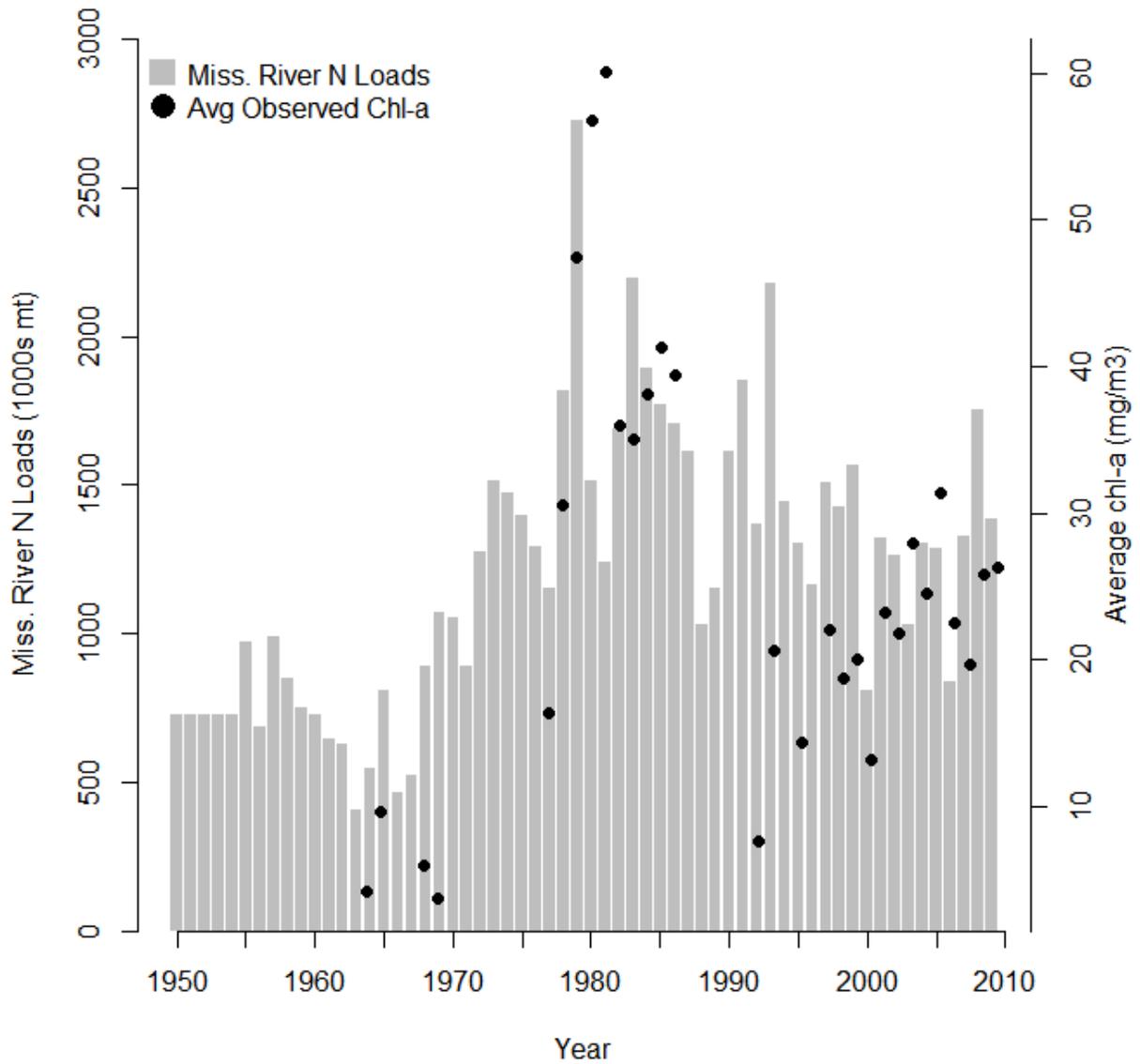


Figure 2-1. Nitrogen loads from the Mississippi River obtained from Goolsby and Battaglin (2000) and Aulenbach et al. (2007) along with average observed chlorophyll-a from both satellite and in-situ measurements of the WFS.

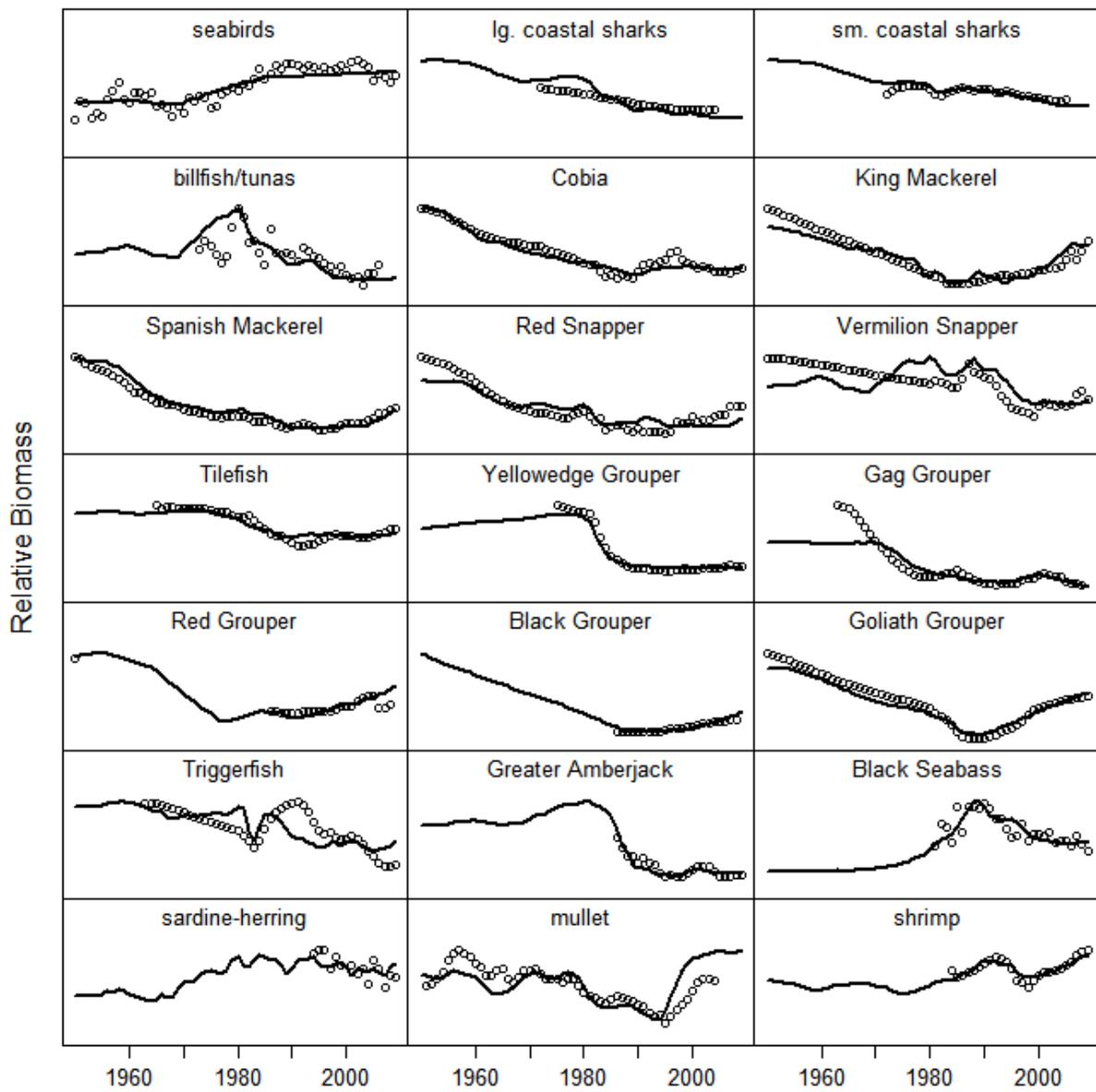


Figure 2-2. Predicted (solid line) biomass values from the Historic model compared to observed trends in biomass (circles) for select species.

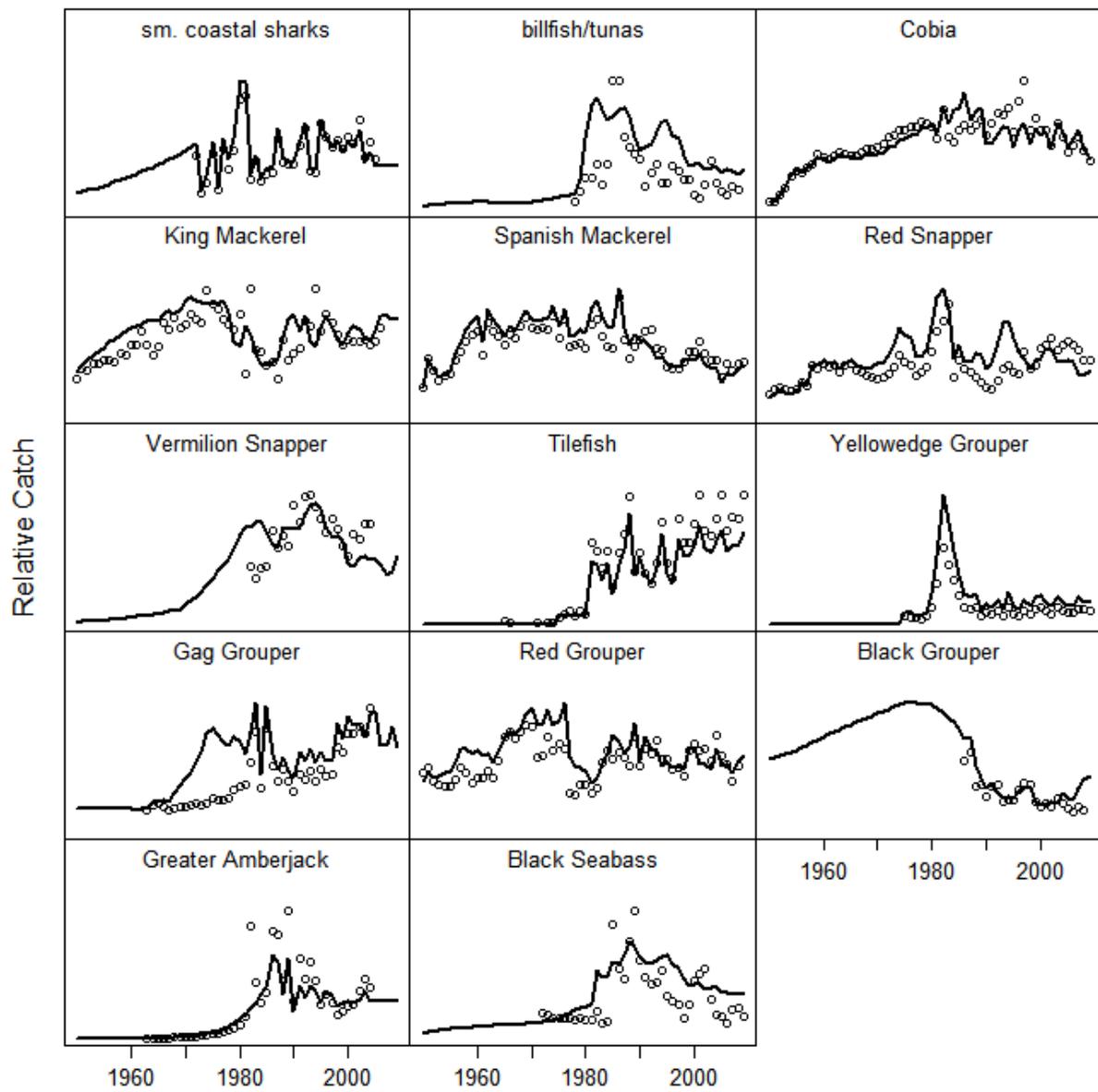


Figure 2-3. Predicted (solid line) catch from the Historic model compared to observed trends in catch (circles) for select species.

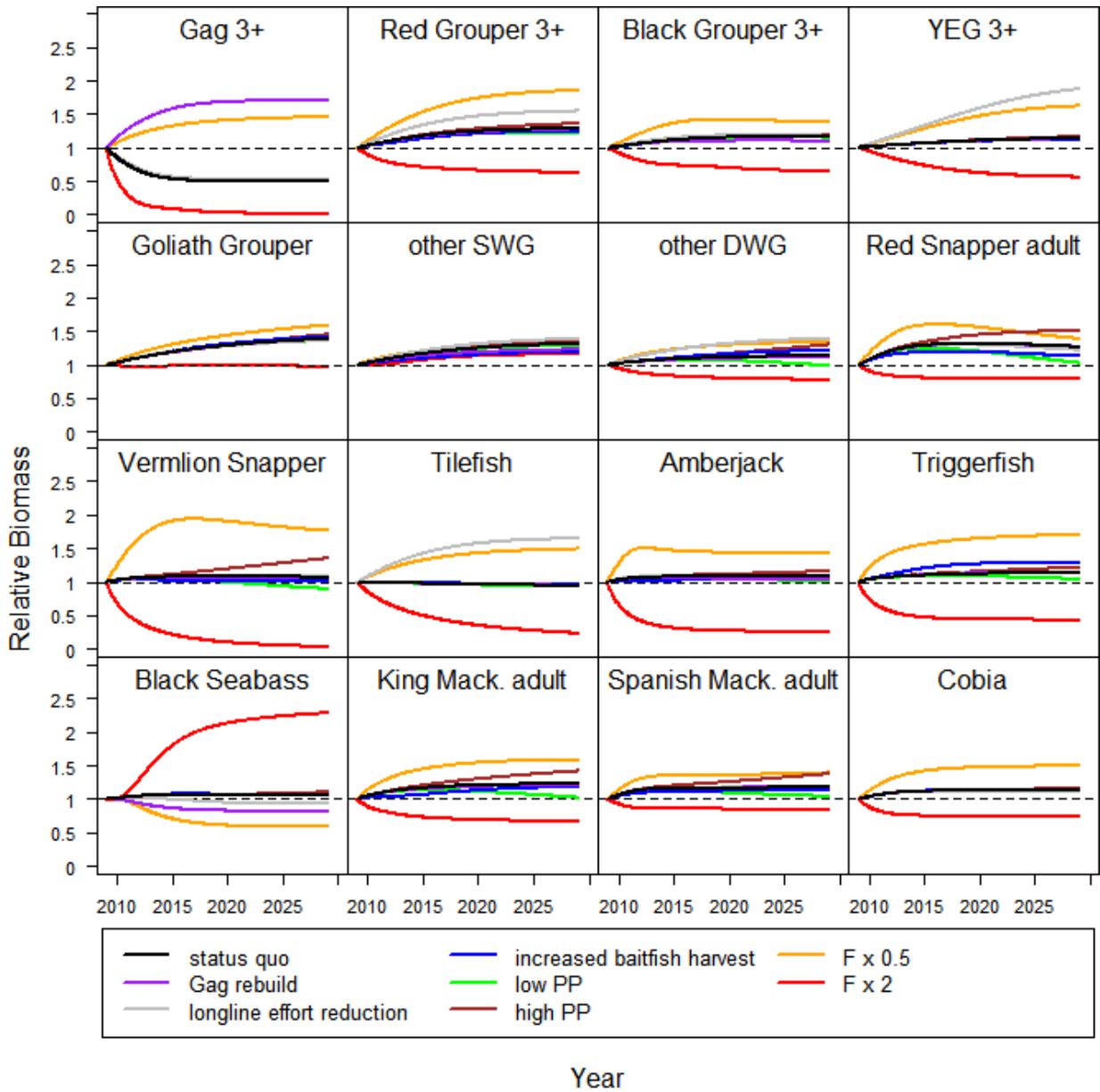


Figure 2-4. Future biomass trajectories simulated by the Ecosim model. Scenarios leading to a reduction in biomass compared to the status quo are those that fall below the solid black line. The dotted line represents the Ecopath base 2009 biomass level. YEG = Yellowedge Grouper, juv. = juvenile, Mack. = Mackerel.

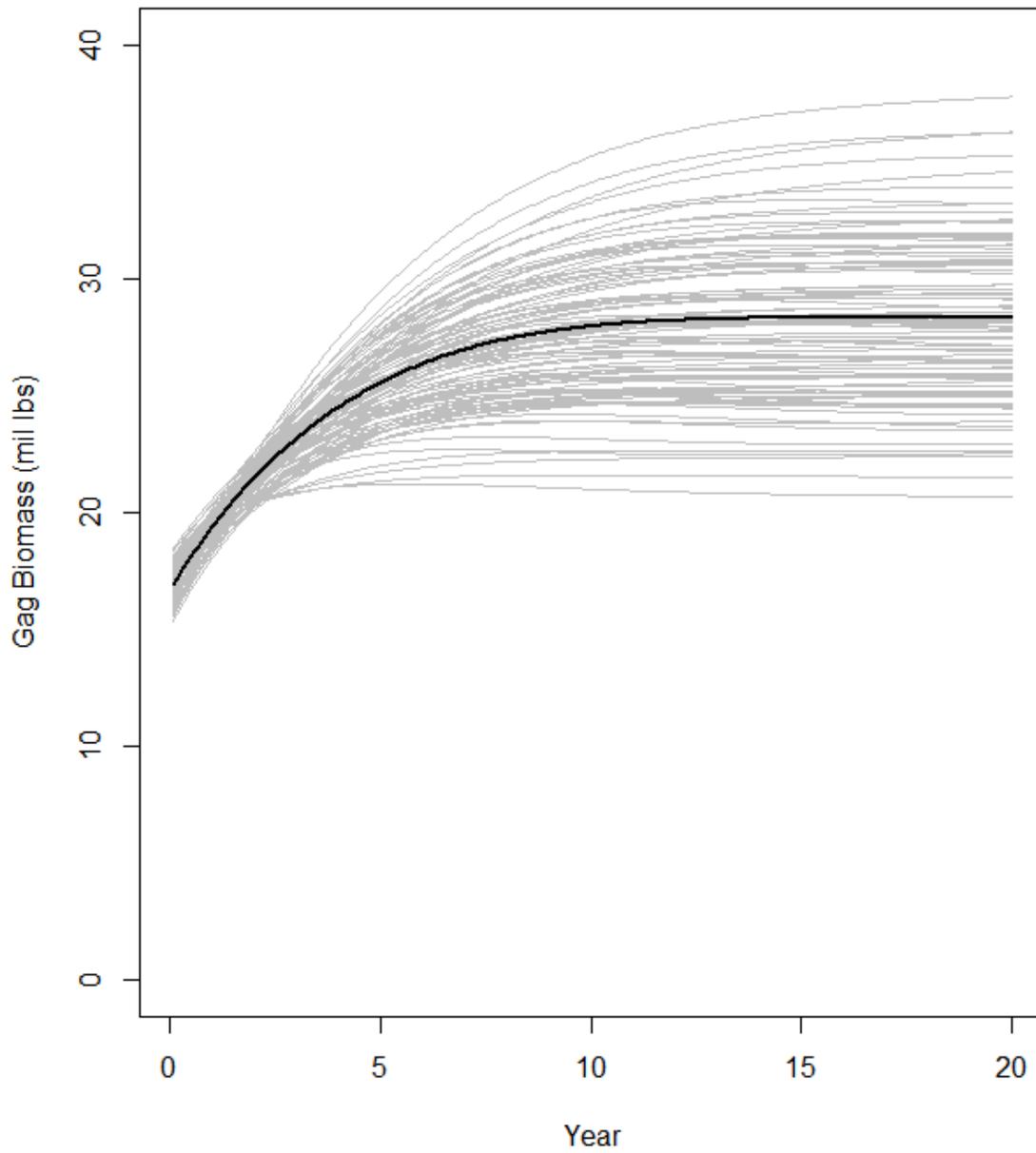


Figure 2-5. Gag grouper projections under $F_{rebuild}$ of 0.16 where the solid black line is the base run and the gray lines are Monte Carlo simulation trials with randomly chosen Ecopath biomass estimates.

