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Bioeconomic Consequences of Fisheries Management: Florida's Commercial Reef Fisheries

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UNIVERSITY OF MIAMI

BIOECONOMIC CONSEQUENCES OF FISHERIES MANAGEMENT: FLORIDA'S COMMERCIAL REEF FISHERIES

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Molly H. Stevens

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Over 84 reef fish species are commercially exploited by U.S. fisheries in waters surrounding Florida and require management actions that promote sustainable resources. The Florida reef ecosystem supports lucrative commercial and recreational industries that provide the state with billions of dollars annually. The goal of this study was to analyze the available demographic, fishery, and economic data for reef fishes, develop an age-structured bioeconomic production model, and assess the economic consequences of fisheries management relative to the commercial sector.

Key demographics data are required for the most basic assessment methodology, but just over half of these species had full parameter sets suitable for stock assessment, and less than a quarter had reliably estimated sets. Six species almost entirely landed in Florida of significant commercial value were analyzed: gag grouper, red grouper, black grouper, yellowtail snapper, mutton snapper, and gray snapper. Two of these species, gag grouper and yellowtail snapper, were used to test model robustness through multiple points of contrast: grouper *vs.* snapper, primarily recreational *vs.* commercial, and multiple *vs.* single gear fisheries. Total mortality rates estimated from the average length of the commercial catch were combined with demographics models to assess the populations within numerical cohort models. Both species' pop-

ulations were estimated to be overfished with fishing mortality rates, F, over double their natural mortality rates for all stocks in 2016. Yellowtail snapper spawning potential ratio was estimated at 26.94%, and Gulf of Mexico gag grouper was 12.25%.

Economic models were built linking commercial catch to revenues and nominal effort to costs. Ex-vessel prices were modeled for all six species within inverse demand functions that explained 53.1%–86.4% of the variability, where catch biomass was the primary explanatory variable. Fleets were defined for gag grouper and yellowtail snapper specified primarily by fishing area and gear. Revenues from 'jointly-caught' species were estimated for each 'fleet,' using fishing effort as the primary explanatory variable. Variable costs including fuel, bait, ice, tackle, groceries, and miscellaneous expenses displayed different functional forms within 'fleets.' Total monthly expenditures were predicted using average trip duration and vessel length. Functions estimating the variable costs explained 55.3%–75.1% of the variation in hook-and-line fleets, 26.5% in longline fleets, and 19.0%–35.9% in spearfishing fleets.

Numerical cohort model outputs served as inputs to economic models, creating a dynamic bioeconomic production model that was validated with observed revenue and cost data. Four management strategies were simulated in this model: (1) a baseline simulation of no change since 1998; (2) actual management regulations; (3) maximizing revenue under actual F and adjusting age at first capture, a_c; and (4) maximizing net revenue adjusting F and a_c. Across all simulations, the strategies designed to optimize economic profitability were also the most sustainable, allowing populations to rebuild and resulting in the highest SPR. Management measures that maximize commercial economic productivity would increase net benefits to the region while providing a more resilient ecosystem through healthier fish populations.

This work is dedicated to those who examine each question

in terms of what is ethically and aesthetically right,

as well as what is economically expedient.

(attrib. Aldo Leopold, 1949)

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CHAPTER 1

Introduction

Fishing in Florida has been a prominent aspect of life for millennia, and historical documentation details dramatic decreases of valuable reef fish resources in recent years (Walker, 1992; Roberts, 2007). In the first half of the 20th century, diaries and photographs documented a reef teeming with life and catches far exceeding anything observed today in terms of numbers and sizes, particularly in higher trophic level species rarely seen anymore (Beebe, 1928; Haas, 1952; Roberts, 2007; McClenacan, 2009). Following World War II, technological advances infiltrated the fishing industry, reduced search times and costs, and facilitated the capture of more fish in a shorter period of time (Hanna *et al.* , 2000; Olsen, 2008). New technologies such as depth sounding machines and GPS were especially effective in locating aggregations of transient spawners, a reproductive strategy observed in many grouper and snapper species where dense spawning aggregations are formed in the same locations year after year (Domeier & Colin, 1997; Farmer *et al.* , 2017). Predictable aggregation behavior made these species easy to locate and catch while also inhibiting reproductive success. By the late 1980s, documented Nassau grouper, Goliath grouper, and mutton snapper spawning aggregations were fished out of existence (GMFMC, 1993; Sadovy & Eklund, 1999). In addition to aggregate spawning, most groupers also display protogynous hermaphroditism, where all species begin as immature females, mature as females, then transition to males. Intense exploitation causes the population's length structure to truncate, resulting in heavily skewed sex ratios for hermaphroditic species. Gag grouper are protogynous hermaphrodites, and in the Gulf of Mexico their population decreased from 17% male in the 1970s to a mere 1% in the early 1990s, which likely further inhibited reproductive success (Collins *et al.* , 1987; Hood & Schlieder, 1992; Coleman *et al.* , 1996). The decimation of reef fish populations was fueled by their vulnerability to technological advances, and exponentially increasing human populations has resulted in continually rising demands for tourism and seafood, an overcapitalized commercial industry, and increased threats to reef fishes year after year (Ault *et al.* , 2009; FAO, 2016; BEBR, 2017).

Reef fish regulation in the United States began just four decades ago and encompasses dozens of teleost fish species landed for consumption in Florida (MSA, 1976). Groupers and snappers are the primary targets in this complex and therefore have the best available documentation of catch and demography. Due to the vast diversity on reef ecosystems, many species of reef fishes do not have key demographics data described, but these data are fundamental components of population dynamics assessment models (Ault *et al.* , 1998, 2014; Arnold & Heppell, 2015; Maunder & Piner, 2015). Ideally, data collection to describe lifetime growth takes place prior to exploitation, but that situation is very rare in most fisheries around the world. Under heavy exploitation, size distributions of fish truncate, as documented for gag grouper, and size at maturity can even decrease as fish are forced to put energy into reproduction instead of growth (Coleman *et al.* , 1996; Ault *et al.* , 1998; Harris *et al.*

, 2002; Ault *et al.* , 2008; McBride & Richardson, 2007). This phenomenon is particularly dangerous from a fisheries standpoint, because for any given age, a fish will be smaller, have less consumable biomass, and far fewer eggs to contribute to future cohorts due to the exponential relationship of weight with length (Richardson *et al.* , 2008). When sampling occurs after these effects have taken place, estimated parameters may not be representative of the true population dynamics. Even under the best conditions, sampling the entire range of a population requires time; often, estimates of maximum age increase the longer a fish is studied, as observed in black grouper and mutton snapper, where maximum age estimates more than doubled and quadrupled, respectively, since the 1980's (Claro, 1981; Mason & Manooch, 1985; Manooch & Mason, 1987; Crabtree & Bullock, 1998; Burton, 2002; SEDAR, 2008). Maximum age is considered the most reliable way to estimate natural mortality rates and translates directly to sustainable fishing mortality rates (Hoenig, 1983; Alagaraja, 1984; Ault *et al.* , 1998; Nadon & Ault, 2016). Doubling maximum age estimates results in halving natural mortality rate estimates and calls for halving fishing mortality rates. Overexploited stocks are more likely to result in the overestimation of natural mortality rates due to their truncated size/age distributions, which in turn results in recommendations of unsustainable fishing mortality rates. The implications of this scenario are wide-reaching, as overexploited stocks are more likely to unwittingly be further subjected to unsustainable levels of fishing mortality and less likely to recover.

Diminishing catch rates and substantial foreign fleets fishing in U.S. waters prompted Congress to pass the 'Fishery Conservation and Management Act of 1976,' subsequently known as the 'Magnuson-Stevens Act' (MSA, 1976). The MSA extended federal waters to 200 nautical miles offshore, known as Exclusive Economic Zones, and established eight Regional Fishery Management Councils (FMCs) to prepare, monitor, and define Fishery Management Plans (FMPs) (MSA, 1976). Federal waters encompassing coral reef habitat surrounding Florida (and extending southeast) are managed by the Gulf of Mexico (GMFMC), South Atlantic (SAFMC), and Caribbean (CFMC) Fishery Management Councils (GMFMC, 1981; SAFMC, 1983; CFMC, 1985). In 1981, GMFMC established the Reef Fish FMP that was "designed to rebuild declining reef fish stocks" (GMFMC, 1981). In 1983, SAFMC created the Snapper-Grouper FMP where it was recognized that out of the 69 species in the snapper-grouper complex, biological data was only available for 17 species, and 13 of these were "likely in a range of growth overfishing" (SAFMC, 1983). In 1985, CFMC formed the Caribbean Reef Fish FMP "to reverse the declining trend of the resource" and "reduce conflicts among users" (CFMC, 1985). Catch rates are indicative of population sizes, and if fishermen noticed reduced catch rates, it is likely the reef fish populations suffered a dramatic decrease during this time (Methot & Wetzel, 2013; Newman *et al.* , 2015). In 1989, Florida Marine Research Institute (later Florida Fish and Wildlife Conservation Commission, FWC) implemented recreational fishing licenses and commercial saltwater products licenses within state waters, extending 9 miles into the Gulf and 3 miles into the Atlantic, generating funding for resource managers but not limiting access (McRae, 2010). Florida represents a particularly complex region for marine resource management because many fish stocks are influenced by the regulations of two federal agencies, SAFMC and GMFMC, as well as the state agency, FWC.

SAFMC enacted the first regulatory actions to manage reef fish stocks in 1983, enforcing a 12 inch minimum size limit for black grouper, red grouper, yellowtail snapper, and red snapper (48FR39463; Figure 1.1). Minimum size limits reduce mortality rates on the population below the size limit, and these regulations were set with the goal of optimizing yield per recruit but lacked accurate life history information (SAFMC, 1983). In 1992, SAFMC increased/introduced a 20" size limit for most groupers, included more snappers in the 12" size limit, increased red snapper size limit to 20", introduced recreational bag limits, and prohibited the landing of Goliath and Nassau groupers (56FR29922). These groupers were nearly fished to extinction and listed as "Endangered" under the Endangered Species Act because of their susceptibility to overexploitation from previously noted spawning behavior and long life spans; both remain under moratorium today. In 1995, SAFMC increased mutton snapper minimum size limit to 16" and implemented a 12" minimum size limit for hogfish, and both remained unchanged for two decades despite scientists recommending stricter regulations throughout this time frame as updated life history data became available (Ault *et al.* , 1998; SEDAR, 2004). To prevent further overcapitalization of commercial reef fish fleets, the South Atlantic closed access to new entrants in 1998. This same year, FWC initiated an increase in the minimum size limit of gag and black groupers from 20 to 24 inches, decreased the recreational bag limit, and effectively closed the commercial fishery in Atlantic state waters during the spawning season, March–April, in an effort to rebuild the resource. Because the bulk of the South Atlantic grouper catch is landed in Florida, SAFMC enforced these regulations in federal waters as well. In 2010, the seasonal closure duration was doubled, extending from January–April to provide more protection for the spawning stock.

In 1990, GMFMC implemented size limits of 20" for most groupers, 12" for most snappers, recreational bag limits, and annual commercial grouper quotas (GMFMC, 1989). When the commercial quota was reached in 1990 resulting in a short closure of the fishery, GMFMC increased the shallow-water grouper commercial quota in 1992 by 1.6 million pounds to "prevent a closure during the fishing year, allowing a continuous supply of filets to the market," ignoring the biological intention of catch quotas (GMFMC, 1991). Under commercial catch quotas, the industry was forced to race to fish before the quota was filled, creating an unsafe work environment and suboptimal market conditions for the industry. While preventing these conditions is an important goal of fisheries management, the decision to increase the quota came at the cost of the biological sustainability of the resource. The ruling cited negligible biological impacts on dominant red grouper catch, but did not consider the impacts on black and gag groupers, rarer and more valuable species compared to red grouper. The commercial reef fishery in the Gulf of Mexico closed access to new entrants in 1992 (through a series of moratoriums on issuing new commercial permits, made permanent in 2005) to limit exploitation and protect the livelihoods of fishermen who depend on this resource (GMFMC Amendments 4,9,11,17,24). In an effort to protect reproductive success of reef fishes, GMFMC prohibited fishing during the spawning season of May–June at Riley's Hump in the Dry Tortugas, the only known spawning aggregation of mutton snapper in 1994 in U.S. Gulf waters (GMFMC, 1993). Riley's Hump was closed to fishing activity year-round by the National Park Service in 2001, and due to these measures, this aggregation still exists today. Complying with FWC regulations, grouper minimum size limits were increased to 24 inches in 2000, and the commercial sale of gag and black groupers was prohibited from February 15–March 15 (GMFMC, 1999). The commercial grouper closure was enforced until 2010, when the fishery opened year-round and shifted to privilege-based fishing.

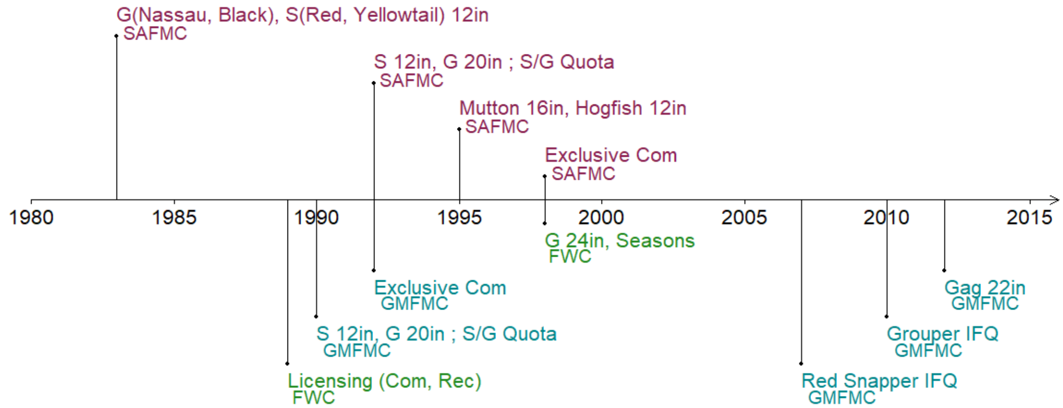

Figure 1.1: Timeline of regulations enacted by SAFMC (red), GMFMC (blue), and FWC (green) included minimum size limits, seasonal closures, bag limits, and clearly defined fishing privileges. Size limits are reported in inches (in) for grouper (G) and snapper (S) species. Major impacts to the commercial (com) and recreational (rec) fishery are noted.

The Gulf of Mexico implemented Individual Fishing Quotas (IFQs), a privilege-based regulatory strategy that allocates a set portion of the quota to individual fishermen, for red snapper in 2007 and grouper & tilefish in 2010. Management strategies that appropriately define property rights, e.g. IFQs, promote sustainability and increase economic benefits (Kellner *et al.* , 2011; Sanchirico & Springborn, 2011; Solis *et al.* , 2014; Cunningham *et al.* , 2016). Globally, it has been estimated that fisheries management reforms could increase annual global benefits by $53 billion (Costello *et al.* , 2016). Natural resources without appropriate management controls or defined property rights, e.g. open access resources, often result in overcapitalized resources and dissipated rent (Hoshino *et al.* , 2018). Currently, the Florida finfish fishery generates over $75 million in annual dockside value, and reef fishes constitute over half of that total. This complex of species contributes to economic production, creation of jobs, provision of food, and maintenance of a healthy ecosystem, which has many biological, aesthetic, and cultural values. The fishes sold within the seafood market create jobs and revenue at the dealer level, which then feed into the restaurant and tourism industries. Historically, U.S. reef fish management has focused on maintaining current catch levels to not disrupt the industries built around this natural resource. Given the exploitation history and overcapitalization of this fishery, it is hypothesized that stricter regulations would be beneficial for the industry rather than detrimental.

Two decades ago, Ault *et al.* (1998) estimated 22 out of 35 species in the snapper-grouper complex to be overfished in south Florida and recommended drastic rebuilding policies. With maintenance of healthy reef fish stocks, the entire reef ecosystem is more resilient to threats including climate change, invasive species, and pollution, creating a more reliable source of income to the state. Furthermore, sustainably managed fish stocks foster a more stable job environment for fishermen through increased long-term profits and reduction of fishery collapse risk. Despite these proposed benefits, short-term economic benefits to society are often chosen over longer-term investments in natural resources, resulting in depleted fish stocks and degraded reef habitats from intense fishing pressures, coastal development, and climate changes (Jackson *et al.* , 2001; Ault *et al.* , 1999, 2014; Seitz *et al.* , 2014; McCauley *et al.* , 2015). The "open-access" nature of the reef ecosystem in Florida has resulted in undervaluation of conservation when making decisions for management (Brander *et al.* , 2007). Conservation of marine resources is typically viewed in direct conflict with economic goals, but it is hypothesized the underlying strategies that would optimize these values are inherently intertwined. Florida has a multi-billion dollar tourism industry, much of which is centered around ocean adventures and seafood (Johns *et al.* , 2001, 2014; Andrews *et al.* , 2005; Ault *et al.* , 2005b). Out of all recreational activities on Florida's reefs, fishing comprised over half the person-days in southeast Florida, surpassing scuba diving and snorkeling (Johns *et al.* , 2001). Regulation of recreational and commercial reef fishing practices directly affects the health of the reef ecosystem and all surrounding industries through gear impacts and reef fish population health (Bohnsack & Ault, 1996). Early monitoring of the U.S. fishing industry focused on commercial catch and profitability, but economic data has been notably absent within the SouthEast Division of Assessment and Review (SEDAR) framework, the current standard for monitoring and regulating reef fishes since 2002 (Radcliffe, 1919; SEDAR, 2002).

The goal of this study was to analyze the available demographic, fishery, and economic data for reef fishes, develop an age-structured bioeconomic production model, and assess the economic consequences of fisheries management relative to sustainability of the commercial sector. The "data-limited" situation in Florida was elevated by investigating the availability and validity of life history parameters to guide efforts to assess the sustainability of all exploited reef fishes (Chapter 2). Commercial fishery and economic data were assimilated, and functions were built estimating revenue and cost from catch and effort, respectively, for commercial reef fish fleets (Chapter 3). Average length of the exploited phase was utilized alongside life history functions to estimate mortality rates of reef fishes in Florida. These biological and economic models parameterized a bioeconomic numerical cohort model that was validated with observed catch, revenue, and cost data (Chapter 4). Using this model, management strategies that strove to optimize bioeconomic sustainability were identified then simulated with implementation beginning in 1998 (Chapter 5). These analyses defined economically favorable management strategies, outlined their associated biological characteristics, and estimated net present value of each strategy throughout the duration of the simulations (1998-2016). Managing Florida commercial reef fisheries considering biological sustainability and economic productivity promotes increasing ecosystem health and net benefits to the region.

CHAPTER 2

Data Assimilation: Demographics and Fisheries

Demographics and fisheries data parameterize population dynamics models, and available sources were assimilated for assessment (Figure 2.1).

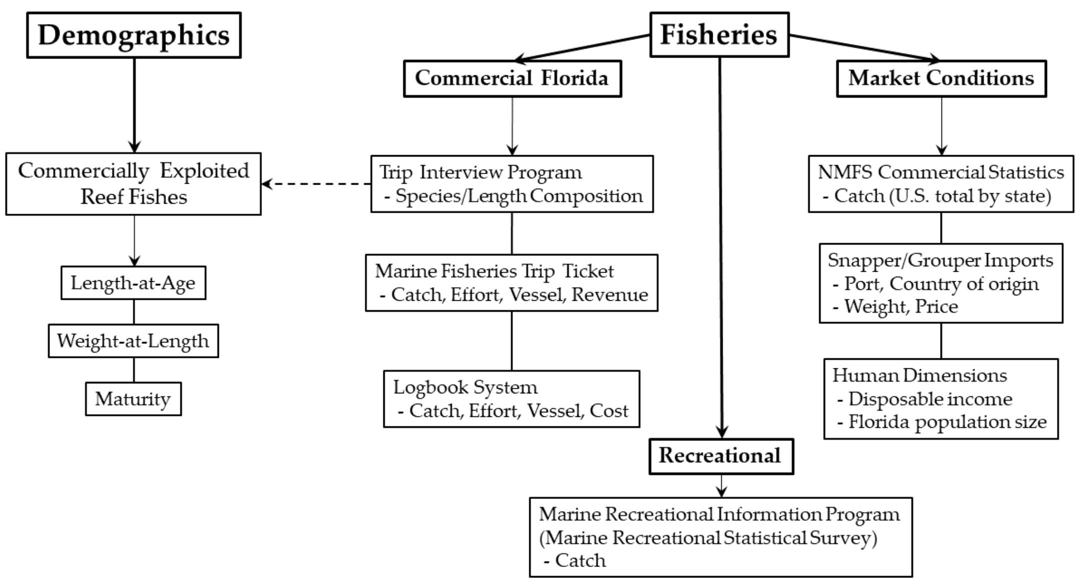

Figure 2.1: Demographics data defining lifetime growth are used to parameterize population dynamics assessment models, and fisheries data are used to estimate the current status of the resource. The data sources listed above were assimilated to facilitate a bioeconomic assessment of commercially important reef fishes.

2.1 Methods

2.1.1 Florida Reef Fisheries Data Assimilation

Commercial fishery data in Florida were collected through three major programs: Trip Interview Program (TIP), Marine Fisheries Trip Ticket (MFTT), and Florida Logbook System (FLS). The TIP is a dockside intercept statistical survey of the commercial fleet that began in the 1980s and was used to estimate mortality rates. The MFTT are bills of sale between fishermen and dealers and were used to estimate revenue. The FLS was filled out by the captain, where a subset was sampled for variable costs, allowing for cost estimation at the trip level. In conjunction, these datasets were used to estimate bioeconomic sustainability of Florida reef fishes, but an individual trip could not be identified across all three datasets. Therefore, when processing these datasets, care was taken to ensure that the same representative subset of vessels was being defined across all three programs (Table 2.1). The recreational landings of these species were calculated using the Marine Recreational Information Program (MRIP), formerly Marine Recreational Fisheries Statistics Survey (MRFSS). Market conditions were defined using data from the Bureau of Economic and Business Research and the U.S. Bureau of Economic Analysis.

The MFTT collected data on landings, ex-vessel price, effort, revenue, gear, and area within United States waters since 1984. The program has undergone sampling redesigns throughout this time frame, with the most recent in 1995. Vessel characteristics, which influence costs, were recorded by the Atlantic Coastal Cooperative Statistics Program (ACCSP) and were linked to the MFTT starting in 2007, when unique vessel identifications were available in the MFTT. The FLS was implemented in 1990 and included information on catch, effort, gear, area, etc. A subset of the FLS was sampled for variable costs including bait, ice, fuel, groceries, and tackle since 2001, and underwent its last major sampling design change in 2014. The FLS had unique vessel identification information throughout the years used in these analyses, 2007–2016.

'Commercially exploited' reef fish species were identified through species-specific length composition data from the TIP. These data were evaluated for the time period 1984 – 2016 for two geographical regions, Florida and the U.S. Caribbean (Puerto Rico and the U.S. Virgin Islands). Reef fishes were refined to 'principal' Florida reef fishes, distinguished by high value and high Florida landings volume relative to U.S. landings (NOAA's National Marine Fisheries Service Annual Commercial Landings Statistics). Reef fishes whose distributions centered around Florida and contributed to a bulk of the commercial landings represented cohesive units that could be used to estimate biological and economic dynamics of the commercial reef fisheries with Florida datasets.