CHAPTER 3 TRADEOFFS IN OPTIMAL HARVEST POLICIES FOR WEST FLORIDA SHELF FISHERIES

Introduction

Management of living marine resources involves contending with a set of policy goals that are often in conflict, such as ensuring viable fisheries and maintaining healthy marine food webs. In such cases, a tradeoff must be made that involves giving up some of one thing in exchange for more of something else. Common tradeoffs in fisheries management include current versus future harvests, harvest of prey versus abundance of predators, and fishery production versus biodiversity (Walters and Martell 2004). The tradeoff relationships may be convex shaped in which one objective can be achieved without negatively impacting the ability to achieve the other, such that a balanced policy exists in which both values are high. Alternatively, the relationship can be concave or linear in which case increases in the performance of one objective reduces the ability to achieve high values of another. As the general public becomes more involved in fisheries conservation and management issues, acknowledging the tradeoffs between conflicting management objectives is increasingly important to help avoid policies that represent the interests of one stakeholder at the expense of others.

In the Gulf of Mexico, fisheries managers are confronted by numerous tradeoffs, especially as they relate to measures that aim to end overfishing. For example, rebuilding plans must balance short term losses with timeframes for stock recovery (GMFMC 2004a; 2004b), and the social and economic impacts of seasonal and spatial closures on fishing communities must be balanced with biological benefits for the species (GMFMC 2008; GMFMC 2011). For example, the costs of bycatch reduction in the shrimp fishery must be balanced with the need to protect other species that support

less valuable fisheries (GMFMC 2004a). These conflicts have sparked intense debates about regulations and allocation rights that often put fishers at odds with each other, conservationists, and management agencies. In many cases all parties involved are ill informed about the tradeoffs they face because stakeholder values often span multiple scientific disciplines, from biology and ecology to economics and social sciences, and are therefore rarely evaluated under a common framework. A more holistic approach to evaluating tradeoffs in policy options is needed to better understand their costs and benefits at the ecosystem level and to identify optimal harvest policies where the tradeoffs are balanced among stakeholders.

Evaluating tradeoffs in fisheries management and optimizing across multiple performance measures has been conducted in a variety of ways. Multi-objective decision analysis revealed a convex set of tradeoffs between yield of Yellowfin Tuna, *Thunnus albacares*, bycatch, and dolphin mortality in the Eastern Pacific Ocean (Enriquez-Andrade and Vaca-Rodriguez 2004). Multi-species bioeconomic models have been used to determine optimal harvest rates in predator-prey systems (Kellner *et al.* 2010) and mixed fisheries (Da Rocha *et al.* 2012; Hilborn *et al.* 2012). Dynamic ecosystem models that include ecological interactions and socio-economic values are capable of optimizing over a broader set of management goals. An Atlantis trophic-dynamic ecosystem model was used to evaluate the tradeoffs associated with protecting the Vaquita Porpoise, *Phocoena sinus*, in the Northern Gulf of California (Morzaria-Luna *et al.* 2012). On the US West Coast, an Atlantis model was linked with an economic input-output model to evaluate broader economic and ecological tradeoffs of fisheries management options (Kaplan and Leonard 2012).

Ecopath with Ecosim (EwE) includes an optimization procedure that searches for the combination of fishing efforts that maximizes a multi-criteria objective function, and is useful for evaluating the tradeoffs among socio-economic and ecosystem policy goals (Christensen and Walters 2004). The approach has been applied in the southwestern Gulf of Mexico (Arreguin-Sanchez *et al.* 2004b), Gulf of California (Arreguin-Sanchez *et al.* 2004a), Prince William Sound, Alaska (Okey and Wright 2004), the northern Benguela ecosystem of western Africa, and the South China Sea (Cheung and Sumaila 2008; Wang *et al.* 2012). In these applications, optimizing for socio-economic criteria resulted in severe depletion of some stocks and ‘cultivation’ of the system towards highly productive low trophic level species. Further, fishing effort was drastically reduced across all fleets in order to improve ecosystem structure.

Here, I have attempted to quantify the tradeoffs between reef fish conservation and profits taken by the combined fishery on the West Florida Shelf (WFS) while accounting for trophic dynamics. My intention is not to make specific policy recommendations but to provide a range of practical and obtainable strategic options for both conservation and economic priorities in the eastern Gulf of Mexico. I focus on the shape of the tradeoff frontiers and when possible identify regions along the frontier where values of both conservation and economic gain are high and describe the policy choices that fall within those regions. This model serves to set limits on what is achievable in terms conservation and economic goals and provides a framework for ranking policy options and stakeholder involvement.

Methods

The policy optimization search was conducted using the forward projecting Ecosim model described in Chapter 2. The model consisted of 70 biomass pools and

emphasized the valuable reef fish fisheries of the eastern Gulf of Mexico. Gag Grouper, Red Grouper (*Epinephelus morio*), Black Grouper (*Mycteroperca bonaci*), and Yellowedge Grouper (*Epinephelus flavolimbatus*) were each divided into 3 age stanzas and Spanish Mackerel (*Scomberomorus maculatus*), King Mackerel (*Scomberomorus cavalla*), and Red Snapper (*Lutjanus campechanus*) were all divided into juvenile and adult age stanzas. Forage groups such as shrimp and baitfish (sardines, herrings, scads) were included along with the fisheries that harvest them and invertebrates and primary producers where necessary for a complete food web. There were 14 fishing “fleets” including the bottom longline and vertical line ‘bandit’ commercial fisheries that target groupers and snappers as well recreational private boat, charter, and headboat fisheries. This model inherits vulnerability exchange rates from a historical Ecosim simulation (1950-2009) that included a forcing function to drive phytoplankton dynamics and was calibrated to time series of biomass and catch.

The fishery policy optimization module in Ecosim attempts to identify an optimum mix of fishing efforts for multiple fleets that catch multiple species (Christensen and Walters 2004). In this approach, fishing efforts are adjusted to maximize a weighted multi-criteria objective function that includes a component for profitability, ecosystem structure, employment, and mandated stock sizes (i.e. stock rebuilding plans). By modifying the weighting factors on the objective function components, more or less emphasis can be put on each policy goal. The search procedure uses a nonlinear optimization known as the Davidon-Fletcher-Powell (DFP) method to iteratively improve an objective function following a conjugate-gradient parameter estimation scheme (Davidon 1991). At each step, the search routine locates a local maximum for the

objective function assuming a quadratic relationship with parameter values, and then makes parameter update steps from the local approximations. This is one of the more efficient algorithms for such optimization problems in complex, nonlinear systems.

The model allows each fleet to be treated as a separate economic industry ('multiple fishing rights' approach) or incomes and costs can be pooled and profits shared among all fleets ('sole owner' approach). The assumption of value and cost pooling of the 'sole owner' approach can lead to optimum solutions where one or more fleets are operated at a loss in order to "cultivate" the system for fleets that target species with higher economic potential. Because this is unrealistic, I conducted optimization using the 'multiple rights' approach that assumes each fleet has an entitlement to harvest and that they must maintain reasonable levels of profitability by ensuring that costs remain less than profits. The productivity of fleets are linked through trophic interactions such that growth of some fleets could either enhance or diminish the productivity of others. In order to account for this in the multiple rights optimization, a cross-impact matrix of costs to every fleet caused by every other fleet is used to adjust profitability prior to each iteration. This is essentially a constrained economic solution that maintains all fleets at viable levels (Christensen *et al.* 2005).

Components of the objective function include economic, social, ecological, and mandated biomass criteria. Net economic value is taken as the sum over fleets of total landed value minus costs and discounted over time at a rate of 0.04. Market values for each species were the statewide average ex-vessel price per pound in 2009 obtained from the Florida FWC commercial landings summaries. I assumed the market prices from commercial landings to be equal across all fleets, including sport fisheries, and

address the impact of alternative assumptions about the relative value of recreationally versus commercially caught fish through sensitivity runs. For each fleet, costs are divided into fixed costs and effort related costs where fixed costs include regular maintenance, major repairs, depreciation and capital equipment, overhead, licensing, and insurance and effort related costs include expenditures such as bait, tackle, ice, wages, oil, and fuel. Revenue shares for the commercial sector (Table 3-1) were based on the NMFS revenues and expenditures surveys as reported in Liese *et al.* (2009) for the trawl fishery and the Southeast Fishery Science Center (SEFSC 2011, personal communications) for other fleets. The two for-hire recreational fleets (charter boat and headboat) were treated as commercial ventures and the cost structure was computed using the same allocation of expenditures used for commercial fishing vessels and based on a cost and earnings survey of charter boat operations in the Southeast U.S. (Liese and Carter 2011). The cost structures of the private recreational fleets were based on expenditure and willingness to pay surveys of South Florida anglers (Johns *et al.* 2001) where fixed costs included boat maintenance, ramp and marina fees, and lodging/camping, and effort related costs included bait, tackle, ice, food and beverage, restaurant and bar expenses, equipment rental, and fishing related shopping. Profitability was then based on consumer surplus which is, in theory, equivalent to the producer surplus or profits enjoyed by commercial fishers. In all cases, the market price, cost structure, and jobs were held constant for the duration of each 40 year simulation.

To represent the societal benefit of the fishery, some measure of employment (number of jobs, total wages per unit catch) must be supplied (Table 3-1). The number

of jobs supported by for-hire, private boat, and shore-based recreational fleets, and the combined commercial fishery were available from a NMFS socio-economic trends report and estimated with an input-output model (NMFS 2010). In order to focus on the employment directly generated by fishing, the retail and distribution employment sectors provided in the report were removed from the jobs calculation. For the commercial and for-hire sectors, whose figures were reported at coarser scales than the fleet composition in the model, the number of jobs was allocated based on total revenues generated by each fleet in the model.

The ecological component of the objective function is usually represented by the sum product of species biomass times 'longevity' (inverse of P/B) and is intended to provide a metric of ecosystem 'stability'. However, users are free to assign any combination of weights to best represent a desired ecosystem structure or policy goal. In this analysis, I assigned all reef fish species a weight of 1 and all other species 0 to emphasize the tradeoffs associated with reef fish conservation and fishery economics. Lastly, the mandated biomass level for Gag was set to 2.0 to represent the current rebuilding plan objective to double stock size from 2009 levels (GMFMC 2008b).

The optimization was conducted over a 40 year simulation of the Ecosim model starting with 2009 Ecopath base F's. To emphasize only the fisheries operating on the shelf I excluded recreational shore-based, cast nets, gill nets, and crab trap fisheries from the analysis. All other fleets were set to optimize for a single effort value over the entire 40 year simulation period. First, I optimized for economic value and reef fish biomass separately by setting the weight in the objective function to 1 for the target objective and others to zero. To generate the tradeoff frontiers, I processed the

optimizations in batch mode using EwE v5.1 and incremented the weights on one component of the objective function from 0-1 in steps of 0.1 while keeping the other components fixed at values of 0.05, .25, .45, .65, .75, and .95 for a total of 264 optimal solutions. The batch process was repeated five times to evaluate the impact of different assumptions about the relative values of the fleets. Because of substantial uncertainty in the value of the recreational fleet, I varied the prices for this fleet. In one case, recreational prices were equal to the average amount of money spent on a fishing trip (i.e. mean access costs) for trips targeting those species, which could be as much as ten times higher than commercial dockside value. In other cases I set recreationally caught fish to ½, 2 and 5 times the ex vessel market price or allowed the commercial operations to have higher profit margins.

Results

Maximizing total profits required at least some buildup of reef fish biomass. In this scenario, total profits increased by 30% to \$42.8 million dollars, reef fish biomass increased by 23% to 288 million pounds, and the mandated rebuilding of Gag was achieved (Figure 3-1). Pelagic and bottom longline, vertical line, and private boat fleets were all reduced to effort levels about half the Ecopath 2009 base level. The most notable increases in effort occurred for the commercial purse seine and shrimp trawl fisheries. Maximizing for reef fish biomass resulted in a \$20 million loss in profits (-60%), a 45% loss of jobs while more than doubling the biomass of reef fish (Figure 3-2). In order to meet this conservative management goal the two commercial fisheries for reef fish (bottom longline and vertical line fleets) as well as the entire recreational sector were effectively shut down, effort in the pelagic troll and shrimp trawl fisheries were reduced by almost half, and effort in the pelagic longline fishery was increased by over

50%. The increase in the longline fleet essentially culled reef fish competitors and thus increased reef fish biomass.

The Pareto-frontiers between reef fish biomass and profits were convex shaped and approximated using a quadratic function (Figure 3-3). Profits were maximized at higher levels of reef fish biomass (red boxes) up until a 60% increase (<400 million lbs) after which profits dropped off severely. While the shapes were similar across all scenarios, the optimum effort configuration differed depending on how the fleets were valued. In the base scenario (equal market prices) the runs that fell within the optimal zone identified in Figure 3-3 increased effort in the purse seine and shrimp trawl fisheries on average by 30 and 8 fold over base levels respectively (Figure 3-4). Pelagic troll fisheries increased by nearly 5 fold, and the for-hire recreational fleets remained mostly unchanged. Effort in the bottom longline and vertical line fleets was reduced by about 80%, and the private recreational boats were effectively shut down. When recreationally caught fish were assumed to be more valuable, from scenarios B to C and E, optimal efforts for the trawl and purse seine fisheries increased to a much lesser extent (Figure 3-4). When a more commercial friendly configuration was optimized (scenarios D and F), efforts for purse seines and shrimp trawls were even higher, bottom longlines continued to be reduced, and the vertical line fishery was affected more by assumptions about market prices than profitability. In all cases the private recreational fleet was shut down, the haul seine and shrimp trawl fisheries were never reduced by more than 25%, and in no cases were the for-hire sectors increased or decreased drastically.

Lastly, when the 2009 base biomass and profits are plotted with the tradeoff frontier it is clear that the system is currently sub-optimal in terms of both reef fish biomass and total profits (Figure 3-5). Adding the Ecosim simulations described in Chapter 2 shows that maintaining status quo, in which several species are currently increasing in biomass, will move the system closer to an optimum state. Rebuilding gag stocks adds even more biomass and profit, whereas reductions in longline fishing effort will increase reef fish biomass without much economic benefit to the combined fishery (Figure 3-5).

Discussion

This analysis revealed the tradeoffs between reef fish conservation and fishery profits, two conflicting fisheries management objectives on the West Florida Shelf. Developing the tradeoff frontiers did not identify a single “correct” solution but instead provided a range of options, all of which are optimal for a stated set of policy goals. In all cases the Pareto-frontier was convex shaped meaning that a “balanced” policy exists in which both values are high. Alternative assumptions about the value of recreationally caught fish resulted in tradeoff frontiers of similar shapes and magnitudes but with different optimal fleet configurations.

The region of balanced policies along the frontier represents a set of policy options for the West Florida Shelf that could achieve reef fish biomass increases of about 50% without severe economic impacts on the combined fishery. The optimal range tended to be somewhere just to the right of the curve’s peak and left of the rapidly descending right arm, with biomasses between 300-375 million pounds and profits between \$40-45 million. Thus, further increases in reef fish biomass were expected to improve both fisheries and ecological objectives.

Single policy optimization runs that maximized entirely for either profits or biomass fell along the frontier but in sub-optimal regions of the curve (to the left of or below and to the right of the optimal zone). Points that fell interior of the frontier, where less of both values is achieved than what is predicted by the relationship, could represent cases where policy goals are oriented towards 'hidden' objectives other than profits or reef fish biomass (e.g. jobs or mandated biomass). This illustrates the ways that decision making (via weighting policy objectives) can fail to achieve larger management goals (Halpern *et al.* 2013; Walters and Martell 2004).

Another important finding from this analysis is that the purse seine and shrimp trawl fisheries on the West Florida Shelf are potentially underfishing their resources. In all scenarios except those with high recreational market prices, the optimum solutions involved considerably increasing effort in these two fisheries. Compared to large industrial reduction fisheries, the purse seine fishery in Florida is a relatively small scale operation that supplies baitfish to domestic and international markets. At its peak in the 1980s, effort in the purse seine fishery was about 20 times higher than it is currently (Mahmoudi *et al.* 2002) and Ecopath base 2009 fishing mortalities on the baitfish group are assumed to be quite low ($F = 0.02$). The commercial purse seine fishery was drastically restricted in 1995 by Florida's constitutional amendment prohibiting the use of purse seines in state waters. The regulations were largely intended to protect other species and not because baitfish on the WFS were overfished in the traditional sense. With effort currently very low and biomass assumed to be quite large, the model almost always prescribed an increase in the purse seine fishery, sometimes to near the historically high levels of the 1980s.

The shrimp trawl fishery on the West Florida Shelf targets mostly Pink Shrimp (*Farfantepenaeus duorarum*) and has undergone a nearly 90% decline in fishing effort since the late 1990s (Hart *et al.* 2012). The decline in effort has been attributed largely to adverse economic conditions driven by low-cost shrimp imports into the American market and an increase in fuel prices. The optimal efforts for the trawl fleet predicted by the model are consistent with historical fishing efforts and suggest a potential for much higher yields than are currently being taken by this fishery. However there is no consideration for the impact of low priced imports on market demand for domestic shrimp and levels of effort predicted by the model are not likely to be attained unless prices become more equitable. In an effort to increase market demand the Gulf of Mexico shrimp industry has been pursuing tariffs on shrimp imports from countries with high government subsidies, which are problematic for the low profit margin Gulf of Mexico shrimp fisheries (Mauldin 2013).

The effort configurations for runs that fell within the optimal zone almost always involved severe effort reductions in the bottom longline and vertical line fisheries while mostly maintaining the for-hire recreational sector at current levels. The GMFMC also recognized the need to reduce effort in the Gulf of Mexico commercial grouper fishery and installed IFQ programs in 2010 with the goal to reduce overcapacity and increase efficiency (NMFS 2011). These programs were predicted to downsize the combined vertical and longline fleets considerably while increasing efficiency and profits (Weninger 2008). Ecosim does not model fisheries at the vessel level and therefore is not capable of rationalizing effort to improve fleet efficiency. However, if the commercial longline and vertical line fleets in Ecopath were more efficient and selective (i.e. higher

profitability and fewer discards), as would be expected from an IFQ program, then optimal efforts would probably be higher than what was predicted here.

The impact that forage fisheries have on predator populations and how this should impact their management has been debated for some time (Knapp 1950; Pikitch *et al.* 2012). The most contentious aspects of these debates is usually not about the ecological role that forage fish play, but the economic impact of the forage versus non-forage fisheries. On the WFS, the extent to which forage fisheries should exist is dependent upon assumptions about the worth of a recreationally caught fish and/or the profitability of the fleets. Willingness to pay (WTP) studies have attempted to measure the value of recreationally caught fish in the Gulf of Mexico (Johnston *et al.* 2006). Estimates of the recreational value (\$/fish) of groupers and snappers range from \$5.41-\$11.47 (EPA 2004), to \$20.09-\$80.40 (Carter and Liese 2012), and as high as \$95 for groupers (\$14/lb) (Gentner 2009). Fish that are caught and released also have value and for some species (e.g. King Mackerel (*Scomberomorus cavalla*)) can be as high as that of a landed fish (Carter and Liese 2012). In EwE prices are per unit weight and no value is given to fish caught and released for sport nor is it possible to account for the values (social, cultural, aesthetic) enjoyed during a recreational fishing trip. Thus the 'profits' obtained by the recreational fishery could be underestimated in this analysis. Furthermore, accounting for the full value chain from producers to processors, distributors, and sellers would add considerable value to the commercial sector (Christensen *et al.* 2010). The policy optimization analyses demonstrates the tradeoffs that exists under different assumptions about fishery valuations and highlights the need

for economic impact studies of the recreational fisheries in the Gulf of Mexico as well as a more appropriate modeling framework for the recreational sector in Ecosim models.

The 2009 base case was suboptimal in both profits and biomass, and the positioning with the other Ecosim scenarios suggests the presence of an alternative tradeoff frontier below the one generated by the optimization analysis. That is, assuming a higher discount rate during optimization would reduce net present value and the frontier would be shifted downward on the y-axis (Cheung and Sumaila 2008; Larkin *et al.* 2006), closer to the base scenarios. Higher discount rates imply that the fisheries favored short-term economic benefits over long term ecological and economic gain, and these results suggest that the fisheries on the West Florida Shelf have been managed as such. I assumed a discount rate of 4%, but The U.S. Government's Office of Management and Budget (OMB) recommends a real discount rate of 7% that reflects the marginal pretax rate of return on an average investment in 2013 (OMB 2013). Discount rates may be higher than 20% in some fisheries (Larkin *et al.* 2006) with higher uncertainty, willingness to take risks, poverty, and indebtedness tending to put less value on future generations (Sumaila 2005).