2.1.2 Exploited Reef Fish Demographics Synthesis

The following life history synthesis was designed to obtain reliable demographic parameters for 'commercially exploited' reef fishes describing lifetime growth, survivorship, and reproductive maturity required for size-age cohort-structured stock assessments (Table 2.2; c.f., Ault *et al.* 1998; Quinn & Deriso 1999).

Table 2.1: Datasets assimilated for a bioeconomic assessment of commercially important reef fishes. “Availalble” years were when the dataset was initiated, “reliable” years for a specific variables were when it appeared to be recorded consistently, and “used” years were the portion of the dataset utilized in these analyses.

Database	**Acronym**	**Variable**	**Available**	**Reliable**	**Used**
Fisheries					
Trip Interview Program	TIP	Length Samples	1984–2016	1995–2016	1995–2016
Marine Fisheries Trip Ticket	MFTT	Landings	1984–2016	1992–2016	1995–2016
Marine Fisheries Trip Ticket	MFTT	Fishing Effort	1984–2016	1995–2016	1995–2016
Marine Fisheries Trip Ticket	MFTT	Vessel Details	1984–2016	2007–2016	2007–2016
Atlantic Coastal Cooperative Statistics	ACCSP	Vessel Details	–	–	2007–2016
NMFS Southeast Regional Office	SERO	Permit Details	1993–2016	1998–2016	2007–2016
Florida Logbook System	FLS	Landings	1990–2016	1990–2016	2007–2016
Florida Logbook System	FLS	Fishing Effort	1990–2016	1990–2016	2007–2016
Florida Logbook System	FLS	Variable Costs	1990–2016	2014–2016	2014–2016
Marine Recreational Fisheries Statistical	MRFSS	Landings	1979–2006	1986–2006	1995–2003
Marine Recreational Information Program	MRIP	Landings	2004–2016	2004–2016	2004–2016
NMFS Commercial Landings Statistics	–	Landings	1992–2016	1992–2016	1995–2016
Market					
NMFS Commercial Fishery Statistics	–	Reef Fish Imports	1990–2016	1992–2016	1995–2016
Bureau of Economic & Business Research	BEBR	Florida Population	1972–2016	1972–2016	1995–2016
U.S. Bureau of Economic Analysis	BEA	GDP Price Deflator	1929–2016	1947–2016	1995–2016
U.S. Bureau of Economic Analysis	BEA	U.S. Disposable Income	1929–2016	1947–2016	1995–2016

Lifetime growth described by von Bertalanffy length dependent on age $L(a)$ growth function,

$$L(a) = L_\infty \cdot [1 - e^{-K \cdot (a - a_0)}] \tag{2.1}$$

where L_∞ is asymptotic length, K is the Brody growth coefficient, and a_0 is theoretical age at length zero. Observed maximum age a_λ was used to estimate the mean length at oldest age L_λ from Equation (2.1). The allometric weight (W) dependent on length relationship,

$$W(a) = \alpha \cdot L(a)^\beta \tag{2.2}$$

has model-fitting parameters α and β. Equations (2.1) and (2.2) are used in conjunction to model lifetime growth of an individual fish in terms of average weight at age. Length at reproductive maturity L_m was described using the logistic function,

$$p(L) = \frac{e^{\beta_0 + \beta_1 \cdot L}}{1 + e^{\beta_0 + \beta_1 \cdot L}} \tag{2.3}$$

where $p(L)$ is the proportion of fish mature at length L, and β_0 and β_0 are model-fitting parameters (Roa *et al.* , 1999; Kutner *et al.* , 2004). The parameter L_m was defined as the associated length at $p(L) = 0.5$, i.e., the length at which 50% of individuals have attained sexual maturity. The corresponding age at 50% maturity, a_m, was computed from L_m using the von Bertalanffy growth function (Equation 2.1) rearranged to compute age as a function of length.

$$a = \frac{-ln\left[1 - \left(\frac{L(a)}{L_\infty}\right)\right]}{K} + a_0 \tag{2.4}$$

An extensive literature review was conducted for the list of commercially-exploited reef fish species in Florida and the U.S. Caribbean to assess life history parameter

Table 2.2: Target life history parameters and additional parameters compiled for exploited reef fishes in this Chapter.

Parameter	Definition	Units	Equation
a	Age	years	2.1
a_λ	Maximum observed age	years	2.1
$L(a)$	Length at age a	mm FL	2.1
K	Brody's growth coefficient	per year	2.1
a_0	Theoretical age at length 0	years	2.1
L_∞	Asymptotic length	mm FL	2.1
L_λ	Length at maximum age	mm FL	2.1
L_{min}	Minimum length sampled	mm FL	
L_{max}	Maximum length sampled	mm FL	
$W(a)$	Weight at age a	kg	2.2
α	Weight-length scalar	kg·mm$^{-\beta}$	2.2
β	Weight-length power	unitless	2.2
L_m	Length at 50% maturity	mm FL	2.3
a_m	Age at 50% maturity	years	2.4
L_d	Desired length units	mm	2.6
W_d	Desired weight units	kg	2.7
L_1	Original length units	cm, in, etc.	2.6
W_1	Original weight units	g, lb, etc.	2.7
u	Length conversion factor for α	L_d/L_1	2.6
v	Weight conversion factor for α	W_d/W_1	2.7
α_1	Original weight-length scalar	$(W_1)\cdot(L_1)^{-\beta}$	2.8
L_{99}	99^{th} percentile of commercial lengths	mm FL	
L_c	Length at first capture	mm FL	
n	Sample size	numbers of fish	

availability and reliability. The review encompassed peer-reviewed publications, dissertations and theses, conference proceedings, published and unpublished technical reports, etc. Detailed information for the various life history parameters from each literature reference were compiled into a synthesis database (Table 2.3). This information included aspects of the study design (location, time frame, sampling gear), age- and length-range of sampled individual fish, sample size, biological and statistical methods, etc. These study characteristics were used to develop hierarchical selection criteria for identifying the best available literature references and associated parameters for length-age, weight-length, and maturity for each species (Figure 2.2). For example, studies conducted in the tropical Western Atlantic region were preferred over studies conducted in the temperate Western Atlantic (hierarchy level 1). For multiple studies within the tropical region reporting von Bertalanffy parameters for the same species, preference was given to length-age models (Equation 2.1) developed from sectioned otoliths for individual fish and fit with nonlinear regression over other biological and statistical methods (hierarchy level 2). Likewise, for maturity preference was given to studies that employed histological examination of gonads and logistic regression (Equation 2.3). For competing weight-length functions (Equation 2.2), preference was given to studies using model-fitting procedures that resulted in homogeneous variance of residual errors for weight along the range of lengths (i.e., the property of homoscedasticity; Kutner *et al.* 2004). If competing studies were similar with respect to level 2 criteria, then level 3 criteria were considered, and so forth.

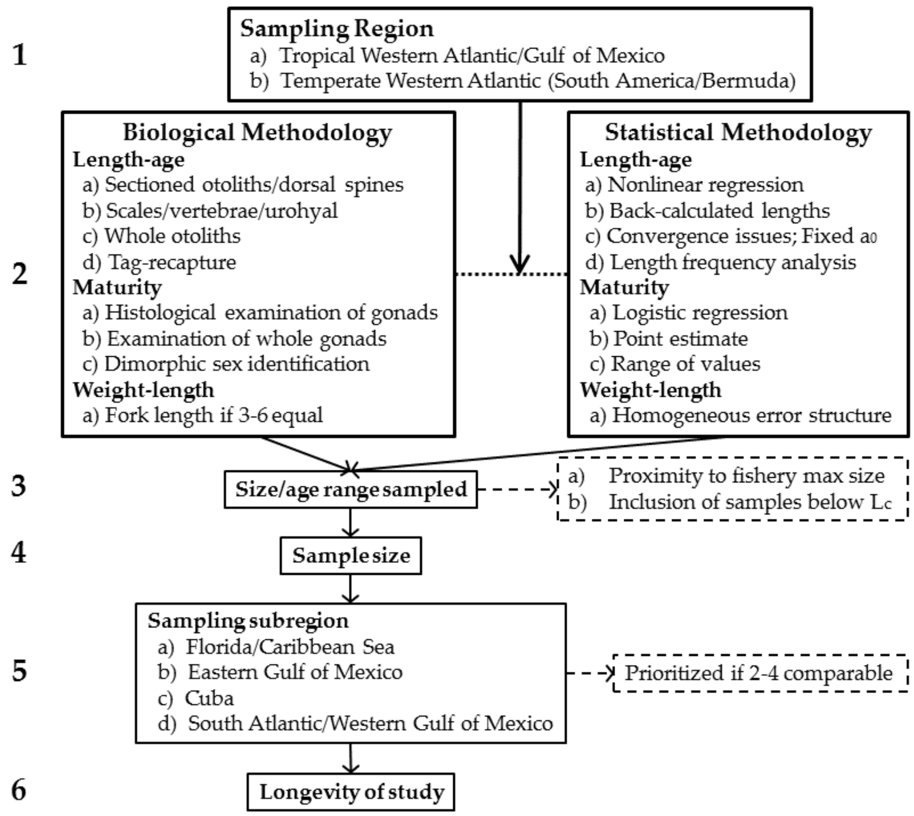

Figure 2.2: Hierarchical selection criteria for determining a best set of life history parameters from the available scientific literature.

The TIP length observations were used to estimate the expected maximum length for each species and represented a final criteria of reliability for length-age studies. Although obvious outliers were removed during the analysis process, it was usually not possible to determine whether extremely large length observations were errors in the database or were true values. Furthermore, the concept of expected maximum length is an average value with some variation of observations above and below. The following criteria were developed to calculate the 99^{th} percentile of length observations (L_{99}) as a measure for expected maximum length for three different ranges of sample sizes:

$$L_{99} = \begin{cases} 99.95^{th} \text{ percentile} & \text{if } n \geqslant 10,000 \\ 99.90^{th} \text{ percentile} & \text{if } 10,000 > n \geqslant 2000 \\ 99.50^{th} \text{ percentile} & \text{if } n < 2000 \end{cases} \tag{2.5}$$

In each case, the defined L_{99} represents the upper end of the TIP length distribution while guarding against potential outliers. The criteria of Equation (2.5) were used to estimate L_{99} for all commercially exploited species.

For this life history synthesis, units of length and weight for demographic functions and parameters were millimeters fork length (FL) and kilograms wet weight (W), respectively. Unit of measure conversions for length (e.g., inches to millimeters) and weight (e.g., pounds to kilograms) were carried out using

$$L_d = u \cdot L_1 \tag{2.6}$$

$$W_d = v \cdot W_1 \tag{2.7}$$

where subscript d denotes the desired unit of measure, subscript 1 denotes the original unit of measure, u is the length conversion factor, and v is the weight conversion factor. Equation (2.6) was applied to L_∞ to convert length-age functions to millimeters; parameters a_0 and K are independent of the unit of measure for length. Length parameters L_λ and L_m were also converted to millimeters using Equation (2.6). For the allometric weight-length model, parameter β is independent of the unit of measure for length, but parameter α is dependent on the unit of measure for both length and weight. Conversions of parameter α to millimeters and kilograms were carried out using the general formula

$$\alpha = v \cdot u^{-\beta} \cdot \alpha_1 \tag{2.8}$$

Table 2.3: Tables and variables comprising the life history parameter synthesis database for reef fishes in Florida and U.S. Caribbean.

Data Table	**Variable**
Reference Details	Short reference
	Publication type
	Full reference
Reference Data Inventory	Common family
	Common name
	Genus
	Species
	Short reference
	Publication type
	Data type
	LengthAge
	WeightLength
	Maturity
	LengthLength
Study design variables for parameter data tables	Publication year
	Short reference
	Common family
	Common name
	Scientific name
	Sampling location
	Sampling timeframe
	Sampling frequency
	Sampling gear
	Sample size
	Sex
LengthAge	Number aged
	L_∞ and SE L_∞
	K and SE K
	a_0 and SE a_0
	Fitting method
	r^2
	Age range
	Length units
	Length type
	Length range
	Aging method
	Type of hard part
	Whole/sectioned
	Age validation
LengthAge *continued*	Back-calculation (BC) method
	BC sample size
	BC age range
	BC length range
	BC equation form
	BC equation units
	BC model parameters
	BC r^2
WeightLength	Equation form
	a and SE a
	b and SE b
	r^2
	Length units
	Length type
	Weight units
	Weight type
	Length range
	Weight range
Maturity	a_{50} and SE a_{50}
	L_{50} and SE L_{50}
	a_m reported
	L_m reported
	L_{100} reported
	Length units
	Length type
	Length range
	Mean length mature
	Mature length range
	Age range
	Mature age range
	Sex change
	Sex determination method
	Months w/ ripe females
LengthLength	Equation form
	Length units
	$b0$ and SE $b0$
	$b1$ and SE $b1$
	r^2
	Length range

using the definitions from Equations (2.6) and (2.7). If either length or weight was already in the desired unit of measure, the respective conversion factor was set to 1 in Equation (2.8).

For life history parameters where length type was different from fork length (e.g., total length, standard length), length-length conversion equations were included as part of the literature search and parameter synthesis database (Tables 2.2 and 2.3). For length-age, L_∞ was reported in the original length type, and a length-length conversion equation was provided where possible. In contrast to unit of measure conversions, parameters a_0 and K of the length-age function are not independent of length type. The same procedure was applied for parameter α for weight-length functions. For modeling purposes, the length-length equations can be used in conjunction with length-at-age or weight-length functions to provide the respective curves in fork length. Values for point estimates of length, L_λ and L_m, were converted to fork length using a length-length equation where possible.

2.1.3 Florida Commercial Reef Fishery Fleets

NMFS Southeast Regional Office recorded federal permitting information, including renewal and expiration dates, for vessels licensed to target reef fishes. Florida reef fishes have been managed within closed-access commercial fisheries since 1992 in the Gulf of Mexico and since 1998 in the South Atlantic. Closed-access fisheries do not allow new entrants into the fishery without one or more current participants exiting the fishery. Gulf of Mexico (GM) Reef Fish FMP originally operated on a quota system then transitioned to an Individual Fishing Quota (IFQ) system for Red Snapper in 2007 and Grouper & Tilefish in 2010. South Atlantic Snapper-Grouper

FMP licensed the commercial fishery with either an Unlimited Trip Limit (SA1) or 225lb Trip Limit (SA2). Federal permitting information (GM, SA1, SA2) was linked to FLS and MFTT datasets using unique vessel identification numbers.

A vessel that paid to renew these permits in a given year, landed the principal species, and fished using a primary reef fish gear was included in further analyses, and unique combinations of these attributes defined fishing 'fleets.' Including all trips operated by vessels within each fleet allowed for estimation of total revenue and cost incurred by the fleets while also accounting for trips where the vessels targeted a 'principal' species, but did not land the 'principal' species. Florida commercial fishery datasets (MFTT, FLS) were subsetted based on these fleet definitions to analyze the bioeconomic dynamics of commercial reef fisheries. Fleet validation statistics were utilized to validate equivalent 'fleet' subsets between MFTT and FLS data for the time periods 2007–2016 and 2014–2016. Validation statistics included gear-specific catch compositions, vessel length distributions, proportion of 'principal' species to total landings, and proportion of trips successfully landing the 'principal' species for each fleet.

2.1.4 Commercial Reef Fish Market Conditions

Market conditions hypothesized to influence prices and costs of the commercial reef fish fleets were compiled. Disposable income and Florida human population size data were obtained from the Bureau of Economic Analysis and University of Florida's Bureau of Economic & Business Research, respectively, to investigate potential influences on ex-vessel prices fishermen receive at the dock. As the U.S. disposable income increases, the population has more money to spend resulting in increased consumption. Demand could increase with the number of Florida residents, driving prices up. Imports often influence demand in the U.S. because they represent substitute goods to domestic products. Grouper and snapper imports into Miami and Tampa have been summarized by U.S. Customs District since approximately July 1990 and were obtained via the National Marine Fisheries Service. Import country of origin, species type (grouper/snapper), catch volume, and price were compiled monthly. Contribution of snapper and grouper imports to the total available biomass in the Florida market was calculated from 1995-2016, and top importing countries throughout the time period were identified.

All temporal price data were intertwined with economic processes and influenced by inflation, requiring standardization to compare monetary values through time. All price data were standardized using the gross domestic product (GDP) implicit price deflator, the current standard for National Marine Fisheries Service (NMFS) economists. The GDP deflator is a measure of the level of prices of all new, domestically produced, final goods and services in an economy. Prices from any month t, P_t, were standardized by creating a ratio between the GDP deflator in the current month, GDP_t, and the GDP deflator from the most recent month, $GDP_{t=2016}$,

$$P_{t=2016} = P_t \cdot \left(\frac{GDP_{t=2016}}{GDP_t} \right) \tag{2.9}$$

converting all price data to 2016 dollars, $P_{t=2016}$, which allowed for direct comparison of all monetary data through time. Prices could have been standardized to any year, but all price data presented here were converted to 2016 dollars.

2.2 Results

2.2.1 Reef Fish Lifetime Growth Parameterization

For the period 1984-2016, the Trip Interview Program sampled commercial catches during all seasons covering the entire respective coastlines of Florida and the U.S. Caribbean each year. The subset of commercially exploited reef fishes was defined by a frequency of 300 or more TIP length samples for a given species from either Florida or the U.S. Caribbean, or a minimum of approximately 10 samples per year on average (Table 2.4). Sample size exceptions were made for three historically important groupers (Table 2.4), two of which have been under fishing moratoria since the early 1990's (Goliath, Nassau; Warsaw). The final list was comprised of 84 reef-fish species from 12 families: groupers (Epinephelidae), snappers (Lutjanidae), grunts (Haemulidae), porgies (Sparidae), triggerfishes (Balistidae), wrasses and parrotfishes (Labridae), barracudas (Sphyraenidae), surgeonfishes (Acanthuridae), squirrelfishes (Holocentridae), goatfishes (Mullidae), boxfishes (Ostraciidae), and bigeyes (Priacanthidae).

Table 2.4: Commercially exploited Florida and U.S. Caribbean reef fishes, with sample sizes (n) of lengths collected by the commercial Trip Interview Program, 1984–2016. The 99^{th} percentile of length distributions, L_{99}, was calculated for sample sizes $\geq$ 300, i.e., a minimum of approximately 10 samples per year on average. Sample size exceptions were made for three important grouper species (Goliath, Nassau, Warsaw).

Common	**Scientific**	**Florida**		**Caribbean**	
Name	Name	n	L_{99}	n	L_{99}
Grouper	**Epinephelidae**				
Atlantic Creolefish	*Paranthias furcifer*	469	378	36	–
Black Grouper	*Mycteroperca bonaci*	7,545	1390	231	–
Coney	*Cephalopholis fulva*	51	–	34,175	639
Gag Grouper	*Mycteroperca microlepis*	91,980	1268	4	–
Goliath Grouper	*Epinephelus itajara*	2	1733	78	2007

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Common Name	**Scientific** Name	**Florida** n	L_{99}	**Caribbean** n	L_{99}
Graysby	*Cephalopholis cruentata*	536	425	2,404	550
Misty Grouper	*Hyporthodus mystacinus*	115	1208	294	1370
Mutton Hamlet	*Alphestes afer*	–	–	309	440
Nassau Grouper	*Epinephelus striatus*	36	893	1,548	798
Red Grouper	*Epinephelus morio*	230,948	890	191	–
Red Hind	*Epinephelus guttatus*	369	698	41,819	778
Rock Hind	*Epinephelus adscensionis*	749	485	629	725
Scamp	*Mycteroperca phenax*	65,225	858	–	–
Snowy Grouper	*Hyporthodus niveatus*	19,081	1178	–	–
Speckled Hind	*Epinephelus drummondhayi*	12,870	1023	–	–
Tiger Grouper	*Mycteroperca tigris*	8	–	3,391	890
Warsaw Grouper	*Hyporthodus nigritus*	1,246	1940	–	–
Yellowedge Grouper	*Hyporthodus flavolimbatus*	40,520	1080	5	–
Yellowfin Grouper	*Mycteroperca venenosa*	463	973	1,134	920
Yellowmouth Grouper	*Mycteroperca interstitialis*	364	846	92	–
Snapper	**Lutjanidae**				
Black Snapper	*Apsilus dentatus*	13	–	492	610
Blackfin Snapper	*Lutjanus buccanella*	2,549	778	10,668	640
Cardinal Snapper	*Pristipomoides macropthalmus*	97	–	4,178	608
Cubera Snapper	*Lutjanus cyanopterus*	394	1223	821	1200
Dog Snapper	*Lutjanus jocu*	270	–	2,186	1192
Gray Snapper	*Lutjanus griseus*	41,252	790	1,195	780
Lane Snapper	*Lutjanus synagris*	12,337	686	50,185	560
Mahogany Snapper	*Lutjanus mahogoni*	14	–	2,224	722
Mutton Snapper	*Lutjanus analis*	20,761	889	10,141	914
Queen Snapper	*Etelis oculatus*	1,917	950	13,389	910
Red Snapper	*Lutjanus campechanus*	140,989	920	7	–
Schoolmaster	*Lutjanus apodus*	87	–	8,964	666
Silk Snapper	*Lutjanus vivanus*	4,820	853	37,121	721
Vermilion Snapper	*Rhomboplites aurorubens*	140,855	554	15,524	560
Wenchman	*Pristipomoides aquilonaris*	308	540	2,295	673
Yellowtail Snapper	*Ocyurus chrysurus*	107,147	633	117,290	701
Grunt	**Haemulidae**				
Barred Grunt	*Conodon nobilis*	–	–	372	401
Black Margate	*Anisotremus surinamensis*	639	700	333	565
Bluestriped Grunt	*Haemulon sciurus*	168	–	11,558	450
Burro Grunt	*Pomadasys crocro*	–	–	978	296
Caesar Grunt	*Haemulon carbonarium*	111	–	2,675	320
Cottonwick	*Haemulon melanurum*	89	–	1,161	393