The purpose of this study was to reveal the tradeoffs between conservation and economic management objectives in West Florida Shelf fisheries and provide advice about how to achieve an optimal state. From this analysis we conclude that 1) the system is suboptimal in both regards; 2) there is potential value to be gained in the purse seine and shrimp trawl fisheries; and 3) effort reduction is needed in the commercial reef fish fishery to reach the optimum balance of ecological and economic objectives. The optimal zone is where biomasses were 50% larger and total profits 30%

higher than they are currently. I demonstrated the utility of the tradeoff frontiers as a scorecard for ranking management options, which can be used to provide strategic advice in complex fisheries ecosystems like the Gulf of Mexico. Models such as this which take into consideration multiple objectives allow stakeholders to engage in better informed discussions about the tradeoffs they face and should be incorporated into the management process.

Table 3-1. Fleet Revenue Shares and the number of jobs per unit catch for all fleets in the Ecosim model.

Fleet	Fixed cost (%)	Effort related cost (%)	Sailing related cost (%)	Profit (%)	Jobs per unit catch
Rec-shore	44.8	45.3	0.0	9.9	79.0
Rec-private boat	29.0	45.3	15.8	9.9	15.0
Rec-charter boat	43.0	41.6	12.3	3.1	16.0
Rec-headboat	43.0	41.6	12.3	3.1	14.0
Com-cast nets	37.9	43.8	13.3	5.0	13.0
Com-offshore gill/trammel	37.9	43.8	13.3	5.0	11.0
Com-trawl	18.2	33.5	43.3	5.0	4.0
Com-purse/haul seines	37.9	43.8	13.3	5.0	20.0
Com-vertical lines	42.6	27.1	16.0	14.3	4.0
Com-long line	25.7	41.3	17.6	15.4	4.0
Com-fish traps	37.9	43.8	13.3	5.0	3.0
Com-crab traps	37.9	43.8	13.3	5.0	4.0
Com-troll	42.6	27.1	16.0	14.3	4.0
Com-pelagic long line	25.7	41.3	17.6	15.4	4.0

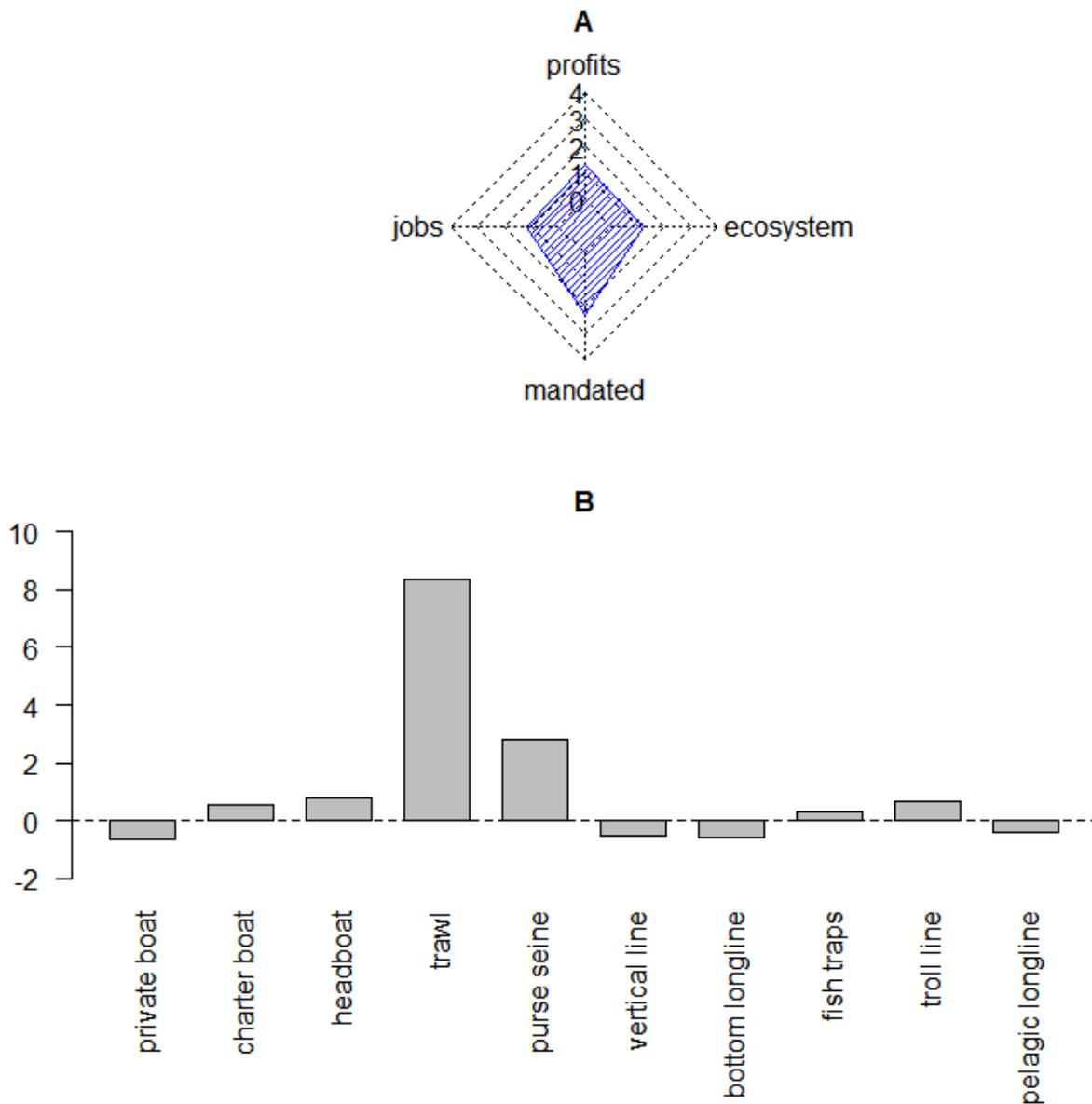


Figure 3-1. Objective performance and optimal fleet efforts from an optimization for fishery profits. Fleet codes are as follows: C indicates commercial and R indicates recreational, PLL = pelagic longline, TL = pelagic troll line, FT = fish traps, BLL = bottom longline, VL = vertical line, PS = purse seine, T = trawl, HB = headboat, CB = charter boat, PB = private boat.

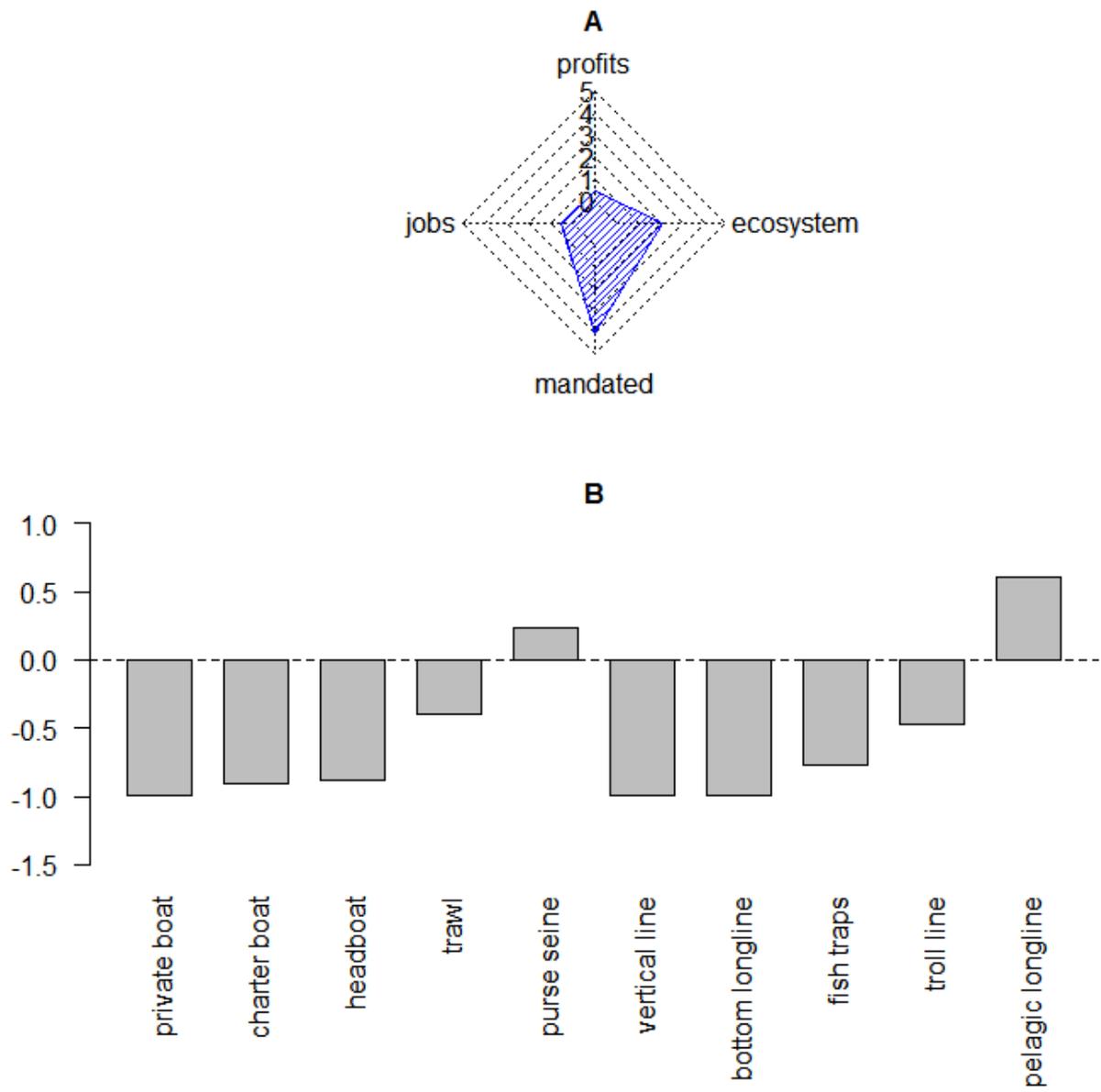


Figure 3-2. Objective performance and optimal fleet efforts from an optimization for reef fish biomass profits. Fleet codes are as follows: C indicates commercial and R indicates recreational, PLL = pelagic longline, TL = pelagic troll line, FT = fish traps, BLL = bottom longline, VL = vertical line, PS = purse seine, T = trawl, HB = headboat, CB = charter boat, PB = private boat.

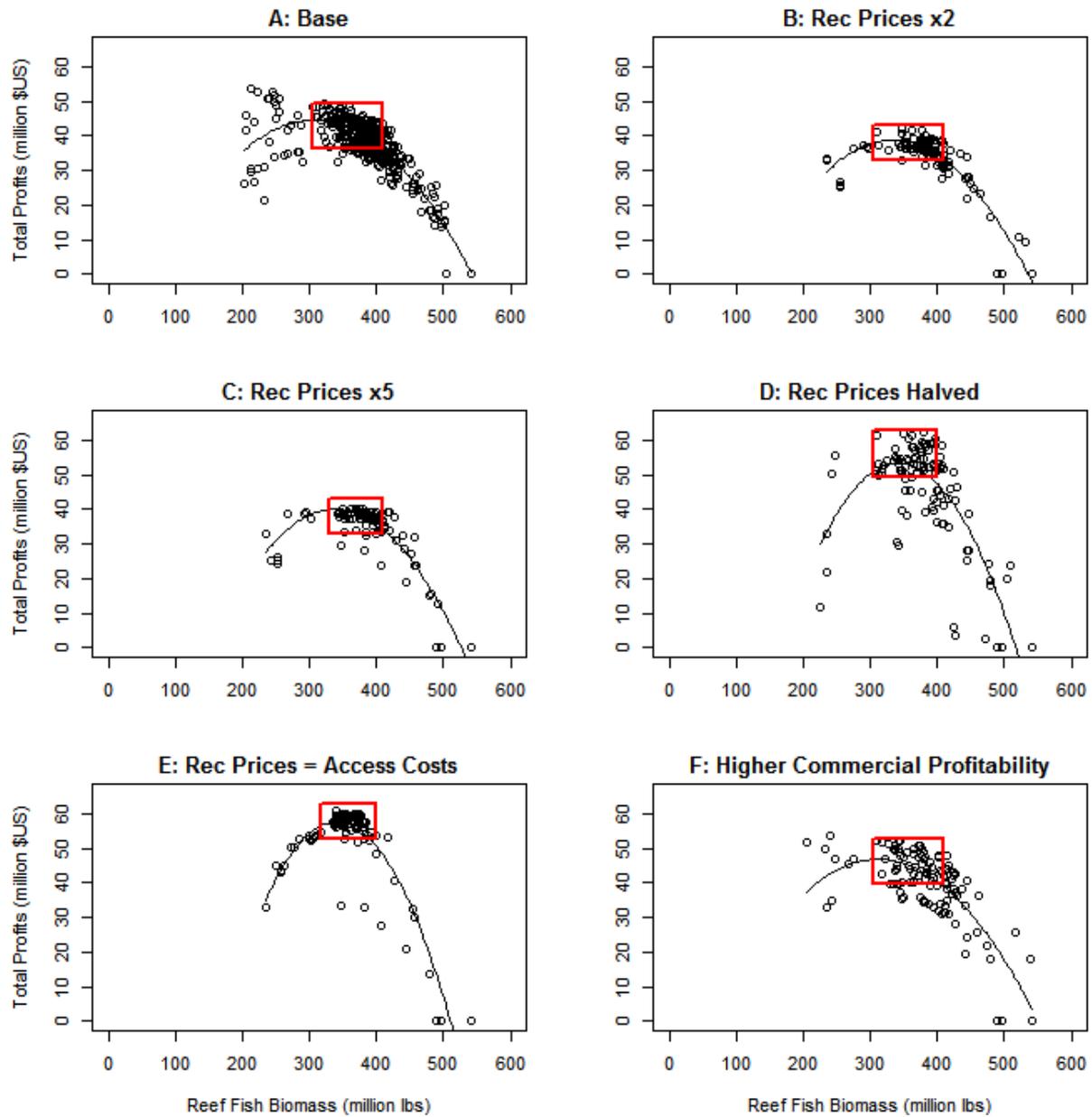


Figure 3-3. Tradeoff frontiers for the base configuration with all prices equal and profits provided in Table 3-1 and 5 sensitivity runs of various recreational prices and commercial profitability. The red squares indicate areas where both reef fish biomass and total profits are high.

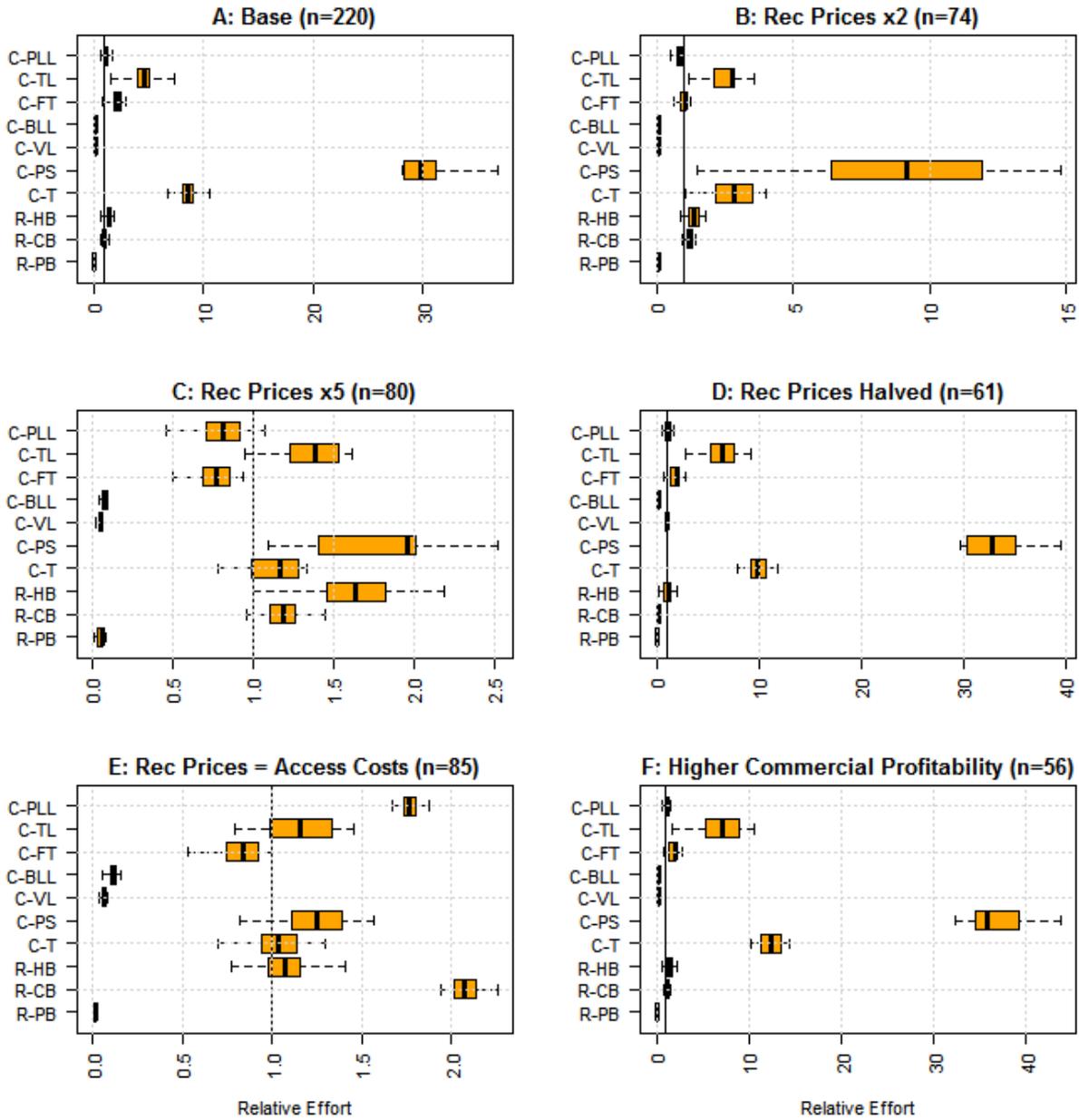


Figure 3-4. Effort configurations from optimization runs within the optimal zones of the tradeoff frontiers (red boxes in Figure 3-3). Whiskers are the lower and upper quartiles, the box is the 95% confidence interval, and the line is the average. Fleet codes are as follows: C indicates commercial and R indicates recreational, PLL = pelagic longline, TL = pelagic troll line, FT = fish traps, BLL = bottom longline, VL = vertical line, PS = purse seine, T = trawl, HB = headboat, CB = charter boat, PB = private boat.