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Common Name	**Scientific** Name	**Florida** n	L_{99}	**Caribbean** n	L_{99}
French Grunt	*Haemulon flavolineatum*	29	–	8,734	320
Margate	*Haemulon album*	721	790	737	640
Pigfish	*Orthopristis chrysoptera*	337	430	4	–
Porkfish	*Anisotremus virginicus*	431	331	1,879	352
Sailor's Choice	*Haemulon parra*	520	364	792	410
Tomtate	*Haemulon aurolineatum*	675	306	693	338
White Grunt	*Haemulon plumieri*	12,610	635	78,956	422
Porgy	**Sparidae**				
Grass Porgy	*Calamus arctifrons*	377	303	–	–
Jolthead Porgy	*Calamus bajonado*	4,206	783	4,784	485
Knobbed Porgy	*Calamus nodosus*	1,632	460	–	–
Littlehead Porgy	*Calamus proridens*	2,423	458	33	–
Pluma Porgy	*Calamus pennatula*	1	–	10,694	446
Red Porgy	*Pagrus pagrus*	29,643	584	1	–
Saucereye Porgy	*Calamus calamus*	194	–	2,974	435
Sheepshead	*Archosargus probatocephalus*	9,042	628	1	–
Sheepshead Porgy	*Calamus penna*	72	–	412	395
Whitebone Porgy	*Calamus leucosteus*	895	690	–	–
Triggerfish	**Balistidae**				
Gray Triggerfish	*Balistes capriscus*	22,619	646	146	–
Ocean Triggerfish	*Canthidermis sufflamen*	528	590	317	621
Queen Triggerfish	*Balistes vetula*	522	573	27,650	596
Wrasse & Parrotfish	**Labridae**				
Hogfish	*Lachnolaimus maximus*	4,154	803	6,729	765
Princess Parrotfish	*Scarus taeniopterus*	1	–	5,277	377
Queen Parrotfish	*Scarus vetula*	2	–	2,094	420
Redband Parrotfish	*Sparisoma aurofrenatum*	–	–	9,467	335
Redtail Parrotfish	*Sparisoma chrysopterum*	98	–	54,474	500
Spanish Hogfish	*Bodianus rufus*	2	–	600	445
Stoplight Parrotfish	*Sparisoma viride*	20	–	47,121	505
Yellowtail Parrotfish	*Sparisoma rubripinne*	6	–	2,987	394
Barracuda	**Sphyraenidae**				
Great Barracuda	*Sphyraena barracuda*	752	1290	596	1245
Surgeonfish	**Acanthuridae**				
Blue Tang	*Acanthurus coeruleus*	15	–	36,696	325
Doctorfish	*Acanthurus chirurgus*	19	–	13,772	378
Ocean Surgeonfish	*Acanthurus bahianus*	–	–	5,232	345
Squirrelfish	**Holocentridae**				
Longspine Squirrelfish	*Holocentrus rufus*	21	–	4,796	305

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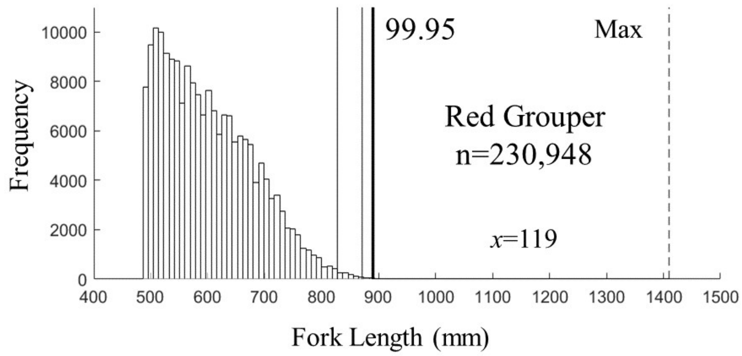
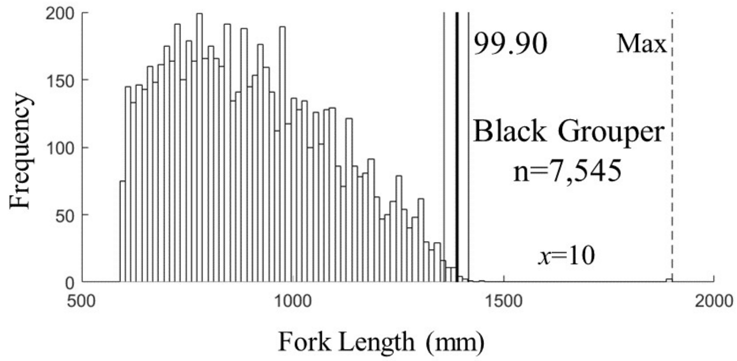
Common	**Scientific**	**Florida**		**Caribbean**	
Name	Name	n	L_{99}	n	L_{99}
Squirrelfish	*Holocentrus adscensionis*	385	387	5,385	485
Goatfish	**Mullidae**				
Spotted Goatfish	*Pseudupeneus maculatus*	6	–	14,853	350
Yellow Goatfish	*Mulloidichthys martinicus*	42	–	8,025	418
Boxfish	**Ostraciidae**				
Honeycomb Cowfish	*Acanthostracion polygonius*	–	–	12,537	610
Scrawled Cowfish	*Acanthostracion quadricornis*	1	–	6,514	545
Smooth Trunkfish	*Lactophrys triqueter*	–	–	2,644	397
Spotted Trunkfish	*Lactophrys bicaudalis*	–	–	2,361	480
Trunkfish	*Lactophrys trigonus*	6	–	2,089	505
Bigeye	**Priacanthidae**				
Bigeye	*Priacanthus arenatus*	1,170	498	243	–

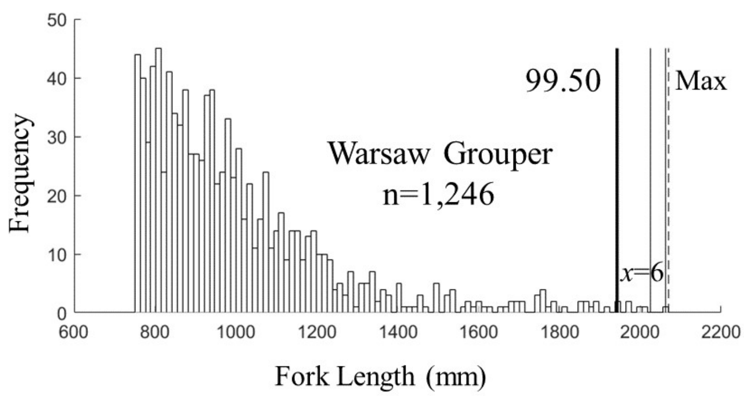
The TIP length observations used to estimate the expected maximum length for each species were an empirical analog to the parameter L_λ, the mean length at maximum observed age. Potential definitions are illustrated in Figure 2.3 using length frequencies for three species with contrasting sample sizes: red grouper (n=230,948; Fig. 2.3a), black grouper (n=7,545; Fig. 2.3b), and warsaw grouper (n=1,246; Fig. 2.3c). For each species, vertical lines denote the maximum length observation (i.e., 100^{th} percentile or L_{100}), and the 99.95, 99.90, and 99.50 percentile length observations.



(a)



(b)



(c)

Figure 2.3: Florida TIP length distributions for (a) red, (b) black, and (c) warsaw groupers with sample size (n). Vertical lines from right to left show the maximum length observation (dashed), and the respective 99.95, 99.90, and 99.50 percentiles. Bolded and labeled vertical lines indicate the L_{99} based on sample size criteria defined in Equation 2.5; x is the number of lengths above L_{99}.

The literature synthesis of life history parameters identified over 300 references for the species listed in Table 2.4. The full list of citations is provided in Appendix A (available upon request). Examples of the completed data tables and variables for the synthesis database (Table 2.3) are provided in Appendix B (available upon request) for five species: black grouper, coney, dog snapper, mutton snapper, and hogfish.

Application of the hierarchical process for selecting the best available life history parameters (Figure 2.2) is illustrated in Table 2.5 for length-age for two example species, mutton snapper and coney. Nine different sets of length-age parameters were obtained for mutton snapper, and seven sets were obtained for coney. Information pertaining to hierarchy level 1 (region), level 2 (biological and statistical methodology), and so forth was summarized for each parameter set. Proceeding left to right from region (level 1) to sample size (n, level 4) criteria, parentheses denote the point in the selection process at which a parameter set was excluded from further consideration. For example, mutton snapper length-age parameters reported in Palazon & Gonzalez (1986) were excluded on the basis of region (temperate vs. tropical), whereas the parameters reported in SEDAR (2008) were excluded on the basis of sample size.

Table 2.5: Examples of the process for selecting the most representative length-age study (and parameters) for mutton snapper and coney following the flow chart from Figure 2.2. For each species, studies are listed by publication date. Major regions were Tropical (Trop) and Temperate (Temp) Western Atlantic; GM denotes Gulf of Mexico subregion. Biological methodology included urohyal bones (U), otoliths (O), otoliths specified as sectioned (O,S), and tag & release (TR). Statistical methodology included nonlinear regression (NLR) with and without back-calculated lengths (BC), and length frequency analysis (LF); CP denotes convergence problems with model fitting. Sample size n was the number of individual fish. Proceeding left to right from level 1 (region) to level 4 (sample size) criteria, parentheses denote the point in the selection process at which a parameter set was excluded from further consideration. The selected study with the best available length-age parameters is highlighted.

Reference	1. Region; subregion	2. Biol.	2. Stats.	3. Age (yr)	3. Length (mm FL)	4. n
Mutton Snapper						
Montes (1975)	Trop; Cuba	–	–	(–)	(–)	(–)
Pozo (1979)	Trop; NE Cuba	U	–	(1–9)	(–)	(2,587)
Claro (1981)	Trop; SW Cuba	O	–	(1–9)	(–)	(–)
Claro (1981)	Trop; NW Cuba	O	–	(1–8)	(–)	(–)
Mason & Manooch (1985)	Trop; E Florida	O	(NLR, BC)	(1–14)	(142–764)	(878)
Palazon & Gonzalez (1986)	(Temp; N Venezuela)	(U)	(–)	(1–8)	(–)	(274)
Burton (2002)	Trop; E Florida	O,S	(NLR, BC)	(1–29)	(170–817)	(1,395)
SEDAR (2008)	Trop; S Atlantic, GM	O,S	NLR	0–40	84–896	(7,172)
O'Hop *et al.* (2015)	Trop; S Atlantic, GM	O,S	NLR	0–40	84–906	13,052
Coney						
Randall (1962)	Trop; St. John	(TR)	(LF)	(–)	(295max)	(–)
Thompson & Munro (1978)	–	(TR)	(LF)	(–)	(–)	(–)
Potts & Manooch (1999)	Trop; S Atlantic	O,S	(NLR-CP)	(2–11)	(150–397)	(55)
Potts & Manooch (1999)	Trop; S Atlantic	O,S	(NLR, BC)	(2–11)	(150–397)	(55)
de Araujo & Martins (2006)	(Temp; Brazil)	(O,S)	(NLR)	(2–25)	(172–428)	(705)
Trott (2006)	(Temp; Bermuda)	(O,S)	(NLR)	(2–28)	(151–384)	(997)
Burton *et al.* (2015)	Trop; S Atlantic	O,S	NLR	1–19	217–430	353

For the example species in Table 2.5, there were many length-age studies available for consideration. In both cases, the studies selected as having the best available length-age parameters were conducted in the preferred region (tropical; Figure 2.2, criterion level 1a) and utilized the most robust biological and statistical methodologies (sectioned otoliths and nonlinear regression, respectively; Figure 2.2, criterion level 2a). For some species, length-age parameters were only available from a single study; accordingly, these parameters were selected as the best available, even though the study may not have been conducted in the preferred region or employed the most preferable biological and statistical methods.