A: Base scenario, equal prices

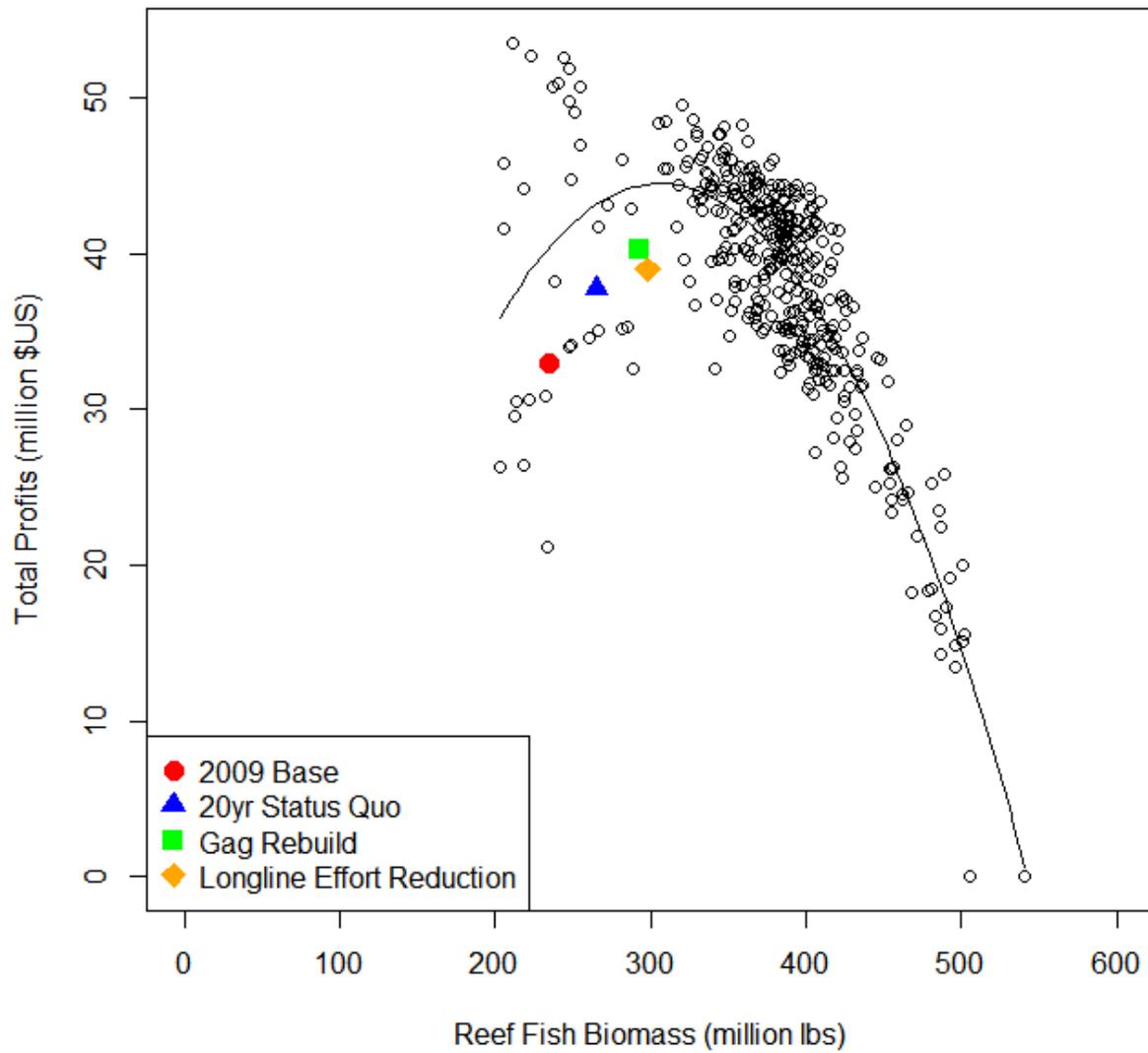


Figure 3-5. Tradeoff frontier for base scenario along with the 2009 base values and those at the end of three Ecosim simulations described in chapter 2.

CHAPTER 4 EVALUATION OF MARINE PROTECTED AREAS IN THE EASTERN GULF OF MEXICO USING AN ECOSPACE FISHERIES MODEL

Introduction

Marine protected areas (MPAs) and other types of spatial closures are becoming more important tools for managing living marine resources. Traditional fisheries management tools such as quotas, size limits, bag limits, and seasonal closures have sometimes been unsuccessful and can create a very complex set of regulations that must be followed and also enforced. For example in the Gulf of Mexico, bag and size limits for Gag Grouper (*Mycteroperca microlepis*) are ineffective because of high discard mortality (Tetzlaff et al. 2013), and the multispecies nature of the reef fish fishery can make species-specific seasonal closures and catch quotas a poor management choice. Marine protected areas are particularly useful in such cases because they can prevent bycatch and discard mortality. Conservationists, scientists, and even fishers have joined the call for MPAs as a means to protect habitat, increase biodiversity, rebuild fish stocks, enhance yield, improve our understanding of the system, buffer against uncertainty, and simplify the regulatory environment (Hilborn *et al.* 2004). Opponents of MPAs express skepticism about their biological and ecological benefits and perceive them to have negative impacts on local fishing communities, especially when MPAs are promoted without sufficient evaluation (Agardy et al. 2003; Stevenson et al. 2013).

The use of MPAs by the Gulf of Mexico Fisheries Management Council (GMFMC) goes back 30 years and in 2004 there were 16 MPAs designated by the GMFMC. These are closed to at least some forms of fishing and designed to prevent gear conflicts, increase yield, protect habitat, and reduce fishing effort (Coleman *et al.* 2004). Other reserves such as National Marine Sanctuaries, National Estuarine Research

Reserves, National Parks, and National Wildlife Refuges were created to preserve areas of special significance due to their ecological, scientific, socio-cultural, or aesthetic qualities. Although not designed to achieve fisheries management objectives, these areas (as well as any local and state aquatic preserves) do limit access and provide some refuge from fishing. Additional MPAs in the Gulf of Mexico have been proposed including a network of oceanographically linked areas that extend from Central America to North Carolina, with several sites positioned in the Loop Current on the West Florida Shelf (Ritchie and Keller 2008).

Existing guidelines for planning and evaluating MPAs are largely based on a broad set of generic, but practical indicators intended to support an adaptive management framework, with less emphasis on sophisticated scientific methods (Hocking, *et al.* 2000; Pomeroy *et al.* 2004). However, simulation models do exist that can provide a more quantitative evaluation of MPA effectiveness. For example, the impact of marine reserves on vessels and reef fishes in the Gulf of Mexico was evaluated using both descriptive and dynamic structural models (Smith *et al.* 2006; Smith *et al.* 2007). Spatially-explicit population dynamic models (e.g. Fishmod and EDOM) that incorporate fish movement patterns along with the distribution of fishing effort have been used to evaluate MPAs (Martell *et al.* 2000) with some application to the Gulf of Mexico (GMFMC 2008; Tetzlaff in prep). Whole ecosystem models that account for population dynamics, movement, spatio-temporal habitat distribution and associations, trophic interactions, and fleet economics are capable of a more comprehensive analysis of MPAs (Kaplan *et al.* 2012; Le Quesne *et al.* 2008; Martell *et al.* 2005; Steenbeek *et al.* 2013; Walters *et al.* 2010). Incorporating these and other types of models into the early

designing stages of MPAs can make the planning process more efficient and enhance stakeholder involvement.

In this chapter, I expand the EcoSim model described in Chapters 2 and 3 to the spatially explicit Ecospace model in order to investigate spatial management options on the West Florida Shelf. Ecospace ecosystem models allow consideration of the effects of MPA establishment on all functional groups and fisheries within a system while accounting for trophic interactions, movement rates, and relationships to key habitat types. The objectives of this chapter were to evaluate current MPAs designed to impact reef fish resources in the eastern Gulf of Mexico and identify tradeoffs between MPA size or shape, biomass, and catch. I do not attempt to predict the optimal spatial placement of MPAs, but instead identify targets for the amount of closed area which can enhance stock biomasses while still maintaining quality fisheries. This model is intended to provide a starting point for MPA planning and can be used as a tool for evaluating specific spatial management options, preferably within an adaptive management workshop (AEAM) environment that incorporates scientific information and stakeholder input.

Materials and Methods

Ecospace Modeling Approach

The Ecopath with Ecosim (EwE) modeling framework includes a spatially explicit component, Ecospace, which replicates the biomass dynamics of Ecosim over a grid of spatial cells and represents mixing (dispersal, migration, ontogenetic habitat shifts) of biomass among the cells while also including trophic interaction processes and spatial effort dynamics including effects of spatial variation in fishing costs (Walters *et al.* 1999). Ecospace is an extension of Ecosim and requires additional data input for dispersal

rates, migration patterns, and habitat preferences along with maps for habitat, port locations, and MPAs. In each MPA, fishing may be prohibited for one or more gears and for all or part of the year, thus allowing for the evaluation of seasonal area closures and those where only certain types of fishing are excluded.

In Ecospace, the biomass dynamics of group i in cell k are predicted by

$$\frac{dB_{ik}}{dt} = e_i Q_{ik} - Z_{ik} B_{ik} - \sum_k m_{ikk'} B_{ik} + \sum_{k'} m_{ik'k} B_{ik'}$$

where Q_{ik} is the total food consumption rate by group i in cell k and Z_{ik} is the total mortality rate of group i in cell k . The consumption rate is calculated as the sum over all prey types and the mortality rate is calculated as the sum over all predators and fisheries plus unexplained mortality. Instantaneous movement rate, $m_{ikk'}$, represents the movement of group i from cell k to k' . The first summation represents movement out of cell k and the second summation represents movement into cell k from four neighboring cells k' ; mixing across water boundaries is based on assuming a flat boundary condition for biomass groups, and no boundary crossing by multistanza (age structured) groups.. For primary producers, spatial variation in productivity can be represented by providing maps of constant or time varying relative primary production rate, e.g. from satellite chlorophyll data.

The movement rates, m (km/yr), are considered to be dispersal due to random movements within and among cells. Movement associated with directed migration patterns is represented by distortion of the rates so as to reduce mixing away from preferred seasonal locations and increase rates toward such locations. Dispersal rates can be modified to decrease movement into cells with 'bad' habitat types and increase rates of movement into cells with more suitable habitat. In earlier versions of Ecospace,

this was accomplished by designating each cell as a discrete habitat type and then specifying the preference of each group for those habitat types. In reality, multiple habitats types are present in each cell, especially when spatial cells are large, and this approach was not capable of capturing habitat gradients and the preferences by species across those gradients. In this analysis, I utilized the new habitat capacity model in Ecospace that allows for a more flexible and realistic representation of species distribution patterns as they relate to environmental characteristics (Christensen *et al.*, in prep.). In the habitat capacity model, each species may be assigned one function for each environmental map layer and these functions are combined across all layers to produce a single capacity map. The values of the computed capacity map are then used to determine the relative foraging arena size (or “carrying capacity”) of each cell for each group, which then affects movement between adjacent cells such that movement toward areas with more foraging habitat is favored. This is a drastic improvement over the old method because users can now bring in habitat layers in the form of GIS referenced raster grids and incorporate survey data to define the relationships of species to those habitats.

The model has also overcome some computational inefficiencies and can now include spatial representation of age structure using either a continuous mixing-rate model or an individual-based modeling (IBM) approach (Walters *et al.* 2010). The spatial simulations in this study were done using the IBM model for species with multiple age stanzas whereas the spatial biomass-dynamics of other groups were represented by the basic Ecospace biomass equation above. In Ecosim, changes in numbers and relative body weights of age cohorts are modeled over a monthly time step (Walters *et*

al. 2008). In Ecospace, the numbers in each age class ($N_{a,t}$) for multistanza species are divided into packets of individuals and the age and size dynamics of each packet are predicted over time while keeping track of the cell that each packet is using at the start of each simulated month. This approach is intended to provide a more realistic representation of fine scale trophic-interaction effects and also enables the model to form distinct local populations with varying age and size structures. This has an effect on the analysis of MPAs because it allows for populations of older and more fecund fish to accumulate in designated cells and thus become local areas of high reproduction.

In ecospace, the fishing effort of each fleet is distributed over space according to the net benefits gained from exploiting a given cell. This is accomplished using a logit choice or gravity model where total effort is proportioned among cells based on the relative profitability of fishing in each cell. Profitability is defined as the sum over all groups of the product of biomass, catchability, and price divided by a spatial cost of fishing. Additionally, fleets can be designated to operate in one or more habitat types and subject to any number of area closures.

I used the Ecosim model described in Chapter 2 to conduct spatial simulations in Ecospace. The model consists of 70 biomass pools and emphasized the valuable reef fish fisheries of the eastern Gulf of Mexico. Gag Grouper, Red Grouper (*Epinephelus morio*), Black Grouper (*Mycteroperca bonaci*), and Yellowedge Grouper (*Epinephelus flavolimbatus*) were each divided into 3 age stanzas and Spanish Mackerel (*Scomberomorus maculatus*), King Mackerel (*Scomberomorus cavalla*), and Red Snapper (*Lutjanus campechanus*) were all divided into juvenile and adult age stanzas. There are 14 fishing “fleets” including the bottom longline and vertical line ‘bandit’

commercial fisheries that target groupers and snappers as well recreational private boat, charter, and headboat fisheries. This model inherits vulnerability exchange rates from a historical Ecosim simulation (1950-2009) that included a forcing function to drive phytoplankton dynamics and was calibrated to time series of biomass and catch.

The geographic extent of the Ecospace model is 25-30.5 degrees north and 81-87.5 degrees west with a spatial resolution of 10 minutes (= 0.167 degrees or appx 20 km²) and has dimensions of 34 rows by 40 columns (Figure 4-1). This covers an area from the Florida Panhandle south to, but excluding, the Florida Keys and extends from shore out to a depth of 250 m. A bathymetry map was obtained from the NOAA National Geophysical Data Center Coastal Relief Model (NOAA 2013). A rugosity map for the WFS, representing the average elevation change between a grid cell and the eight neighboring grid cells (m/cell), was available from the United States Geological Survey (Robbins *et al.* 2010). I downloaded time-averaged maps for sea surface temperature (11 micron day) and chlorophyll-a from the MODIS aqua satellite using the NASA Giovanni Interactive Visualization and Analysis website (Acker and Leptoukh 2007). Salinity fields were obtained from the HYCOM + NCODA Gulf of Mexico hydrodynamic model and for each cell values were averaged across all depths for that day and then across all days from 2008-2012. I used the 'raster' package in R to crop, aggregate, and resample all gridded data (rugosity, temperature, chlorophyll-a, and salinity) to match the spatial dimension and resolution of the 10 min grid cell bathymetry map (Figure 4-2).

Dispersal in Ecospace was considered to be the result of random movements and ontogenetic habitat shifts rather than directed seasonal migrations. When available,

tagging data were used to approximate dispersal rates. For example, 50% of gag tagged in the southeastern Gulf of Mexico were recaptured within 2 months and 80% had traveled 9 km, which converts to about 50 km/yr (Ortiz *et al.* 2006). Greater Amberjack were observed to travel 0.3-1.0 km/d (MARMAP 2007) whereas some King Mackerel marked and recaptured in the southeast U.S. and Gulf of Mexico traveled at least 2,000 km with an average time at large of 360 days (Schaefer and Fable 1994). For most groups where this information was not available, I borrowed from known groups that are similar. Juveniles tended to be restricted in their distribution to small and sometimes distant areas and we learned through diagnostic simulations that they needed to have rather high dispersal settings to prevent them from being trapped in poor habitat types. Dispersal for plankton groups remained at the default value of 300 km/yr (predicted for eddy diffusion and weak tidal mixing) while benthic invertebrates, infauna, seagrass, and detritus were assigned much lower rates (Table 4-1).

As described above, each species may be assigned one function for each environmental input map, in this case depth, rugosity, temperature, and salinity (Table 4-2; Figure 4-3; Figure 4-4). Most of the fish groups were assigned a depth function. In some cases, relationships were based on survey or observer information presented in SEDAR data and assessment workshop reports (<http://www.sefsc.noaa.gov/sedar/>). In other cases a simple linear relationship was used to represent basic inshore-offshore distributions. Rugosity was also used to influence the distribution of reef fish species. Unlike depth, relationships with rugosity were not readily available. Therefore, I established 4 functions with rugosity to represent preference for no relief, low relief, high relief, and high relief with asymptote. Most of the grouper and adult snappers were

assigned a preference for high relief habitat except for Red Grouper which prefer low relief habitat. The functional relationships with rugosity are based on expert opinion and personal communication with scientists that conduct routine sampling in the area. A salinity function was used to represent the distribution patterns of estuarine species groups. Temperature was used only for red snapper, black grouper, and lobsters as a means to represent broad north-south distribution patterns. Benthic invertebrate groups, zooplankton, and primary producers were not assigned any habitat capacity functions with the exception of seagrass which was restricted to cells shallower than 20 m. A habitat capacity map was then computed for each species where the cell values are the product of the functional responses for each environmental input. The computed capacity was then used to determine foraging arena area which affects movement between adjacent cells and represents habitat preferences and spatial distribution patterns.

Ecospace calculates the relative spatial cost of fishing for all fleets based on the sailing related costs specified in Ecopath and the distance from port locations to each cell. The cost structure of commercial fleets was based on surveys of annual revenues and expenditures of commercial fishing permit holders (Liese *et al.* 2009; SEFSC personal communication). The for-hire recreational fleets (charter boat and headboat) were treated as a commercial venture, and the cost structure was computed using the same allocation of expenditures as the commercial fleets and based on data provided in Liese and Carter (2011). The cost structure of private recreational fleets was based on the idea of consumer surplus using a survey of marine anglers in South Florida (Alvarez and Larkin, University of Florida Food and Resource Economics Department, personal

communication; Johns *et al.* 2001). For the commercial fleet, I assigned port locations based on the total landings of all species from 2005-2009 made by each fleet in each county along the west coast of Florida. For the recreational shore and private boat fleets, all of the shoreline cells were assigned as ports. Charterboat and headboat services are more abundant in highly populated and tourist areas, especially those along the Panhandle from Panama City to Pensacola and along the southwest coast from Tampa Bay to Naples. All fisheries were allowed to operate anywhere over the grid except for shore-based anglers which were restricted to cells adjacent to the shoreline. All parameter values the ecospace model, including environmental layers and habitat capacity functions were saved in an access database, which can be examined and changed using the EwE 6.4 user interface. The model is available at the dropbox link https://dl.dropboxusercontent.com/u/31499559/WFS_EwE_files.zip.

Simulations

I evaluated the impact of six area closures that were designed either to protect reef fish species or regulate fisheries that target reef fishes, especially groupers (Figure 4-1). The Madison-Swanson and Steamboat Lumps Marine Reserves were implemented in 2000 in an effort to protect a portion of gag spawning aggregations and the offshore population of male gag. These are closed to all bottom fishing year round and combined they cover an area of approximately 750 km² (GMFMC 2003). The Edges extends northwest from the Steamboat Lumps reserve at the 40 fathom contour and encompasses an area of about 1,300 km². It was established in 2008 as part of Reef Fish Amendment 30B (GMFMC 2008) to address the overfishing of gag and prohibits all fishing from January through April, the peak spawning season for gag. The Florida Middle Grounds is a shallow (<60 m) high relief area located about 150 miles off

the coast of Florida between Tampa Bay and Cape San Blas and covers an area of about 1,200 km². In 1984 it was designated as a Habitat Area of Particular Concern (HAPC) and fishing with bottom gear (longlines, trawls, dredges, pots, traps) was prohibited to protect the sensitive octocoral community which is at its northernmost extent in North America (Coleman *et al.* 2004). Recreational and commercial hook and line fishing is permitted within the Florida Middle Grounds. Combined, these four areas represent about 2% of the entire West Florida Shelf.

Since 1990 longlines have been prohibited year round from fishing inside of the 20 fathom contour east of and the 50 fathom contour west of Cape San Blas, FL. In 2009, a seasonal area closure was established to reduce the number of interactions with endangered sea turtles by bottom longlines (GMFMC 2009) and from June through August bottom longlines are prohibited from fishing inside the 35 fathom contour east of Cape San Blas. The locations of closed areas were available online from the Gulf of Mexico Data Atlas as GIS referenced shapefiles (Frick 2011) and mapped to the Ecospace grid using the 'raster' package in R. Each of these six closures was evaluated independently and then all closures were evaluated together.

In addition to the area closures described above, I also evaluated a suite of hypothetical MPAs that were closed to fishing by all fleets for the entire year. These MPAs varied in their size and placement and were selected with a random point pattern process (using the spatstat package in R). Each MPA was square and they could overlap so long as the centers did not lie within the radius of other MPAs and so long as the edges were at least 1 grid cell away from land and deep water boundaries. If a randomly selected point did not meet those criteria, it was discarded and another point

was generated. This was repeated until the desired number of MPA blocks were mapped. I generated 100 MPA grids ranging from few large MPAs to many small MPAs with 20 each of the following randomized configurations: 3 – 7x7 blocks (XL), 5 – 5x5 blocks (L), 10 – 3x3 blocks (M), 20 – 2x2 blocks (S), and 50 – 1x1 blocks (XS). Although each randomly placed MPA was square, it was allowed to be adjacent to and partially overlapping other MPA blocks which resulted in total closed areas of varying shapes and sizes and with different edge properties (Figure 4-5).

Simulations of hypothetical MPA's were conducted where the total percent area closed ranged between 23 and 91%. This created a larger range of closure sizes and shapes for evaluation than was possible to generate given the block size and/or separation constraints of the point pattern process method. For all random MPA grids, an edge ratio was calculated as the total number of open cells along the outside perimeter of all MPA blocks divided by the total number of protected cells and can be as high as 4 when all MPA cells are isolated. Percent area closed was taken simply as the number of closed cells divided by the total number of cells. The purpose of these evaluations was to define the relationships and tradeoffs associated with the size and shape of MPAs.

Results

The computed habitat capacity maps reflect the relative foraging areas in each cell as determined by the combined relationships with rugosity, depth, temperature, or salinity (Figure 4-6). Pelagic species such as Cobia and mackerels that only had a relationship with depth exhibited high capacity over a broader extent of the shelf. In contrast, reef associated species (groupers and snappers) were estimated to have high capacity over restricted depth ranges and in patchy reef habitats.