A further consideration of the robustness and reliability of length-age parameters, beyond biological and statistical methods, was the range of age and length observations in a particular growth study (hierarchy level 3, Figure 2.2 and Table 2.5). Of particular concern was how well the oldest and largest fishes in a length-age study corresponded with the maximum age (e.g., a_λ) and length (e.g., L_λ, TIP L_{99}) for a species, in light of the potential for truncated age and length distributions due to exploitation. For length, the largest length in a study (L_{max}) was compared with the L_{99} estimated from the TIP data (Figure 2.4). As illustrated in Figure 2.4 for mutton snapper, the L_{max} (906 mm) reported by O'Hop *et al.* (2015) corresponded well with the TIP L_{99} from Florida (889 mm; Table 2.4) and the U.S. Caribbean (914 mm). There was no comparable sampling program for age composition (e.g., TIP) that might provide an independent estimate of maximum age; thus, the maximum observed age a_λ for a species was obtained from among the same set of length-age studies that provided the parameters of the von Bertalanffy growth function. For mutton snapper, as was typical for most species, the study reporting the oldest age

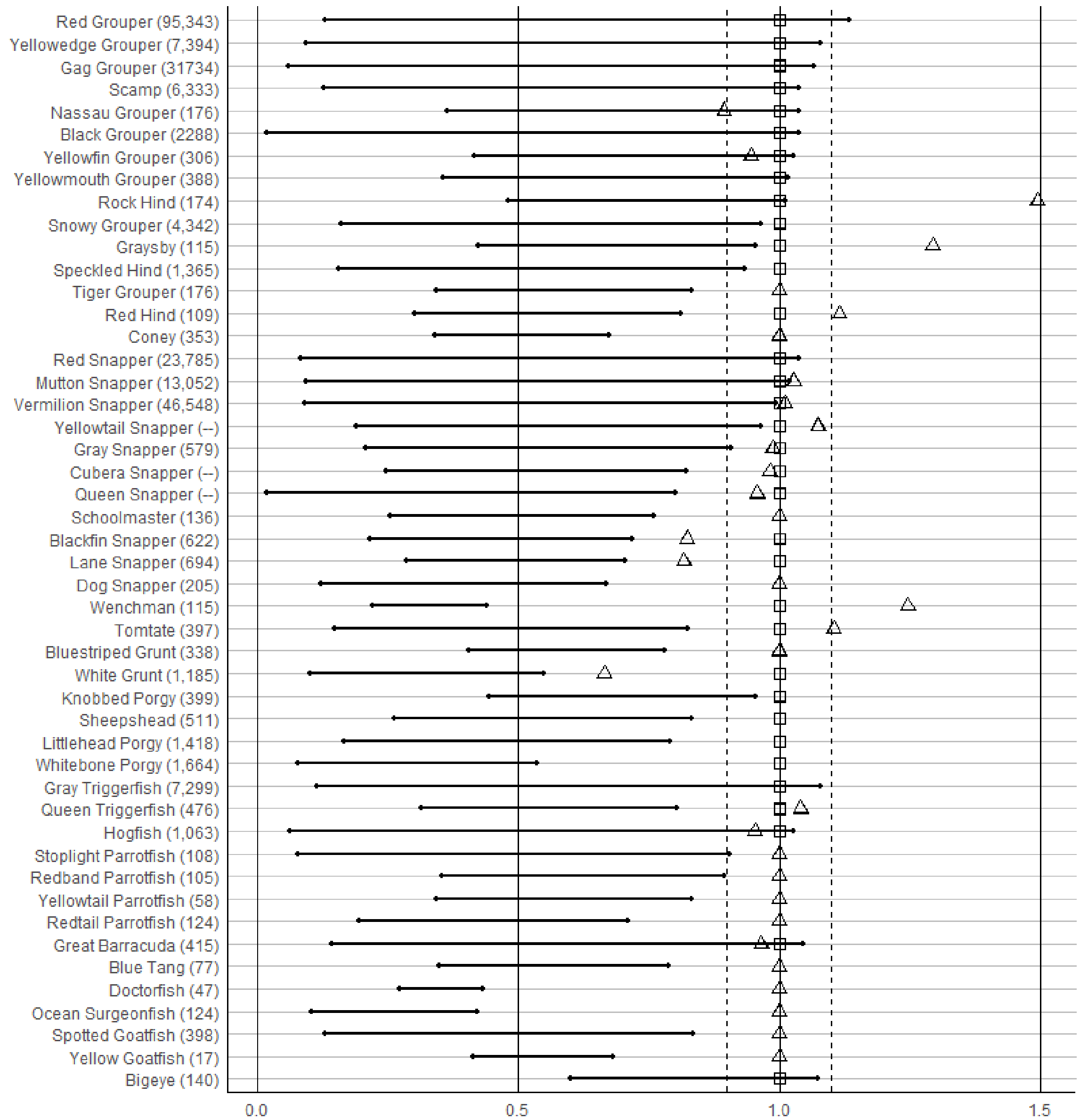


Figure 2.4: Comparison of length ranges from length-age studies (L_{min} to L_{max}, horizontal solid line) with the maximum expected length (L_{99}) from TIP sampling (open squares and triangles). Lengths were standardized to the TIP L_{99} (Table 2.4) from Florida (square), or to the U.S. Caribbean (triangle) if Florida data were not available. Sample sizes of length-age observations are given in parentheses for each species. Length-age parameter reliability was assessed based on whether or not a study's L_{max} exceeded 90% of the TIP L_{99}. Vertical dotted lines notated 10% above and below the TIP L_{99}.

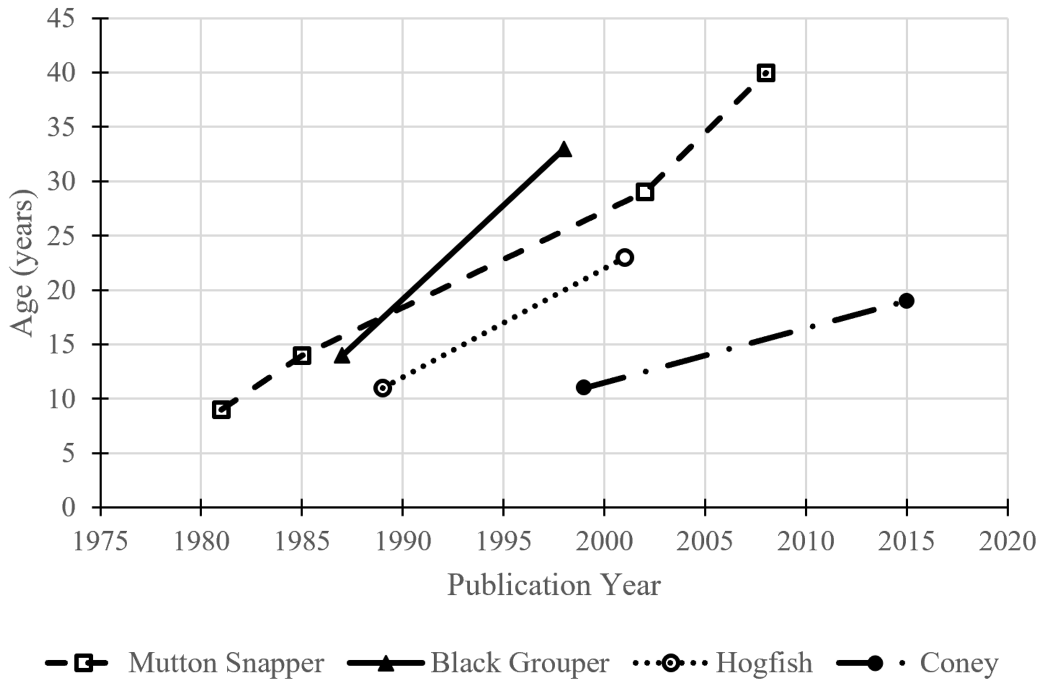


Figure 2.5: Maximum age estimates for mutton snapper (Claro, 1981; Mason & Manooch, 1985; Burton, 2002; SEDAR, 2008), black grouper (Manooch & Mason, 1987; Crabtree & Bullock, 1998), hogfish (Claro *et al.* , 1989; McBride, 2001), and coney (Potts & Manooch, 1999; Burton *et al.* , 2015) have all increased through time.

(O'Hop *et al.* , 2015) was also the study reporting the largest length and the highest sample size of aged fish. An interesting finding for mutton snapper and other species was that as the geographic extent and sample sizes for length-age studies have increased over the past several decades, estimates of maximum age a_λ have also increased, doubling or even quadrupling in some cases (Figure 2.5).

Evaluating length- and age-range criteria for parameter selection was less straightforward for coney (Table 2.5). The studies by de Araujo & Martins (2006); Trott (2006) conducted in temperate regions sampled older fish (25-28 yrs) compared to the tropical study of Burton et al. (2015; 19 yrs), but the L_{max} in the temperate studies (384-428 mm) was similar to or smaller than the tropical study L_{max} (430 mm). Likewise, the minimum age of sampled fish was younger but the minimum length was larger in the tropical study compared to the temperate studies. This indicated that growth was generally slower in the cooler temperate environments and faster in the warmer tropical environment. The coney length-age parameters from Burton *et al.* (2015) were considered to be the most representative for the target region (tropical) of this synthesis.

There were some exceptions to the region criterion (hierarchy level 1). An example case was dog snapper. Length-age curves are shown in Figure 2.6 for the tropical study (Cuba) by Claro *et al.* (1999) and the temperate study (Brazil) by Previero *et al.* (2011). Claro *et al.* (1999) developed sex-specific growth functions, which showed that the average length-at-age of males was larger than females at older ages ($> 10 - 15$ yrs). The pooled-sex growth model of Previero *et al.* (2011) predicted mean length-at-age between the respective male and female curves at older ages. While it is possible to account for sex-specific growth in stock assessments, more data are required (e.g., sex-specific catch composition) than are typically available. Weighing practical considerations over increased biological realism, the length-age parameters of Previero *et al.* (2011) were selected as the best available set.

The sampling sub-region within the tropics was given lower priority in the hierarchical selection process (Figure 2.2, hierarchy level 5). This was based in part on the information shown in Figure 2.4. There were 19 species meeting the following conditions: (i) the range of lengths was reported for the length-age study selected as having the best available parameter set; and (ii) the TIP L_{99} was estimated for both the Florida and Caribbean sub-regions. For comparison purposes, the TIP L_{99}

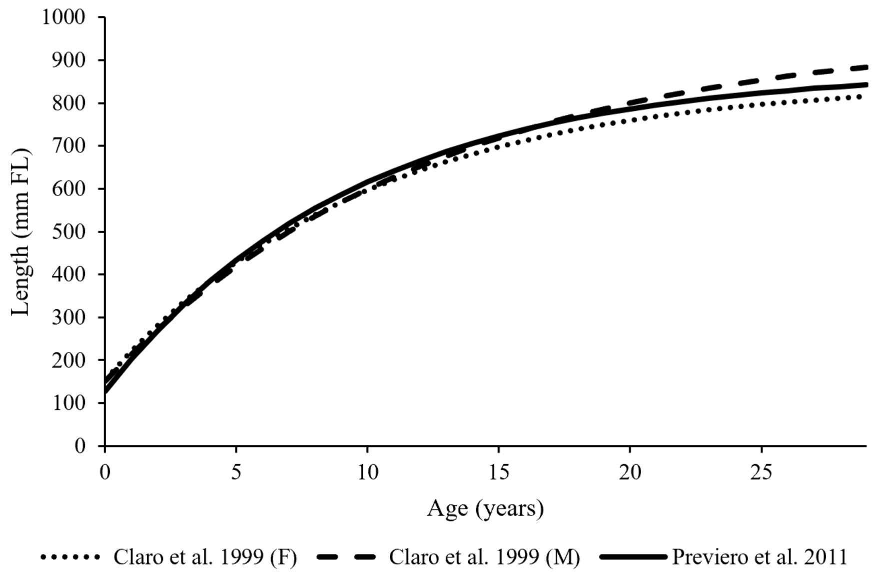


Figure 2.6: Comparison of Dog Snapper sex-specific growth curves for the tropical study (Cuba) of Claro *et al.* (1999) with the pooled sex growth curve for the temperate study (Brazil) of Previero *et al.* (2011).

was standardized to the Florida value (i.e., Florida L_{99}=100%). The Caribbean L_{99} was within 10% of the Florida value for 10 of the 18 species, was greater than 10% of the Florida value for 5 species, and was lower than 10% of the Florida value for 4 species. Thus, there was no discernible trend in maximum expected length between the Florida and Caribbean sub-regions.

After selection of the best available parameters for length-age, weight-length, and maturity, a species-level score (LH) was assigned to distinguish the completeness and reliability of the parameter set as a whole (Table 2.6). An initial score was given based on the completeness of the parameter sets (LH=0 for incomplete, LH=1 for complete). Complete sets were further distinguished with respect to the robustness of the biological and statistical methodologies used to develop the length-age and maturity parameters. A score of LH=2 was given if animals were aged from sec-

Table 2.6: Criteria for a reliability score, LH, for the complete set of length-age, weight-length, and maturity life history parameters for a given species.