Predicted equilibrium spatial distributions of biomass generally reflect the computed habitat capacity grids with a few exceptions (Figure 4-7). For instance, adult Gag Grouper were distributed much closer to shore than what was predicted from the habitat functions alone. This is a reflection of ontogenetic movements of juveniles out of Tampa Bay, Charlotte Harbor, and Apalachicola Bay to adjacent nearshore patch reefs as sub-adults and then into deeper water as adults. In other words, the predicted spatial distribution patterns were driven by both the habitat capacity functions and also recruitment of younger fish from areas closer to shore.

The existing closures designed to enhance gag populations had little impact on gag biomass (Figure 4-8) and did not increase stock size by more than 2% when compared to simulation with no MPAs. The Madison-Swanson, Steamboat Lumps, and The Edges reserves occupied 6 grid cells total which is equivalent to just over 1% of all water cells. These locations were selected for protection because they are considered to be areas where gag aggregate to spawn. The Edges, which is closed only during the spawning season, was even predicted to have a negative impact on gag, however, when gag egg production was doubled during the spawning season (January – April) the effect of the Edges closure on Gag biomass was positive though still small. These results suggest that the existing area closures intended to enhance gag populations need to be much larger to be effective, and those designed to protect spawning fish can have more impact than closures that do not coincide with spawning seasons and areas.

The longline closures had the largest positive effect on red grouper, with the 20 fathom longline restricted area resulting in 43% more biomass than simulations with no area closures. Vermilion Snapper (*Rhomboplites aurorubens*), other shallow water

grouper (SWG), and goliath grouper (*Epinephelus itajara*) all had higher biomasses with the longline closures, but to a lesser degree. Tilefish (*Lopholatilus spp.*) and Yellowedge Grouper biomasses were nearly 50% lower after 20 years with longline restrictions because effort was forced out into deeper waters. Other shallow water species such as Black Seabass (*Centropristis striata*) and the more pelagic associated Cobia (*Rachycentron canadum*) and Greater Amberjack (*Seriola dumerili*), which are not targeted by the longline fishery, were slightly lower with the longline closure because of an increase in predators and/or competitors.

The MPA block designs had mixed effects among pelagic and demersal species, deepwater and shelf species, and heavily exploited versus less targeted species. Cobia and mackerels remained within +/- 2% of the biomass predicted without any MPAs while some reef fish increased by over 50% in the XL scenarios (Figure 4-9). The reef fish with the largest response were those species that are more heavily exploited including Gag, Red Grouper, Red Snapper, and Vermilion Snapper. Deepwater species declined in response to any area closures perhaps due to trophic impacts or bias in the point pattern selection process for large closures in the mid-shelf region that would have forced effort far offshore. Black Seabass, Goliath Grouper, and other SWG, which are under less fishing pressure, declined by up to 5% in the XL scenarios because of a buildup of predators and competitors.

Three general patterns emerged in the relationship between the total area closed and biomass change (Figure 4-10). Species either a) declined gradually up to about 50% closure and then decreased sharply, b) remained mostly stable even at large closures, or c) increased slowly from 20% to 60% closures before rising sharply.

Species that exhibited a declining relationship included Cobia, other SWG, Goliath Grouper, and Black Seabass. These species did not benefit from large closures because the increase of predators and competitors outweighed the reduction of fishing mortality, which was already low for these species. King and Spanish Mackerels, Tilefish, Yellowedge Grouper, and other deep water grouper (DWG) remained relatively stable (+/- 5%) across all potential closure sizes. Pelagic species such as mackerels are not expected to be effected by area closures due to high movement rates. Deep water species showed little change because they do not overlap spatially with the reef fish species that benefit from the MPAs, and the majority of closed cells were located in shallower depths. Lastly, Red Snapper, Vermillion Snapper, Gag Grouper, Red Grouper, and Greater Amberjack are all heavily exploited by the commercial and recreational reef fish fisheries and therefore had a positive response to the area closures, especially after about 50% of the area was closed.

There was also a stark contrast between the MPA block designs and those generated by simple random cell selection. For example, large MPA blocks that closed just over 20% of the area were predicted to generate as much biomass of gag as a random selection of 70% of the cells (Figure 4-10). When comparing biomass with edge ratios, scenarios with larger single MPAs generated by the blocked pattern (i.e. those with low edge ratios) resulted in stronger effects than the widely scattered but large MPAs chosen at random (Figure 4-11). This was expected and illustrates the importance of edge to the efficacy of marine reserves.

In most cases total catch by each fleet declined when the closed area was large with the exception of the commercial pelagic longline and troll fisheries (Figure 4-12,

Figure 4-13). Predicted catch for pelagic fisheries remained mostly stable even at very large closures, because the species they target have high dispersal rates and therefore mix rapidly in and out of protected areas. While some of the scenarios resulted in drastic declines in catch, several of the large MPA block configurations reduced catch only slightly, and in some cases increased total catch. For example, longline catch was only about 10% lower in several of the L and XL scenarios and recreational private boats had higher catch even when about 25% of the total area was closed to fishing. Total catch from all fleets improved in a few of the MPA configurations but declined when more than 30% of the area was closed to fishing. Catch was less impacted by MPAs with large edge ratios because fleets had more access to closed cells (Figure 4-14).

Discussion

This paper described the first application of a trophic-dynamic ecosystem model to evaluating the efficacy and impacts of MPAs in Gulf of Mexico. The model predictions were consistent with the general understanding that a) MPAs have little impact on highly mobile species; b) large closed areas provide better protection to harvested species than small closed areas; c) heavily exploited species benefit more from closed areas than lightly exploited species; and d) increasing MPA area results in declining total catch except for species that would otherwise be overfished. Moreover, because Ecospace accounts for food web interactions, the model was able to demonstrate that competition and predation can cause biomass of some species to decline when predator biomass accumulates in protected areas. That is, MPAs may have a positive impact on some species and a negative impact on others, due to trophic interactions. Because biomass of fish spilled over into unprotected cells, some MPA scenarios had

little to no negative impact on the fishery and in some cases provided net benefits to the fishery. The win-win scenarios where there was gain in both biomass and catch usually required between 15%-30% of the WFS to be closed to fishing. Existing MPAs in the eastern Gulf of Mexico only account for about 2% of the total area, and the 15%-30% finding could be used by managers and stakeholders to as a guideline in design of more expansive system of MPAs that better balances the tradeoff between conservation and fishery productivity.

The spatial distribution patterns predicted by the Ecospace model represented broad inshore to offshore structure as well as fine scale spatial structure. For most reef fishes I assumed that only depth and rugosity drive their spatial distributions, but other factors could also be important such as submerged aquatic vegetation (Switzer *et al.* 2012), mangroves (Koenig *et al.* 2007), fine scale fractal structure (Cady 2007), or contiguousness of reef structures (Farmer and Ault 2011). While seasonal dynamics play a big role in the migration patterns of pelagic species (Houde 1976), reef fish on the West Florida Shelf have been observed to vary with depth rather than season (Smith 1976), although seasonal inshore-offshore migrations do occur for some species. Fine scale spatial structure was predicted mostly for juvenile groups and reef associated species that had a dome shaped relationship to both depth and rugosity. Gag was abundant from Tampa Bay to Charlotte Harbor as well as Apalachicola Bay, but less abundant in the Big Bend region near Cedar Key. The paucity in this region is consistent with survey data and could be explained by the openness of this area compared to the other semi-enclosed estuaries, influence from the highly variable Suwannee River, and perhaps that the vast expanse of sub-aquatic vegetation dilutes

gag densities or allows them settle further away from shore (Switzer *et al.* 2012). The habitat capacity model offers the flexibility to easily improve upon the spatial distributions as more data become available and the ability to represent spatio-temporal dynamics in habitat layers (temperature, salinity, chlorophyll-a) has improved (Steenbeek *et al.* 2013). The habitat capacity model and inclusion of time-dynamic habitat layers are expected to greatly improve the model through a better representation of seasonal spatial distribution patterns and enable us to address a wider range of environmental and policy issues.

The MPAs evaluated in this chapter ranged from small reserves designed with narrowly focused objectives (e.g. protect male Gag) to hypothetical, randomly placed closures of various sizes and shapes. In general, the model predicted MPAs to have less of an impact on highly migratory species than those with lower dispersal rates, as has been the conclusion of most other analyses of MPA benefits (Gerber *et al.* 2005; Hilborn *et al.* 2004; Walters 2000). This is because highly mobile species are more likely to leave MPAs as a result of random movements and the loss to the MPA is not balanced by random movements into the MPA from non-protected cells which on average have less fish. Thus, any attempt to protect migratory species with area closures may fail or would require even larger MPAs with lower edge ratios than those that may be effective for reef fishes. Nearly all of the block MPA designs outperformed closures of the same total area designed with randomly chosen cells. This supports the argument for large reserves over smaller ones (“SLOSS - single large or several small” debate), which has been controversial in the past (Diamond 1975; Simberloff and Abele 1982; Simberloff and Abele 1976). Spillover of adult fishes from MPAs does indeed

occur (Goni *et al.* 2008; McClanahan and Mangi 2000; Roberts *et al.* 2001) and is often touted as a benefit that MPAs can provide to fisheries. While some fleets did benefit from certain closures, overall my results indicated that closed areas will mostly lead to lower catches. However, the losses are disproportionately lower than the total amount of area closed, suggesting that the tradeoff between MPA size and catch is not severe and large area closures of about 15-30% of the shelf may actually be economically and politically feasible.

The current MPA's intended to enhance gag stocks were predicted to have very little impact on their biomass, and so far this seems to be the case for their recovery. This does not mean that the MPAs have offered no benefits to the population. For instance, Gag caught inside the Madison-Swanson Marine Reserve were observed to be larger, older, and composed of more males than outside the reserve (Koenig and Coleman 2013). Relative abundance was higher in areas closer to the reserve than farther away suggesting a spillover effect. Possible reasons that Madison-Swanson, Steamboat Lumps, and The Edges have done little to help rebuild gag stocks include insufficient spatial coverage, poaching, and stock declines in other regions (Koenig, presentation to council). The buildup of larger and more fecund 'packets' of fish inside closed cells is predicted using the Ecospace IBM option, however it is possible that the reproductive value of these locations was still underestimated in the model. Accounting for seasonal egg production did make The Edges closure more effective, but the model did not account for migration to and aggregation in these sites during the spawning season, the effect of improved sex ratios inside the reserves, or the transport mechanisms that successfully disperse larvae from these sites towards nursery areas

(Karnauskas *et al.* 2013). Despite these limitations of the model, the most likely reason for such a small impact on biomass is simply the small area of protection offered by these sites and the attraction of fishing effort to their borders. Furthermore, the results suggest that areas need to be about 5-10 times larger and with low edge ratios in order to substantially improve the Gag stock.

The 20 fathom closure was established in 1990 to reduce intense exploitation of spawning reef fish populations, however despite intense recreational fishing pressure only the longline fishery was affected by the regulation. The mixed impacts of this closure are a result of catch composition in the fishery and displacement of effort into other areas. The negative impact that the longline closures had on deepwater species, resulting from effort being displaced to deeper waters, should come as no surprise because this potential was recognized by the GMFMC prior to its implementation (Coleman *et al.* 2004). If sailing costs were higher for the longline fishery then perhaps less effort would be shifted onto deep water species and the biomass declines would not be as severe. In either case, this finding emphasizes the need for accurate information about operating costs and spatial effort dynamics when evaluating the impact of potential closed areas.

The evaluation of randomly placed MPAs was intended to define the likely responses of species and fisheries to closures of various sizes and shapes. By identifying a feasible range of size and edge requirements, this study has provided a starting point for more specific MPA considerations. For instance, the model suggests that MPAs between 15-30% of the area can increase population sizes without causing economic collapse of the fishery, and that edge ratios should be less than about 1.5. In

most cases, MPAs are located in some of the best habitats and would surely be more effective than what was predicted from randomly placed areas. Several international organizations have set targets to have 15-30% of the oceans closed to fishing in order to increase biodiversity and improve social benefits (Lubchenco *et al.* 2003; Wood *et al.* 2008). While percent area closures as large as 30% may seem large, they do not limit access any more than extended seasonal closures currently in place for some species (e.g. Red Snapper) and could potentially simplify the complex regulations imposed in the Gulf of Mexico. In the future, simulation models like Ecospace can be used to optimize the placement of marine protected areas (Beattie *et al.* 2002; Christensen *et al.* 2009) but should only be a part the participatory process that includes stakeholder input and traditional ecological knowledge (Drew 2005; Scholz *et al.* 2004). Furthermore, experimental design and adaptive management of MPAs would provide a wealth of scientific information that could be used to improve models and make even better decisions about their size and placement.

Table 4-1. Dispersal Rates used in the Ecospace spatial simulation model.

Group name	Base dispersal rate (km/year)	Group name	Base dispersal rate (km/year)
dolphins	30000	black sea bass	100
seabirds	3000	reef carnivores	50
large coastal sharks	2000	reef omnivores	50
small coastal sharks	1000	coastal piscivores	100
rays/skates	500	large coastal carnivores	1000
billfish/tunas	3000	small coastal carnivores	50
oceanic small pelagics	1000	coastal omnivores	500
cobia	1000	sardine-herring-scad	500
king mackerel juv	300	anchovies-silversides	500
king mackerel adult	2000	mullet	100
Spanish mackerel juv	300	squid	500
Spanish mackerel adult	2000	shrimp	100
jacks, whaoo, dolphinfish	3000	lobsters	50
red snapper juv	300	crabs	100
red snapper adult	100	octopods	100
vermillion snapper	100	stomatopods	100
other snapper	50	echinoderms/large gastropods	10
tilefish	50	bivalves	10
YEG 0-1	300	sessile epibenthos	10
YEG 1-3	300	small infauna	10
YEG 3+	50	small mobile epifauna	50
other DWG	50	meiofauna	10
gag 0-1	300	small copepods	300
gag 1-3	300	mesozooplankton	300
gag 3+	50	carnivorous zooplankton	300
red grouper 0-1	300	ichthyoplankton	300
red grouper 1-3	300	carnivorous jellyfish	300
red grouper 3+	50	microbial heterotrophs	300
black grouper 0-1	300	macroalgae	10
black grouper 1-3	300	microphytobenthos	10
black grouper 3+	50	phytoplankton	300
other SWG	50	seagrass	5
goliath	100	water column detritus	300
triggerfish	100	sediment detritus	1
amberjack	300	dead discards	300

Table 4-2. Habitat Capacity function assignments. The numbers correspond to the shapes in figures 2 and 3.

Group name	depth	rugosity	temperature	salinity
1 dolphins	19			
2 seabirds	19			
3 large coastal sharks	19			
4 small coastal sharks	19			
5 rays/skates	19			
6 billfish/tunas	18			
7 oceanic small pelagics	18			
8 cobia	10			
9 king mackerel juv	2			
10 king mackerel adult	10			
11 Spanish mackerel juv	1			
12 Spanish mackerel adult	6			
13 jacks, whaoo, dolphinfish	3			
14 red snapper juv	10	11	12	
15 red snapper adult	9	4	12	
16 vermilion snapper	15	16		
17 other snapper	10	4		
18 tilefish	14			
19 YEG 0-1	2	4		
20 YEG 1-3	3	4		
21 YEG 3+	13	4		
22 other DWG	13	4		
23 gag 0-1	1	4		
24 gag 1-3	2	4		
25 gag 3+	3	4		
26 red grouper 0-1	1	5		
27 red grouper 1-3	6	5		
28 red grouper 3+	7	5		
29 black grouper 0-1	1	4	8	
30 black grouper 1-3	2	4	8	
31 black grouper 3+	3	4	8	
32 other SWG	3	4		
33 goliath	3	4		
34 triggerfish	7	4		
35 amberjack	10	4		
36 black sea bass	7	5		
37 reef carnivores		16		
38 reef omnivores		16		
39 coastal piscivores	1			
40 large coastal carnivores	1			
41 small coastal carnivores	1			
42 coastal omnivores	1			
43 sardine-herring-scad	19			
44 anchovies-silversides	1			
45 mullet	1			
46 squid				
47 shrimp	21			
48 lobsters			8	
49 crabs	19			20

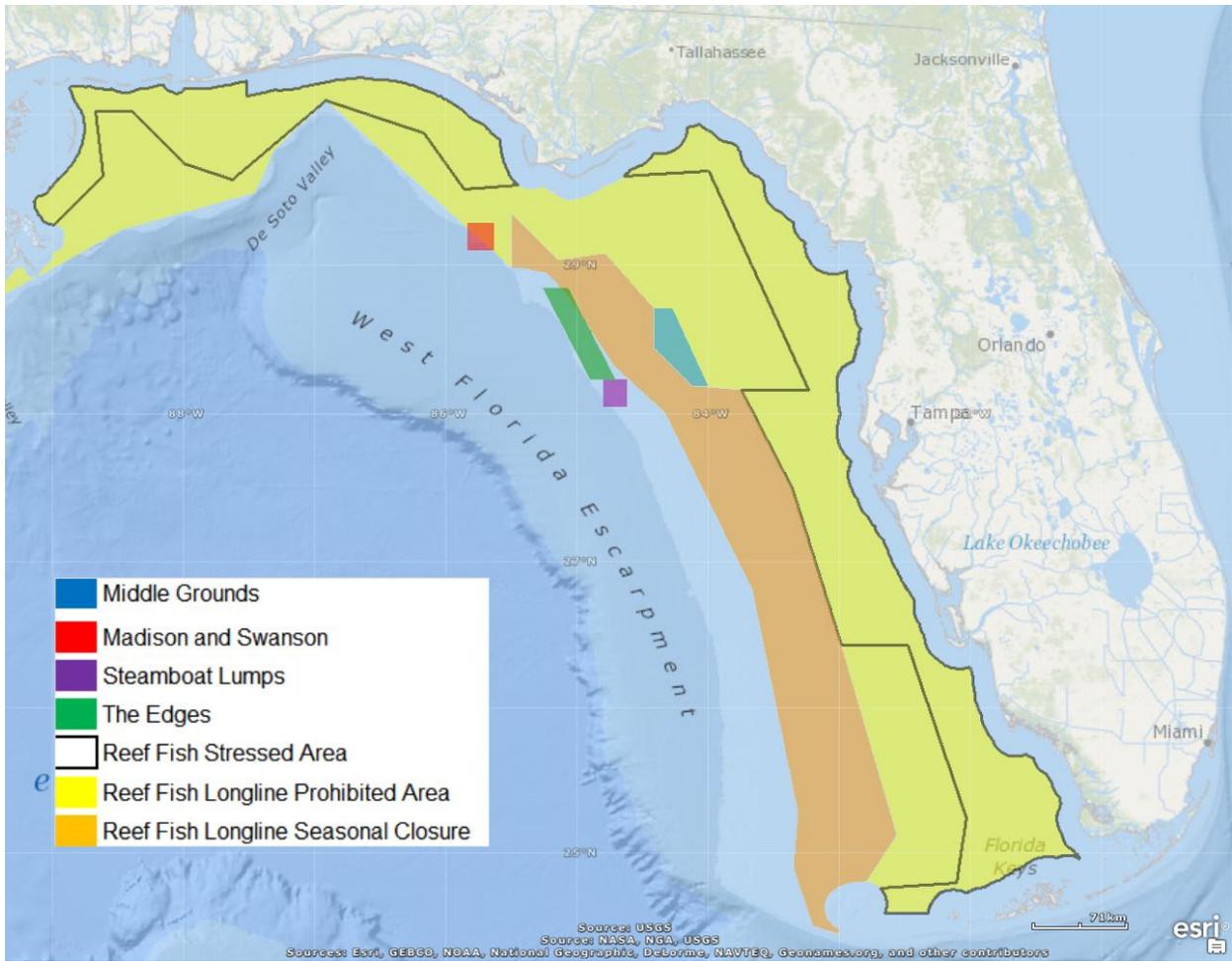


Figure 4-1. Map of the WFS and the MPAs considered in this analysis.