LH	**Criteria**
0	Missing life history information for length-age, weight-length, or maturity
1	Complete set of life history parameters: length-age, weight-length, and maturity
2	Conditions for LH=1, and parameters for both length-age and maturity developed from the most robust biological and statistical methodologies (Figure 2.2, hierarchy level 2a)
3	Conditions for LH=2, and L_{max} of length-age study >90% of TIP L_{99} (Figure 2.4)

tioned otoliths or spines, maturity was based on histological examination of gonads, length-age functions (Equation 2.1) were fit using nonlinear regression, and maturity functions (Equation 2.3) were fit using logistic regression. For species meeting the criteria for LH=2, the highest score (LH=3) was given if the L_{max} of the length-age study was greater than 90% of the expected maximum length (TIP L_{99}; Figure 2.4). For species that were commercially exploited in both Florida and the U.S. Caribbean, the L_{max} condition was required to be met for L_{99} values from both sub-regions. A pre-condition of the LH=3 score was that the length range was reported for the length-age study; this was not always the case.

Length parameters are provided in units of millimeters, and weight parameters are provided in units of kilograms, converted from the original units where necessary using Equations (2.6-2.8). Parameters for the length-age growth function (L_∞, K, a_0) and weight-length function (α, β) are given in the original length type. If length type differed from fork length, length-length conversion equations are provided in Table 2.7 where possible. The length type for parameters L_λ and L_m were converted to fork length if necessary. The life history parameters selected as the best available are provided in Table 2.8 for 84 species. The associated references for length-age, weight-length, and maturity parameters are listed in Table 2.9. In some cases the maximum observed age a_λ was obtained from a reference that was different from the study that provided the parameters of the length-age growth function. For these species, two length-age references are listed in Table 2.9 with the second citation providing the source for a_λ. Of the 84 species in Table 2.8, 46 had a complete set of life history parameters (LH$\geq$ 1). Of these, 18 species were given a reliability score of LH=2 and 14 species met the criteria for the highest score of LH=3.

Table 2.7: Length-length conversions utilized to standardize all lengths from standard length (SL), total length (TL), or maximum total length (TLm) to fork length (FL) mm.

Species	**Conversion**	**Units**	**Reference**
Black Grouper	TL = -1.40 + 1.028 FL	mm	SEDAR (2010)
Speckled Hind	FL = -1.88 + 0.982 TL	mm	Ziskin (2008)
Yellowedge Grouper	FL = 15.87 + 0.935 TL	mm	SEDAR (2011)
Yellowfin Grouper	FL = 18.63 + 0.93 TL	mm	Burton *et al.* (2015)
Blackfin Snapper	FL = 3.38 + 0.91 TL	mm	Burton *et al.* (2016)
Gray Snapper	TL = 8.35 + 1.048 FL	mm	Fischer *et al.* (2005)
Mutton Snapper	TL = 10.02 + 1.065 FL	mm	SEDAR (2008)
Queen Snapper	FL = -1.003 + 0.837 TL	cm	Gobert *et al.* (2005)
Red Snapper	TLm = 0.39 + 1.06 FL	in	SEDAR (2013a)
Yellowtail Snapper	FL = 25.85 + 0.75 TL	mm	O'Hop *et al.* (2012)
Tomtate	TL = -1.82 + 1.154 FL	mm	Manooch & Barans (1982)
White Grunt	TL = 1.15 FL	cm	Gaut & Munro (1974)
Knobbed Porgy	See whitebone		
Red Porgy	TL = 6.07 + 1.14 FL	mm	SEDAR (2006)
Whitebone Porgy	FL = -2.0 + 0.86 TL	mm	Waltz *et al.* (1982)
Redband Parrotfish	SL = 0.418 + 0.788 TL	cm	Molina-Urena & Ault (2007)
Redtail Parrotfish	SL = -0.293 + 0.792 TL	cm	Molina-Urena & Ault (2007)
Stoplight Parrotfish	SL = 0.83 FL	mm	Choat *et al.* (2003)

Table 2.8: Life history parameter estimates for all commercially exploited reef fishes in Florida and the U.S. Caribbean where lengths are in mm fork length and weights are in kg. Exceptions for length type are denoted by superscripts: a, length type is total length; b, length type is standard length; c, length type was converted to fork length using corresponding equation in Table 2.7. Life History (LH) denotes the reliability score (Table 2.6).

		Length-Age					Weight-Length		Maturity	
Species	**LH**	a_λ	L_λ	L_∞	K	a_0	α	β	L_m	a_m
Groupers										
Atlantic Creolefish	0	–	–	314.0	0.28	0.00	1.22E-08	3.04	–	–
Black	3	33	1289.4[c]	1334.0[a]	0.14	-0.90	8.75E-09	3.08	834.4[c]	6.48
Coney	1	19	372.8	377.0	0.20	-3.53	1.45E-08	3.03	220.0	0.85
Gag	3	31	1259.7	1278.0	0.13	-0.67	1.17E-08	3.02	543.0	3.50
Goliath	1	37	2156.1	2221.1	0.09	-0.68	6.49E-09	3.15	1200.0	6.50
Graysby	1	13	378.4	446.0	0.13	-1.51	8.81E-09	3.12	165.0	2.04
Misty	0	150	–	–	–	–	–	–	–	–
Mutton Hamlet	0	–	–	–	–	–	–	–	180.0	–
Nassau	3	22	844.9	932.0	0.10	-1.70	4.17E-09	3.20	435.0	4.59
Red	3	29	810.0	829.0	0.13	-1.20	5.46E-09	3.18	292.0	2.27
Red Hind	2	18	514.9	571.0	0.11	-3.10	6.17E-09	3.14	215.0	1.20
Rock Hind	0	33	498.1	499.4	0.17	-2.50	1.39E-08	3.03	–	–
Scamp	3	31	740.1	772.0	0.09	-4.40	2.46E-08	2.91	332.0	1.85
Snowy	3	35	1034.7	1065.0	0.09	-2.88	4.63E-08	2.82	585.1	5.60
Speckled Hind	3	35	859.6[c]	888.0[a]	0.12	-1.80	1.10E-08[a]	3.10	520.5[c]	6.60
Tiger	2	18	680.9[a]	758.0[a]	0.12	-1.88	7.13E-09[a]	3.12	342.0	3.34
Warsaw	1	41	2182.6	2394.0	0.05	-3.62	2.09E-08	2.98	1188.8	9.00

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Species	**LH**	a_λ	L_λ	L_∞	K	a_0	α	β	L_m	a_m
Yellowedge	3	85	950.3[c]	1004.5[a]	0.06	-4.75	1.73E-08	2.96	527.3[c]	8.00
Yellowfin	3	31	935.1	958.0	0.11	-2.94	2.89E-08	2.91	540.4[c]	4.66
Yellowmouth	1	31	747.0	755.0	0.14	-1.42	8.89E-09	3.07	420.0	4.38
Snappers										
Black	0	–	–	560.0	0.30	0.00	–	–	420.0	4.62
Blackfin	1	27	524.8[c]	579.0[a]	0.16	-1.60	9.54E-09	3.11	240.0	2.17
Cardinal	0	–	–	–	–	–	–	–	180.0	–
Cubera	1	54	1031.9	1033.0	0.13	-0.98	7.43E-09	3.12	536.0	4.87
Dog	1	29	842.5	878.0	0.11	-1.49	2.15E-08	2.97	476.0	5.94
Gray	1	28	670.4[c]	717.0[a]	0.17	-0.03	7.22E-09[a]	3.11	230.0	2.42
Lane	1	17	432.9	449.0	0.17	-2.59	5.92E-08	2.86	240.0	1.91
Mahogany	0	–	–	–	–	–	8.18E-08	2.72	–	–
Mutton	3	40	798.0[c]	861.0[a]	0.17	-1.23	1.48E-08	3.03	323.0[c]	2.07
Queen	0	–	–	1020.0[a]	0.40	-0.29	4.02E-08	2.83	–	–
Red	3	48	800.1[c]	856.4[a]	0.19	-0.39	1.87E-08	2.95	455.8[c]	4.00
Schoolmaster	1	42	479.8[a]	482.0[a]	0.12	-2.79	9.26E-09[a]	3.11	250.0[a]	3.30
Silk	0	–	–	756.7	0.10	-2.08	1.66E-08	3.03	500.0	8.73
Vermilion	3	26	344.0	344.0	0.33	-0.80	2.19E-08	2.92	140.9	0.82
Wenchman	0	14	231.8	240.0	0.18	-4.75	3.00E-08	2.91	–	–
Yellowtail	3	23	474.2[c]	618.0[a]	0.13	-3.13	6.14E-08	2.78	232.1[c]	1.70
Grunts										
Barred	0	–	–	325.0	0.43	0.00	–	–	–	–
Black Margate	0	–	–	–	–	–	2.39E-09	3.39	–	–

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Species	LH	a_λ	L_λ	L_∞	K	a_0	α	β	L_m	a_m
Bluestriped	1	23	313.9	314.0	0.32	-1.80	9.31E-09	3.13	204.8	1.50
Burro	0	–	–	–	–	–	2.16E-08	2.93	–	–
Caesar	0	–	–	–	–	–	–	–	–	–
Cottonwick	0	–	–	350.0	0.32	-0.10	2.52E-08	2.95	190.0	2.35
French	0	–	–	350.0	0.24	0.00	9.06E-09	3.16	160.0	2.55
Margate	0	–	–	730.0	0.19	-0.30	1.52E-08	3.04	310.0	2.61
Pigfish	0	4	–	–	–	–	9.71E-09	3.19	210.0	–
Porkfish	0	–	–	–	–	–	1.01E-08	3.17	–	–
Sailor's Choice	0	–	–	388.0	0.24	-0.27	2.02E-08	2.99	–	–
Tomtate	1	9	242.1[c]	310.0[a]	0.22	-1.28	6.19E-09	3.21	131.6[c]	1.75
White	2	18	280.9[c]	323.1[a]	0.52	-0.58	8.49E-08	2.75	167.0	1.16
Porgies										
Grass	0	–	–	–	–	–	–	–	–	–
Jolthead	0	–	–	756.0	0.18	-0.12	6.67E-08	2.82	300.0	2.69
Knobbed	0	17	418.8[c]	512.0[a]	0.17	-0.88	7.65E-09[a]	3.13	–	–
Littlehead	2	10	290.3	306.0	0.25	-1.69	6.61E-08	2.82	132.0	0.53
Pluma	0	–	–	–	–	–	1.35E-08	3.11	–	–
Red	1	14	424.4[c]	510.0[a]	0.21	-1.32	2.70E-08[a]	2.89	197.6[c]	1.50
Saucereye	0	–	–	–	–	–	6.78E-08	2.80	–	–
Sheepshead	1	14	478.9	490.4	0.26	-0.42	4.40E-08	2.89	229.0	2.00
Sheepshead Porgy	0	–	–	376.0	0.28	0.00	2.81E-07	2.54	–	–
Whitebone	1	12	304.7	331.0	0.17	-2.64	4.30E-08	2.91	256.8	6.00
Triggerfishes										