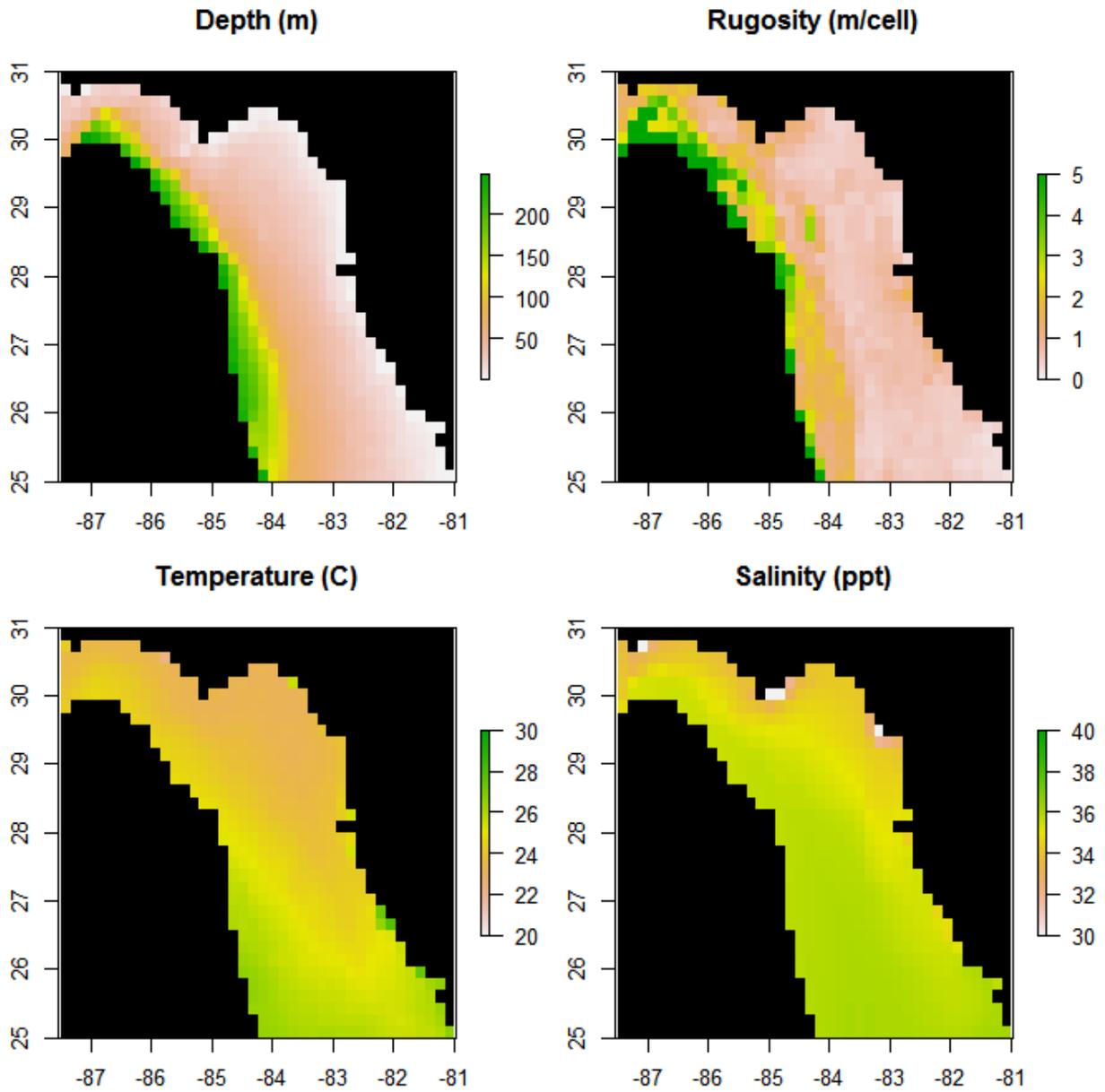


Figure 4-2. Environmental Maps for the Ecospace habitat capacity model.

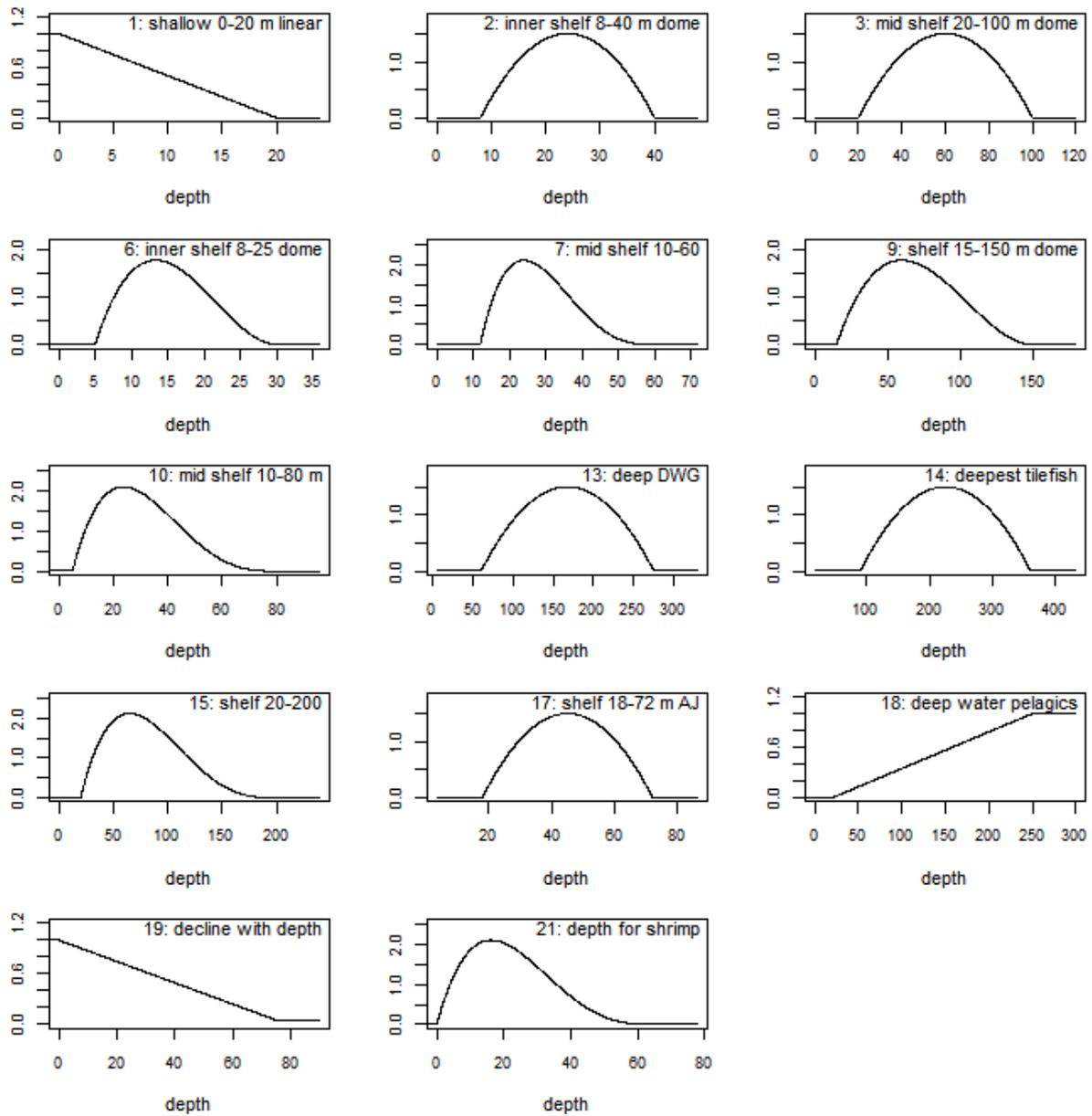


Figure 4-3. Depth functions used in the Ecospace habitat capacity model. The y-axis indicates the relative foraging arena area (a multiplier in the foraging arena functional response equation).

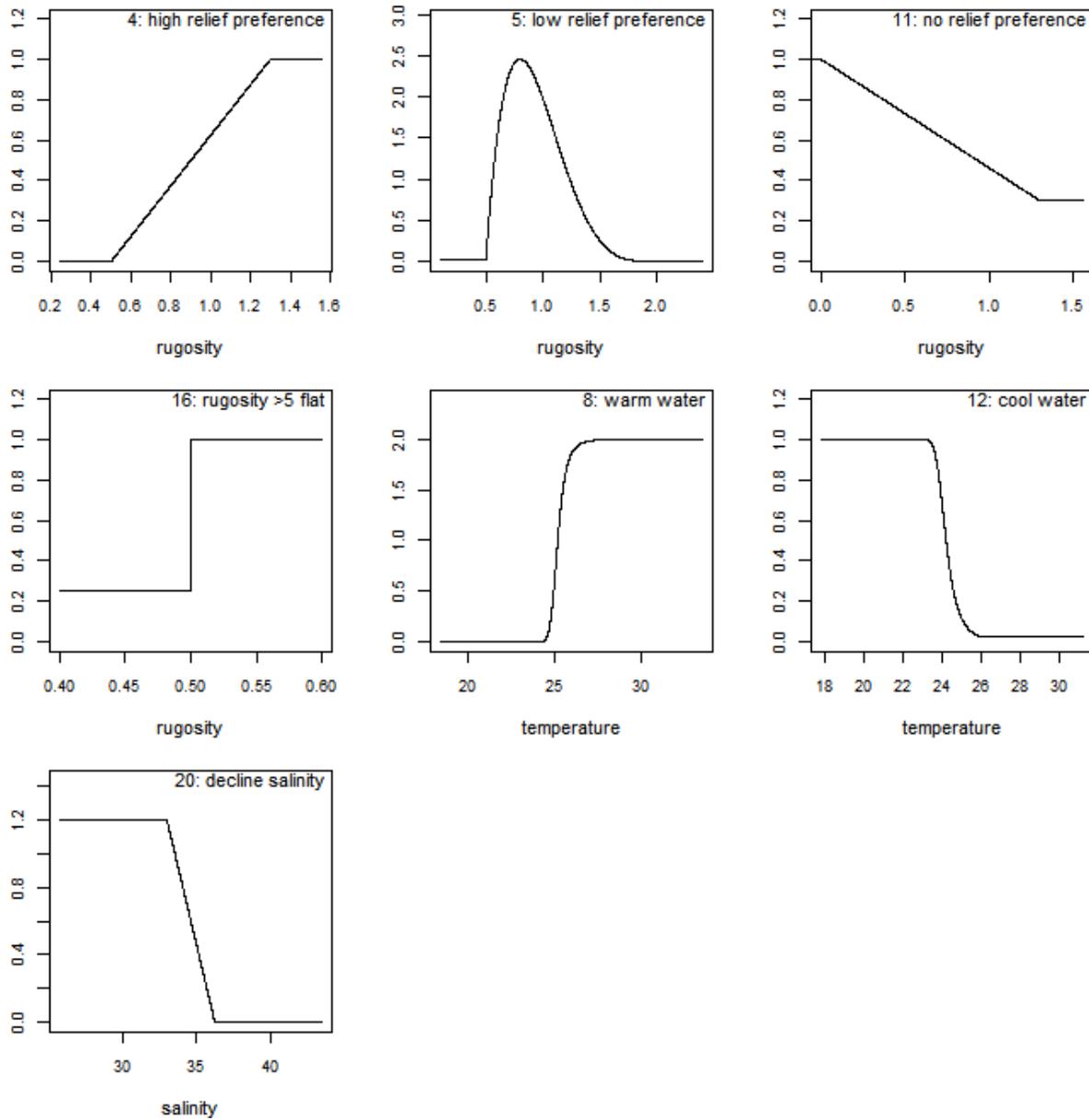


Figure 4-4. Rugosity, temperature, and salinity functions used in the habitat capacity model. The y-axis indicates the relative foraging arena area.

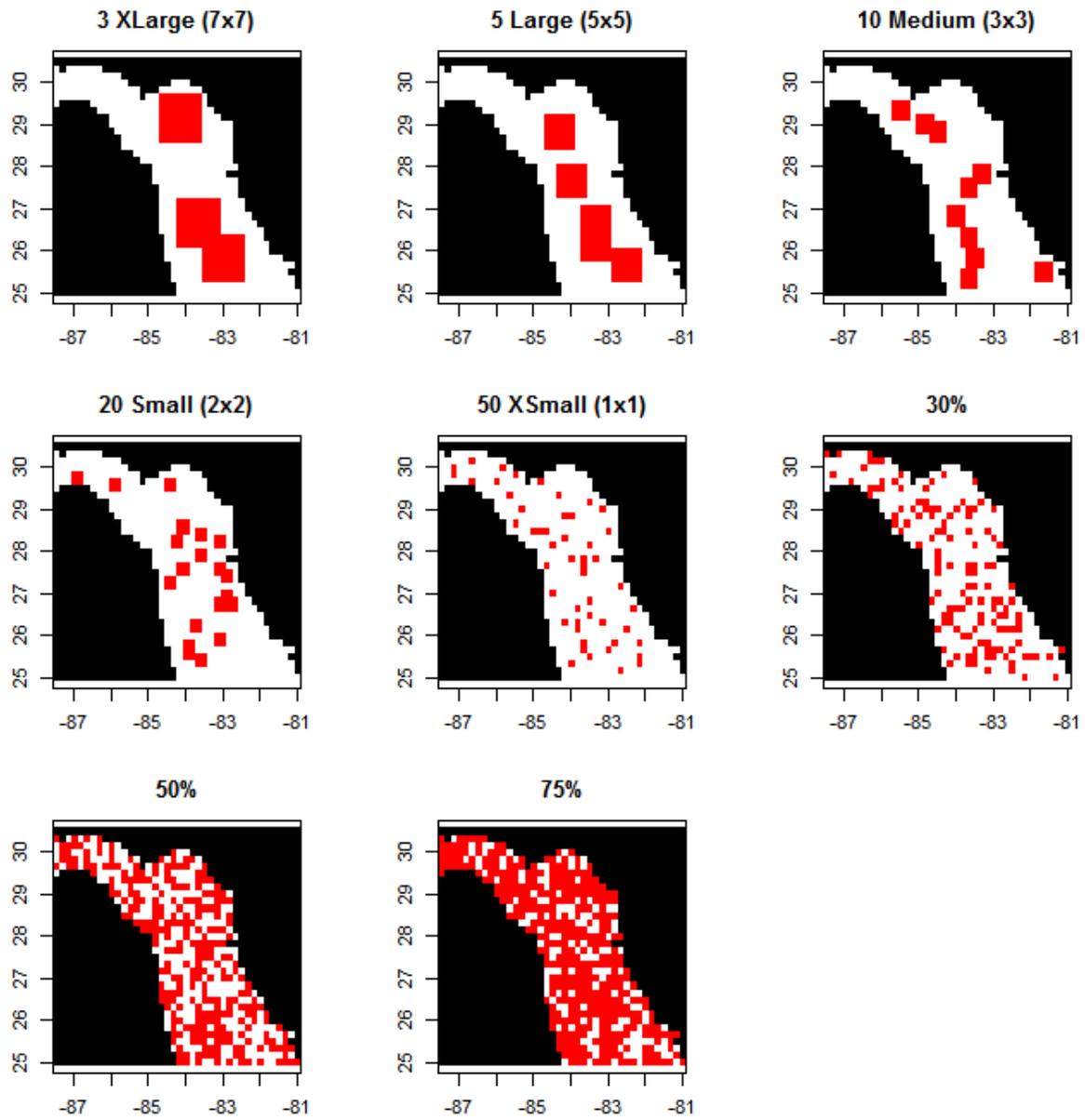


Figure 4-5. Examples of random MPAs. The black areas indicate deep water cells west of and land cells east of the West Florida Shelf.

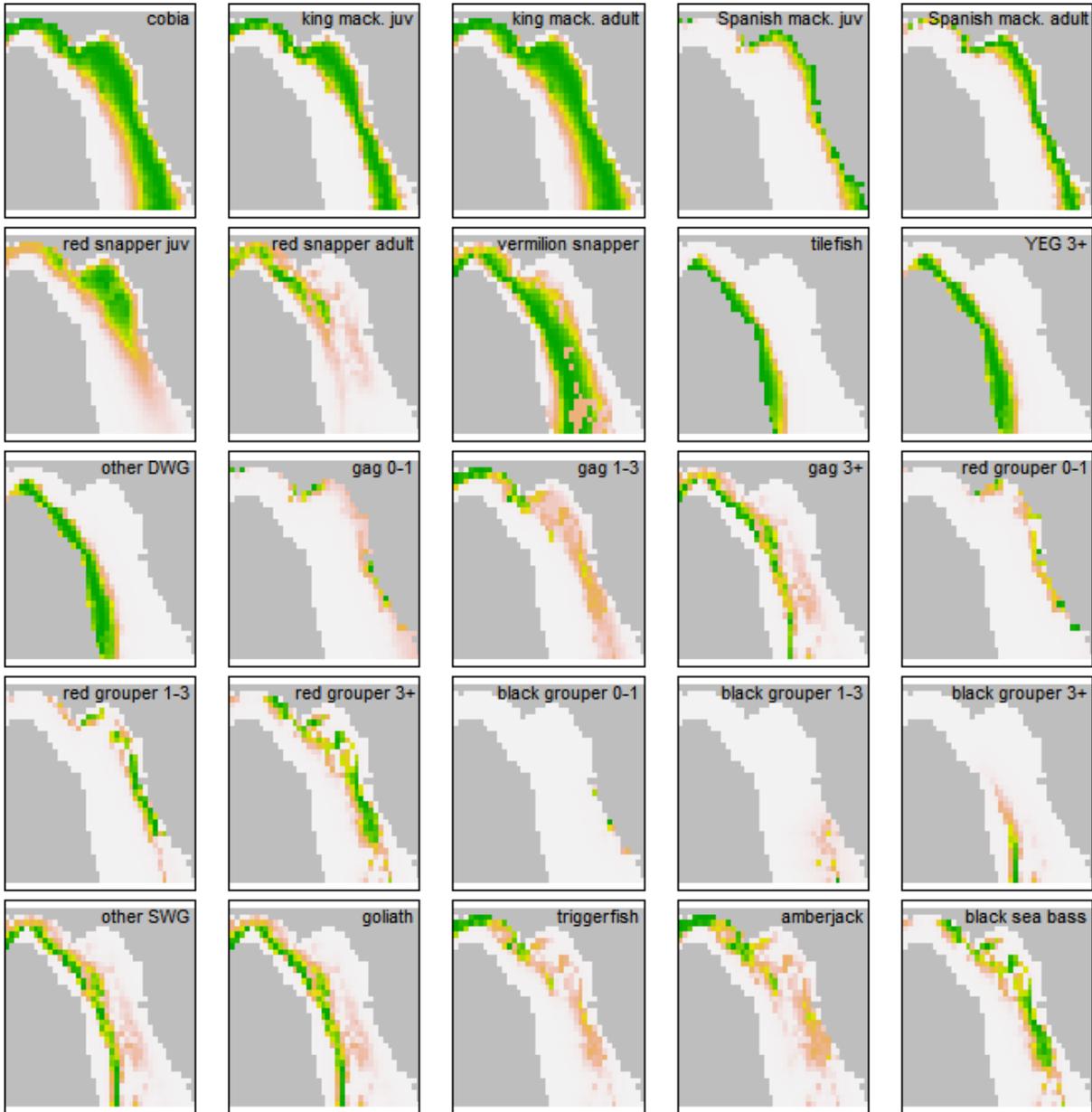


Figure 4-6. Computed capacity map for 25 biomass groups based on the relationships with depth, rugosity, temperature, and salinity described above.

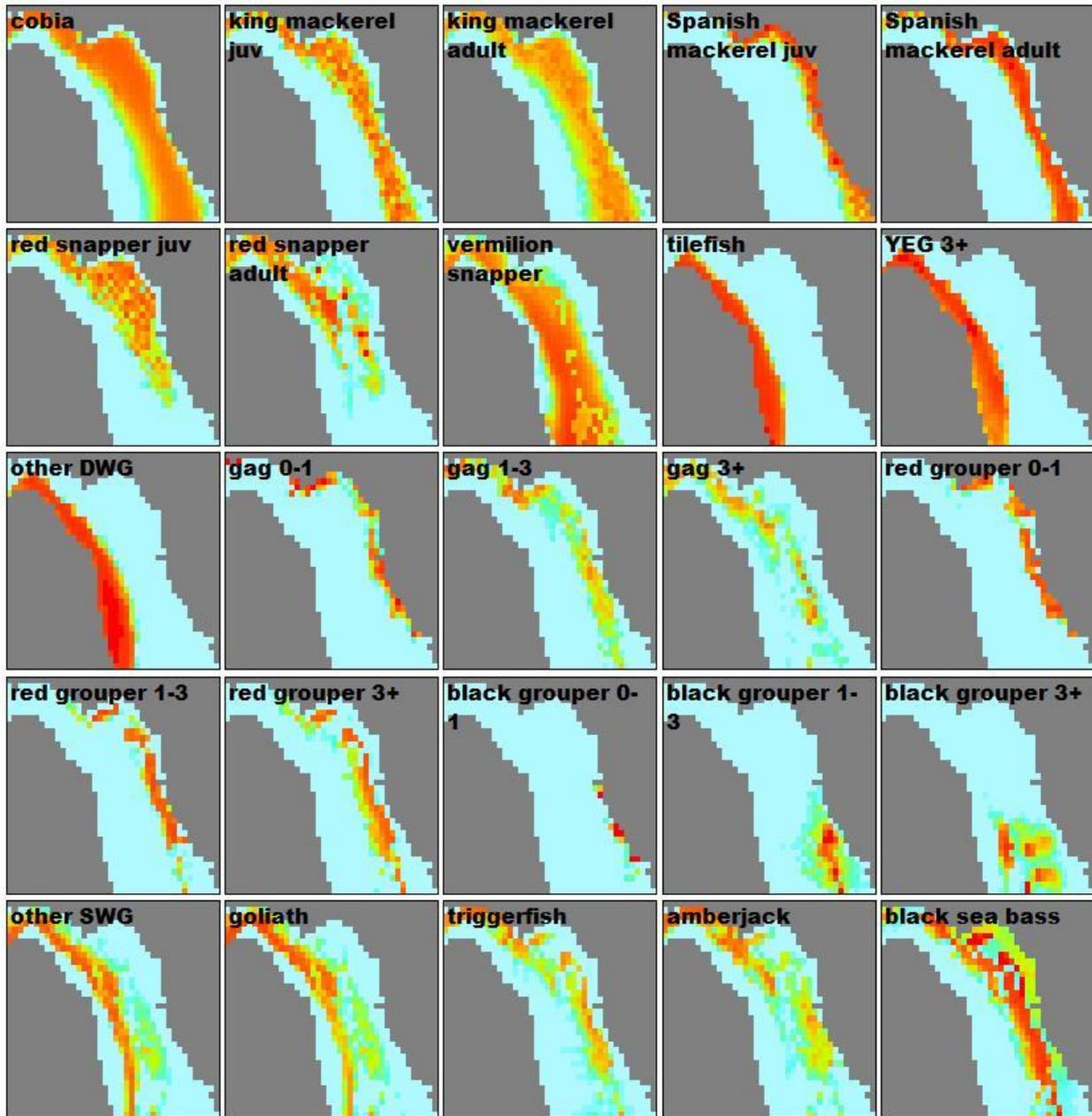


Figure 4-7. Predicted average spatial distributions at system biomass equilibrium under no MPAs made with the Ecospace IBM model. IBM packet positions are not shown to ease visualization; cells shown in red had more packets on average.

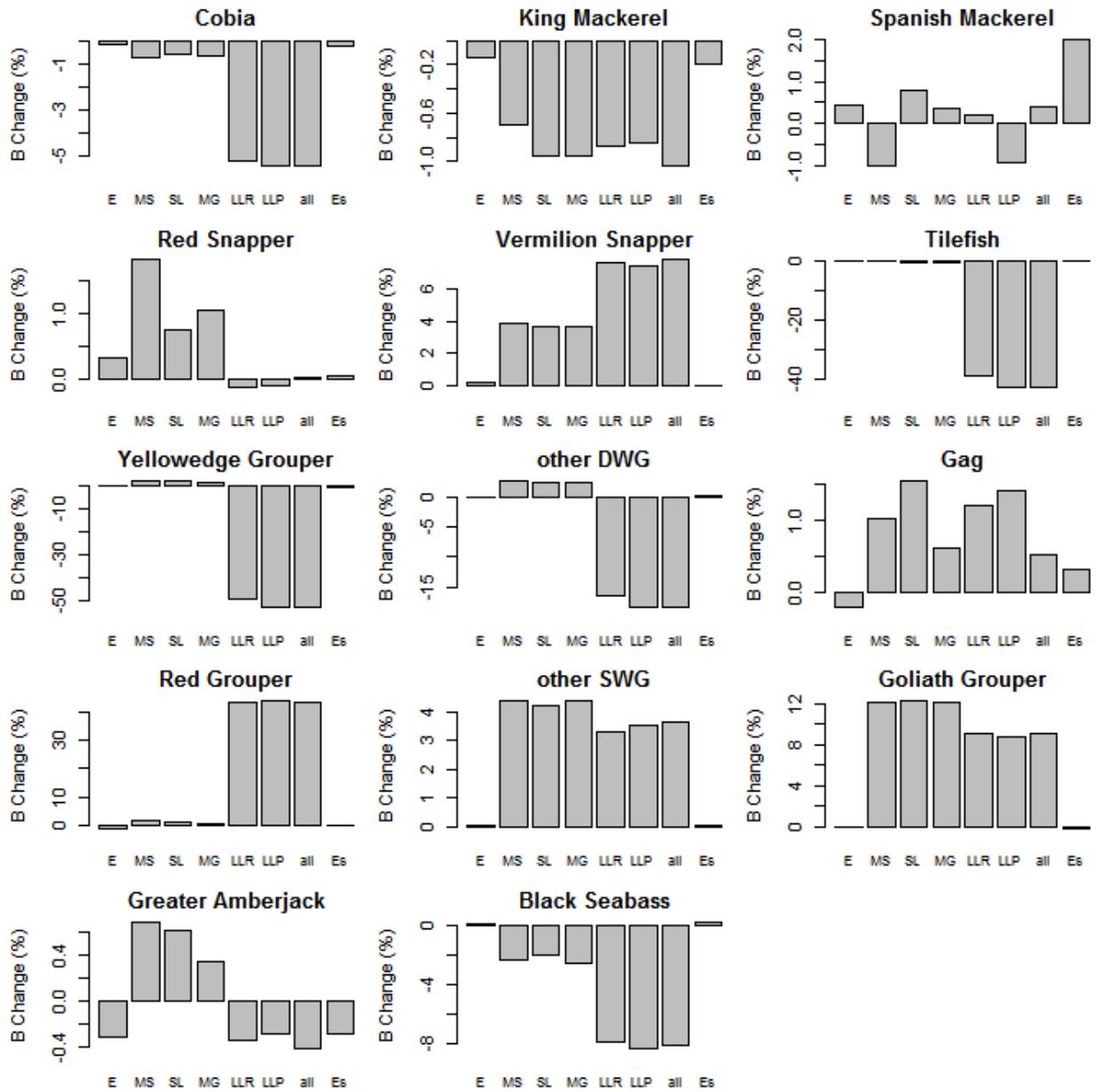


Figure 4-8. Percent biomass change predicted for current spatial regulations relative to a simulation with no MPAs. E = The Edges; MS = Madison-Swanson; SL = Steamboat Lumps; MG = Florida Middle Grounds; LLR = permanent longline restricted area; LLP = seasonal longline prohibited area; Es = The Edges with Gag seasonal egg production.

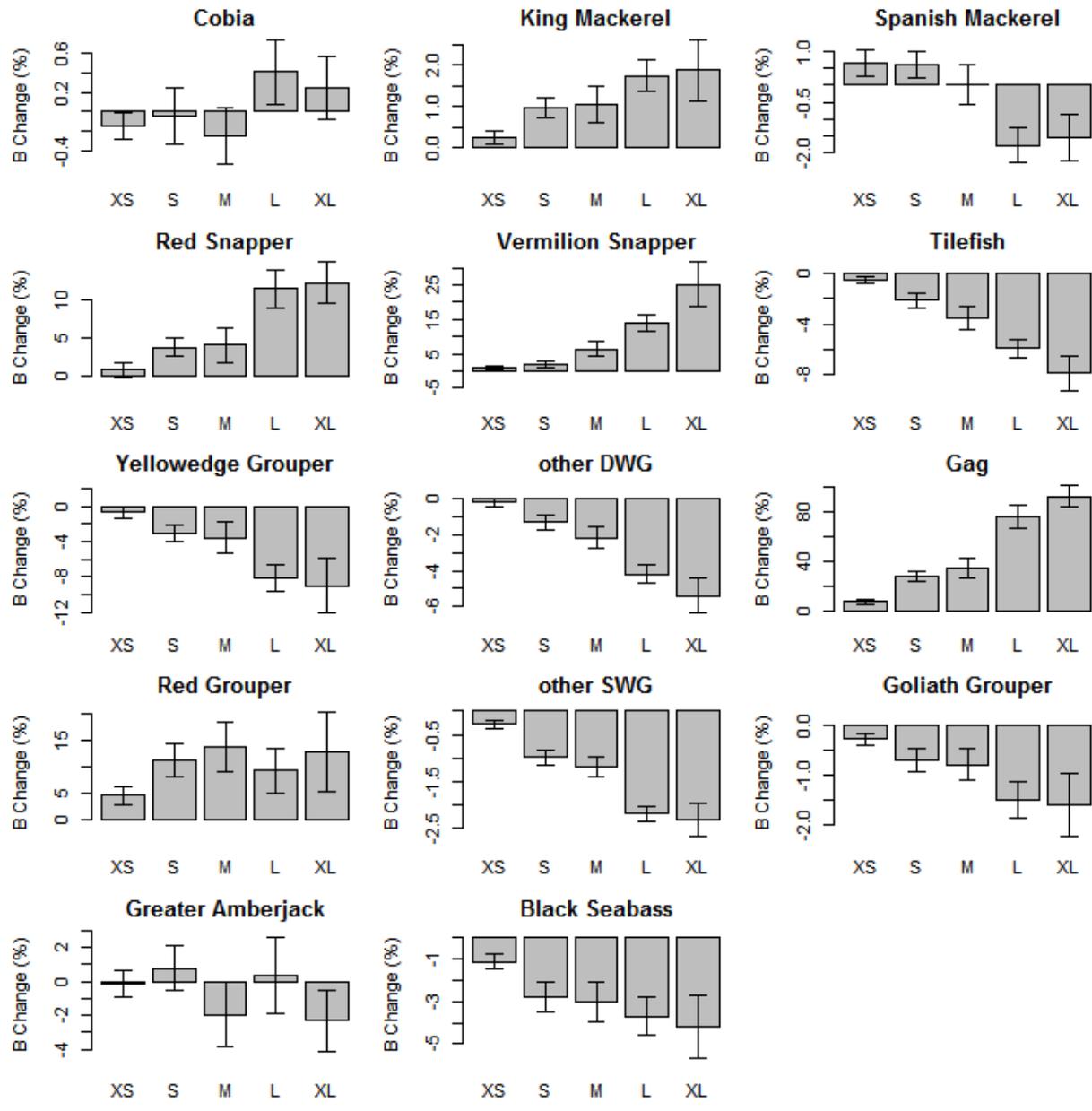


Figure 4-9. Percent biomass change predicted for randomly placed MPA blocks of various numbers and sizes selected by point pattern process. Bars represent the average biomass over 20 random configurations of each type and whiskers are the 95% confidence intervals.

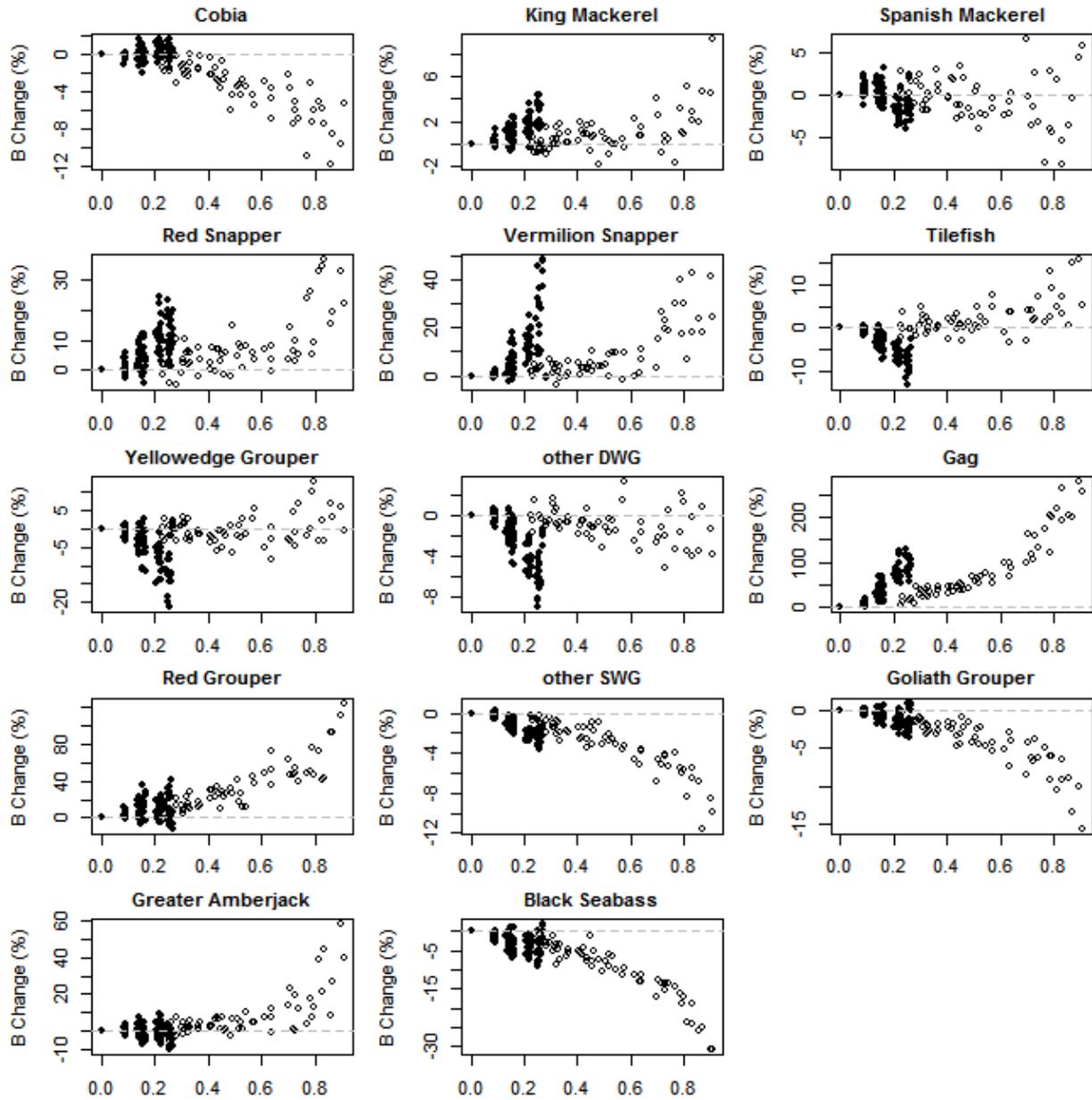


Figure 4-10. Percent biomass change predicted at various proportions of the total area that is closed to fishing (x-axis). Solid circles are random MPA block configurations and open circles are those where cells were closed based on simple random selection.

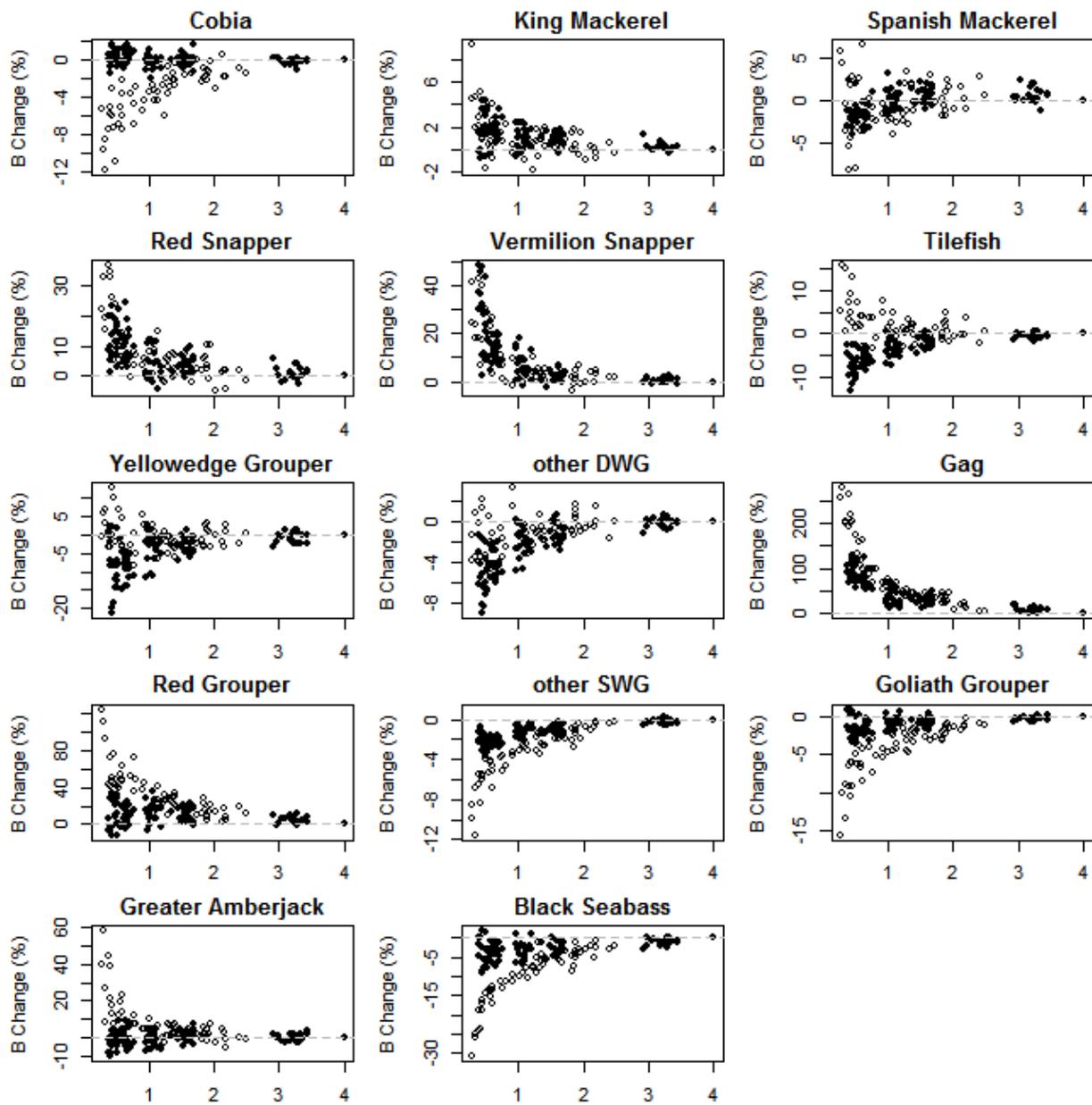


Figure 4-11. Percent biomass change predicted at different MPA edge ratios (x-axis). Solid circles are random MPA block configurations and open circles are those where cells were closed based on simple random selection.

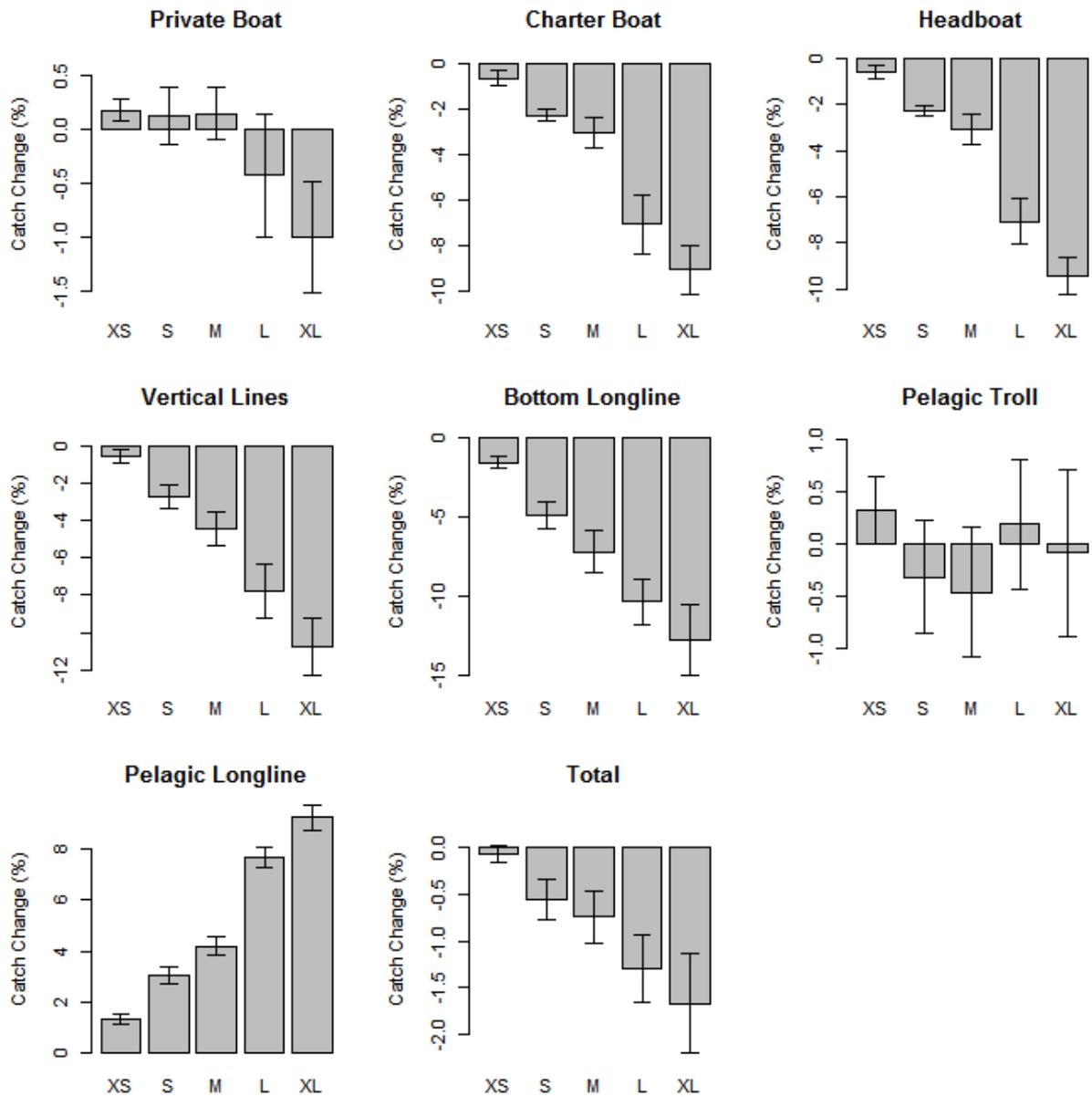


Figure 4-12. Percent change in catch predicted for randomly placed MPA blocks of various numbers and sizes selected by point pattern process. Bars represent the average biomass over 20 random configurations of each type and whiskers are the 95% confidence intervals.

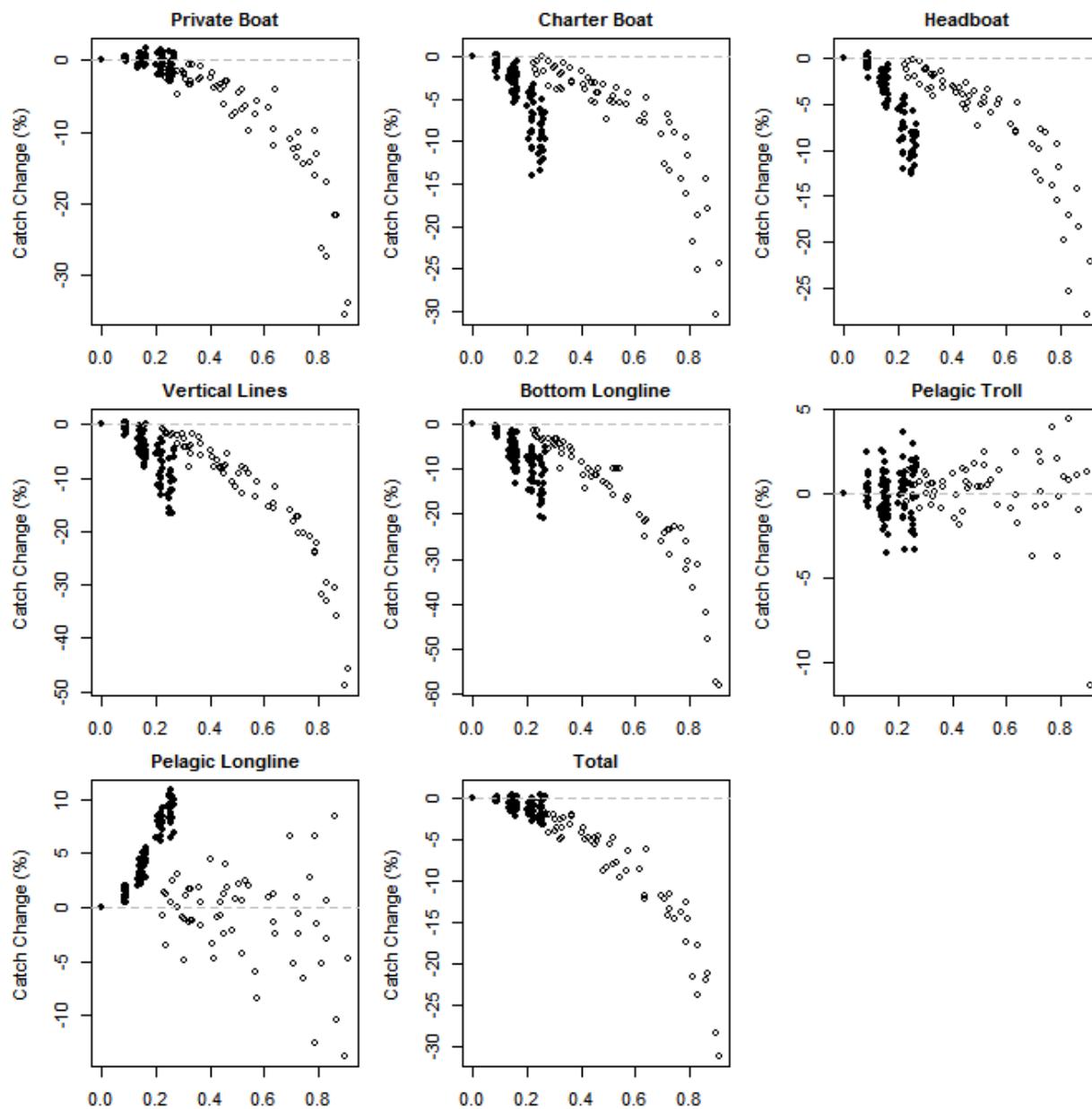


Figure 4-13. Percent change in catch predicted at various proportions of the total area that is closed to fishing (x-axis). Solid circles are random MPA block configurations and open circles are those where cells were closed based on simple random selection.

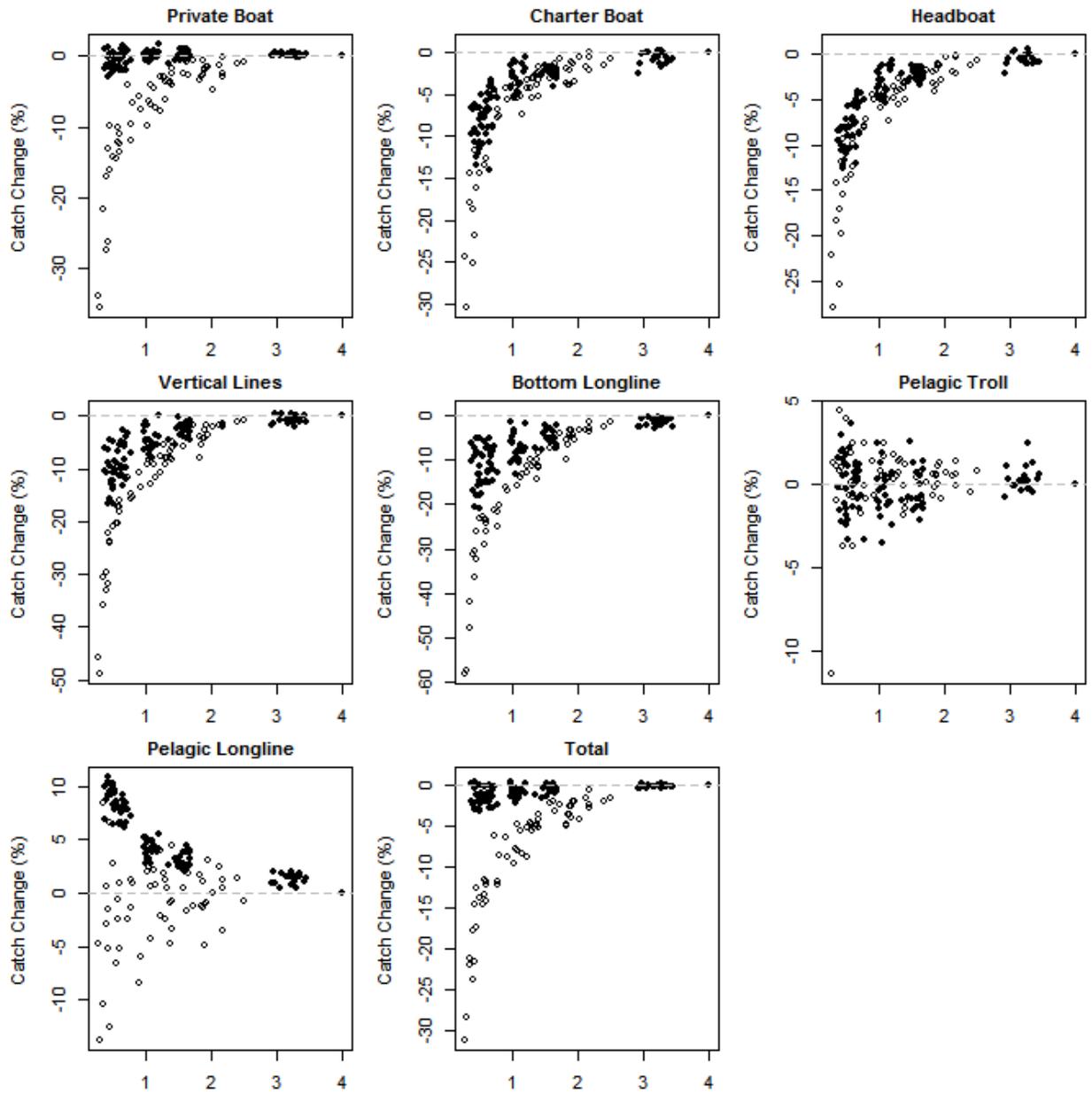


Figure 4-14. Percent change in catch predicted at different MPA edge ratios (x-axis). Solid circles are random MPA block configurations and open circles are those where cells were closed based on simple random selection.

CHAPTER 5 SYNTHESIS, LESSONS LEARNED, AND RECOMMENDATIONS

Summary of Important Findings

An overall conclusion from this research is that there are winners and losers in all policy options and that trophic interactions are important and should be taken into consideration by fishery managers. Predation and competition could potentially have a negative impact on several valuable commercial and recreational species, particularly Vermillion Snapper and Black Seabass, which were often predicted to be at a competitive disadvantage to other reef fishes. Because of competition, simultaneously rebuilding multiple predator species that share a common food source could require more conservative management action than what is determined from single species stock assessment models. The model did not predict catastrophic changes to the system due to any current or proposed policy options. On the other hand, climate change and its impact on primary production could lead to widespread ecosystem change and should be of utmost concern to natural resource managers.

I conclude from the policy optimization analysis that the forage base (e.g. baitfish and shrimp) on the WFS is healthy and could withstand moderate increases in fishing pressure (assuming there is demand for product) without severely impacting other fisheries. However, the level of exploitation on the forage base must be balanced with the ecological and economic impact on other species and fisheries. How other species and fisheries are valued, especially recreational fisheries, is critical to setting appropriate catch levels of forage groups. Lastly, while existing MPAs are generating some localized benefits to groupers in the form of improved age and sex structures and higher catch rates adjacent to reserves, they are not large enough to significantly

enhance overall stock condition on the WFS. Marine reserves that are intended to help rebuild overfished stocks should cover between 15-30% of the total area and have low edge-to-area ratios. Because of spillover, the predicted impact of such area closures on fisheries was only mildly negative, and in some cases provided long term benefits to the fisheries.

Best Practices for Model Development

The development of ecosystem models can be a daunting task and I offer several recommendations based on the success of this research project. These 'best practices' are applicable to any research project, but are particularly useful if constructing an ecosystem model with the intention of management application.

First, and most important, is to collaborate with a group of interdisciplinary researchers and scientists. The core group involved in this research consisted of fisheries scientists, stock assessment and ecosystem modelers, field ecologists, and economists. Each had extensive knowledge of at least some important components of the model while also offering insight gained from research conducted in marine and freshwater systems of the southeast U.S. and other parts of the world. At various stages of development, meetings should be held to review progress and ensure errors are not introduced into the model.

Second, develop clear objectives and outcomes of the project prior to constructing the model. Doing this is sure to lead to a more parsimonious model with fewer unnecessary variables and parameters, hence less potential for failure. Also, constructing the model around a set of defined issues prevents the creation of a model that is a 'jack of all trades, master of none', and essentially useless for management. In this project we were interested in evaluating policy on the WFS with respect to reef fish

fisheries and therefore did not attempt to model the entire Gulf of Mexico or even every component on the WFS. Rather, we identified some of the more contentious fishery management issues and built the model around them by including necessary predators, prey, and likely competitors as well as their associated fisheries.

Third, the model should be reviewed internally by members of the core workgroup and then by a larger group of scientists and stakeholders. Workshops are valuable when building ecosystem models because they bring together people that have more extensive knowledge about specific parts of the ecosystem than what is usually possessed by those building the model. In this case, a science workshop was held at the end of the first year when the model was nearly complete and only preliminary results were available. The purpose of the science workshop was to subject the model to critical review by a group of scientists that have a long history of research on the WFS and reconcile any major concerns about model structure and input values. A second workshop was held at the end of the project with an audience consisting of fishery stakeholders, state and federal management agencies, and non-governmental organizations. At the stakeholder workshop, we presented the results for review and discussed any issues and potential applications with the model. Regardless of the number, timing, agenda or audience, the workshops should aim to reduce model errors, build credibility, explain caveats, and expose the research to stakeholders and the broader scientific community.

Research Recommendations

Data collection programs on the WFS have been ongoing for some time now, however data gaps do remain. Here I list a few important data and modeling needs specific to this research project that would also improve any existing and future

ecosystem models in the Gulf of Mexico. Probably the most critical need is a better assessment of baitfish biomass and exploitation rates on the WFS. This group is important in the diets of almost all commercially and recreationally valuable species yet biomass estimates vary widely and the impact from harvesting is virtually unknown. Overestimating biomass in the model would reduce the impact of competitive interactions while underestimating baitfish biomass would exaggerate the trophic impacts of harvest policies. I have accounted for some of this uncertainty through Monte-Carol simulation in Ecosim but that approach is not possible in the policy optimization or ecospace analyses. An expansion of the Florida FWC baitfish trawl and acoustic survey and a thorough analysis of those data would be a first step towards improving our understanding of this important group.

The diet data for large offshore reef fish is outdated and biased by the use of baited gear, or simply doesn't exist for some species. The ongoing diet research being conducted by the Florida FWC relies heavily upon samples from inshore and nearshore areas that are mostly collected with trawls and seines which cannot be deployed in deepwater reef habitat. Hook and line sampling of fish from deep water does not produce quality stomach content data because those fish suffer from barotrauma and evert their stomachs when being reeled up from depth. A short term study to collect stomach samples using other methods such as spearfishing would provide the data needed to improve this model and future models in the region. However, efforts to procure funding for such research have so far been unsuccessful.

At both the science and stakeholder workshops, participants considered the model to be reasonable in regards to its ecology but expressed concern over how the

economics were represented, especially for the recreational sector. In Ecopath, the ex-vessel market price is entered per unit weight for each species in each fleet and profits are calculated after subtracting fixed and effort related costs. This is straightforward for commercial fisheries but presents a challenge for the recreational sector. To improve the EwE modeling approach, recreational fisheries must be treated differently. For example, 'market price' could be per individual fish and not per unit weight, value for catch-and-release species should be included, and 'profit' should be redefined. In Florida, the Gulf of Mexico, and Southeast U.S., the recreational sector is a very large component of the fishery and they must be represented appropriately if these models are to be used by managers.

Another improvement to the model would be a better representation of seasonal dynamics in Ecospace. For example, accounting for the monthly spatial patterns of primary production, water temperature, and salinity is expected to influence baitfish distribution and therefore drive migration patterns of pelagic species that often follow baitfish schools up and down the WFS. Additionally, juveniles of reef fish that are spawned during late winter and early spring would settle in estuaries during peak periods of system productivity and provide a better estimate of growth and natural mortality. This capability is currently under development and satellite imagery or oceanographic models will be used to generate spatial grids for the Ecospace model.

Management Application

The ultimate goal for this model, as well as most other ecosystem models, is for it to be used to inform management. The current management framework is not well suited to include ecosystem level information, but there are some ways in which these model could be applied under the existing single-species based fisheries management

process. Initially, ecosystem models need to be vetted through a formal review process, just as single species stock assessment undergo strenuous review. This could be accomplished as part of the Southeast Data Assessment and Review (SEDAR) process or through a series of workshops and peer review publications. In either case, the models should be held to a set of standards similar to those used in single-species stock assessment, but tailored to ecosystem models. Only after the models have passed some basic requirements will managers be comfortable using them to inform decision making.

There are several ways in which ecosystem models could be included in the single species stock assessment process. Ecosystem models can generate estimates of natural mortality and recruitment deviations that take into consideration bottom-up processes as well as predation and other mortality events. These vectors could then be used in stock assessment models such as stock synthesis 3 (SS3) and should include some uncertainty bounds with them. In a less formal application, ecosystem models could be used to quickly evaluate ecosystem-level hypotheses about a particular species and provide guidance for how to account for the dynamic implications of such hypotheses in a single-species approach. Projected future stock sizes made by the single species model should be compared with those predicted by ecosystem models. Ecosystem models have an advantage in this regard because they account for the trophic impacts of other species that are also changing over time as well. Projections made by ecosystem models are particularly useful when designing and testing the performance of concurrent rebuilding plans.

Ecosystem models should be integrated with the Scientific and Statistical Committees (SSC) and Advisory Panels (AP) to rank policy options and communicate ecosystem implications to council members. The SSCs and APs are tasked with evaluating policy options that relate to multiple species and impact multiple fisheries, yet they do so on a case by case basis. Ecosystem models provide the ability to simultaneously evaluate several policy options and to reveal the tradeoffs that they create. In the end, ecosystem models could add quantitative information to the fishery management plans, for example when drafting environmental impact statements. By adopting a more holistic and comprehensive approach to policy design and evaluation, management agencies will be better informed about the potential impacts of their choices. In summary, ecosystem models can be immediately integrated at several stages of the management process with little modification to the existing framework, which in turn would sharpen the focus of EBFM in the Gulf of Mexico.

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BIOGRAPHICAL SKETCH

David Chagaris is from New Bern, North Carolina and grew up playing sports and spending time outdoors, at the beach, and on the river. He graduated from New Bern High School and then received a Bachelor of Science in biology from East Carolina University in Greenville, NC (2002) where he was first introduced to marine ecology during a summer study abroad course in Belize, C.A.. The summer after completing his undergraduate degree, David gained valuable field experience as a field technician at the Bimini Biological Field Station, Bahamas tagging juvenile lemon sharks, collecting samples for a biodiversity study, spearfishing, and enjoying the island life. He then returned to ECU to begin working on his master's degree and each summer traveled to Belize to conduct his thesis research on the feeding habits and trophic dynamics of coral reef fishes from Calabash, Caye. In 2006, David finished his graduate work and moved to St. Petersburg, Florida to join the Florida FWC Fish and Wildlife Research Institute as a biological scientist in the Fisheries Independent Monitoring Program's gut lab. After about a year working under the microscope and in the field, he took a position with the stock assessment group and began focusing on trophic dynamic ecosystem modeling and stock assessments. David received his Ph.D. in fisheries and aquatic sciences from the University of Florida in December 2013, where he developed a food web model of the West Florida Shelf for ecosystem based fishery management in the Gulf of Mexico.