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Species	LH	a_λ	L_λ	L_∞	K	a_0	α	β	L_m	a_m
Gray	3	14	523.9	589.7	0.14	-1.66	2.16E-08	3.01	210.8	1.50
Ocean	0	–	–	–	–	–	1.55E-08	3.06	–	–
Queen	1	14	393.0	441.3	0.14	-1.80	8.64E-08	2.78	215.0	2.97
Wrasses and Parrotfishes										
Hogfish	3	23	784.3	849.0	0.11	-1.33	9.50E-08	2.75	176.8	0.88
Princess	0	–	–	–	–	–	5.95E-07	2.39	175.0	–
Queen	0	–	–	–	–	–	–	–	–	–
Redband	1	7	172.8^b	178.0^b	0.67	0.00	8.35E-08	2.74	140.0	0.88
Redtail	1	5	246.8^b	258.0^b	0.63	0.00	8.89E-07	2.32	235.0	1.22
Spanish Hogfish	0	–	–	–	–	–	–	–	100.0^b	–
Stoplight	1	9	350.9^b	357.0^b	0.45	-0.06	3.70E-08	2.91	205.0	–
Yellowtail	1	7	237.2^b	238.0^b	0.81	-0.05	1.35E-08	3.06	170.0^b	1.49
Barracudas										
Great Barracuda	1	19	1229.0	1236.4	0.26	-0.71	7.94E-09	2.97	800.0	3.30
Surgeonfishes										
Blue Tang	1	27	219.0	219.0	0.88	-0.15	1.50E-06	2.26	130.0	0.87
Doctorfish	1	12	210.0	210.0	1.10	-0.12	9.23E-08	2.74	170.0	1.38
Ocean Surgeonfish	1	13	183.0	183.0	1.06	-0.15	2.51E-08	2.98	110.0	0.72
Squirrelfishes										
Longspine Squirrelfish	0	–	–	188.0	0.48	0.00	–	–	135.0	2.64
Squirrelfish	0	–	–	261.0	0.23	0.00	2.39E-07	2.56	145.8	3.56

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Species	**LH**	a_λ	L_λ	L_∞	K	a_0	α	β	L_m	a_m
Goatfishes										
Spotted	1	5	241.9	332.3	0.27	0.09	2.29E-08	2.96	175.0	2.91
Yellow	0	–	–	300.0	0.40	0.00	1.10E-08	3.09	160.0	1.91
Boxfishes										
Honeycomb Cowfish	0	–	–	–	–	–	1.08E-07	2.68	–	–
Scrawled Cowfish	0	–	–	–	–	–	9.56E-07	2.26	222.0	–
Smooth Trunkfish	0	–	–	–	–	–	1.82E-06	2.23	–	–
Spotted Trunkfish	0	–	–	–	–	–	–	–	–	–
Trunkfish	0	–	–	–	–	–	1.43E-07	2.66	–	–
Bigeyes										
Bigeye	1	18	644.5	665.0	0.17	-2.90	1.19E-08	3.04	138.0	-1.52

Table 2.9: Citations for the selected life history parameters in Table 2.8. If a length-age or maturity study did not use ideal biological or statistical methodology outlined in Figure 2.2 (hierarchical level 2*a*) or if a weight-length study had less than 30 samples, the reference was denoted with superscript *a*. For species with two length-age citations, the first is the source for parameters of the length-age function (Equation 2.1), the second is the source for maximum age; superscript *b* denotes the source for maximum age where there was no length-age function.

Species	Length-Age	Weight-Length	Maturity
Groupers			
Atlantic Creolefish	Posada & Appeldoorn (1996)[a]	Bohnsack & Harper (1988)[a]	–
Black	SEDAR (2010)	SEDAR (2010)	SEDAR (2010)
Coney	Burton *et al.* (2015)	Burton *et al.* (2015)	Trott (2006)[a]
Gag	SEDAR (2014)	SEDAR (2014)	SEDAR (2014)
Goliath	SEDAR (2016c)	SEDAR (2016c)	SEDAR (2016c)[a]
Graysby	Potts & Manooch (1999)	Potts & Manooch (1999)	Nagelkerken (1979)[a]
Misty	Luckhurst & Dean (2009)[b]	–	–
Mutton Hamlet	–	–	Marques & Ferreira (2011)[a]
Nassau	Cushion (2010)	Cushion (2010)	Cushion (2010)
Red	SEDAR (2015b)	SEDAR (2015b)	SEDAR (2015b)
Red Hind	Cushion (2010), Sadovy *et al.* (1992)	Sadovy *et al.* (1992)	Sadovy *et al.* (1994)
Rock Hind	Potts & Manooch (1995), Burton *et al.* (2012)	Burton *et al.* (2012)	–
Scamp	Lombardi *et al.* (2012)	Matheson *et al.* (1986)	Lombardi *et al.* (2012)
Snowy	SEDAR (2013b)	SEDAR (2013b)	SEDAR (2013b)
Speckled Hind	Ziskin *et al.* (2011)	Ziskin (2008)	Ziskin *et al.* (2011)
Tiger	Garcia-Arteaga *et al.* (1999)	Garcia-Arteaga *et al.* (1999)	Caballero-Arango *et al.* (2013)

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Species	Length-Age	Weight-Length	Maturity
Warsaw	Manooch & Mason (1987)	Manooch & Mason (1987)	Manooch (1984)[a]
Yellowedge	SEDAR (2011)	SEDAR (2011)	SEDAR (2011)
Yellowfin	Burton *et al.* (2015)	Burton *et al.* (2015)	Cushion (2010)
Yellowmouth	Burton *et al.* (2014)	Burton *et al.* (2014)	Bullock & Murphy (1994)[a]
Snappers			
Black	Thompson & Munro (1983)[a]	–	Thompson & Munro (1983)[a]
Blackfin	Burton *et al.* (2016)	Burton *et al.* (2016)	Thompson & Munro (1983)[a]
Cardinal	–	–	Thompson & Munro (1983)[a]
Cubera	Baisre & Paez (1981), Shertzer *et al.* (2017)	Claro & Garcia (2001)	Baisre & Paez (1981)[a]
Dog	Previero *et al.* (2011)	Previero *et al.* (2011)	Claro & Garcia-Arteaga (1994)[a]
Gray	Burton (2001), Fischer *et al.* (2005)	Burton (2001)	Starck (1970)[a]
Lane	SEDAR (2016a)	SEDAR (2016a)	SEDAR (2016a)[a]
Mahogany	–	Bohnsack & Harper (1988)[a]	–
Mutton	O'Hop *et al.* (2015)	SEDAR (2008)	SEDAR (2008)
Queen	Murray & Moore (1993)[a]	Gobert *et al.* (2005)	–
Red	SEDAR (2013a)	SEDAR (2013a)	SEDAR (2013a)
Schoolmaster	Potts *et al.* (2016)	Potts *et al.* (2016)	Thompson & Munro (1983)[a]
Silk	Poso & Espinosa (1982)	Poso & Espinosa (1982)	Poso & Espinosa (1983)[a]
Vermilion	SEDAR (2016b)	SEDAR (2016b)	SEDAR (2016b)
Wenchman	Anderson *et al.* (2009)	Anderson *et al.* (2009)	–
Yellowtail	O'Hop *et al.* (2012)	O'Hop *et al.* (2012)	O'Hop *et al.* (2012)
Grunts			

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Species	Length-Age	Weight-Length	Maturity
Barred	Garcia & Duarte (2006)[a]	–	–
Black Margate	–	Bohnsack & Harper (1988)[a]	–
Bluestriped	Pitt *et al.* (2010)	Pitt *et al.* (2010)	Garcia-Cagide (1986)[a]
Burro	–	Claro & Garcia (2001)[a]	–
Cottonwick	Nelson *et al.* (1985)	Bohnsack & Harper (1988)	Billings & Munro (1974)[a]
French	Dennis (1988)[a]	Bohnsack & Harper (1988)	Billings & Munro (1974)[a]
Margate	Garcia-Arteaga (1983)	Bohnsack & Harper (1988)[a]	Garcia-Cagide (1986)[a]
Pigfish	Taylor (1916)[b]	Bohnsack & Harper (1988)	Hildebrand & Cable (1930)[a]
Porkfish	–	Bohnsack & Harper (1988)	–
Sailor's Choice	Claro *et al.* (2001)	Bohnsack & Harper (1988)	–
Tomtate	Manooch & Barans (1982)	Bohnsack & Harper (1988)	Manooch & Barans (1982)[a]
White	Murphy *et al.* (1999)	Murie & Parkyn (2005)	Murphy *et al.* (1999)
Porgies			
Jolthead	Olaechea *et al.* (1975)	Bohnsack & Harper (1988)	Liubimova & Capote (1971)[a]
Knobbed	Horvath *et al.* (1990)	Horvath *et al.* (1990)	–
Littlehead	Tyler-Jedlund & Torres (2015)	Tyler-Jedlund & Torres (2015)	Tyler-Jedlund & Torres (2015)
Pluma	–	Claro & Garcia-Arteaga (1994)	–
Red	SEDAR (2006)	SEDAR (2006)	SEDAR (2006)[a]
Saucereye	–	Bohnsack & Harper (1988)	–
Sheepshead	Dutka & Murie (2001)	Dutka & Murie (2001)	Render & Wilson (1992)[a]
Sheepshead Porgy	Garcia & Duarte (2006)[a]	Bohnsack & Harper (1988)	–
Whitebone	Waltz *et al.* (1982)	Waltz *et al.* (1982)	Waltz *et al.* (1982)[a]
Triggerfishes			
Gray	SEDAR (2015a)	SEDAR (2015a)	SEDAR (2015a)

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Species	Length-Age	Weight-Length	Maturity
Ocean	–	Bohnsack & Harper (1988)	–
Queen	de Albuquerque *et al.* (2011)	de Albuquerque *et al.* (2011)	Aiken (1975)[a]
Wrasses and Parrotfishes			
Hogfish	Cooper *et al.* (2013)	Cooper *et al.* (2013)	Cooper *et al.* (2013)
Princess	–	Bohnsack & Harper (1988)[a]	Reeson (1975a)[a]
Redband	Choat & Robertson (2002)[a]	Bohnsack & Harper (1988)	Reeson (1975a)[a]
Redtail	Choat & Robertson (2002)[a]	Bohnsack & Harper (1988)	Figuerola *et al.* (1998)[a]
Spanish Hogfish	–	–	Robertson & Warner (1978)[a]
Stoplight	Choat *et al.* (2003)	Bohnsack & Harper (1988)	Figuerola *et al.* (1998)[a]
Yellowtail	Choat & Robertson (2002)	Bohnsack & Harper (1988)[a]	Robertson & Warner (1978)[a]
Barracudas			
Great Barracuda	Kadison *et al.* (2010)	Kadison *et al.* (2010)	Kadison *et al.* (2010)[a]
Surgeonfishes			
Blue Tang	Mutz (2006)	Bohnsack & Harper (1988)	Reeson (1975b)[a]
Doctorfish	Mutz (2006)	Bohnsack & Harper (1988)	Reeson (1975b)[a]
Ocean Surgeonfish	Mutz (2006)	Bohnsack & Harper (1988)	Reeson (1975b)[a]
Squirrelfishes			
Longspine Squirrelfish	Munro (1999)[a]	–	Wyatt (1983)[a]
Squirrelfish	Munro (1999)[a]	Bohnsack & Harper (1988)	Mendes *et al.* (2007)
Goatfishes			
Spotted	Santana *et al.* (2006)	Bohnsack & Harper (1988)	Munro (1976)[a]

continued on next page

Species	Length-Age	Weight-Length	Maturity
Yellow	Munro (1976)[a]	Bohnsack & Harper (1988)	Munro (1976)[a]
Boxfishes			
Honeycomb Cowfish	–	Bohnsack & Harper (1988)	–
Scrawled Cowfish	–	Bohnsack & Harper (1988)	Ruiz *et al.* (1999)
Smooth Trunkfish	–	Bohnsack & Harper (1988)	–
Trunkfish	–	Claro & Garcia (2001)	–
Bigeyes			
Bigeye	Ximenes *et al.* (2009)	Bohnsack & Harper (1988)	Tapia *et al.* (1995)[a]

2.2.2 Description of Florida Commercial Reef Fisheries

Florida commercial reef fish landings and value were dominated by grouper and snapper species. ‘Principal’ species were defined as red grouper, gag grouper, black grouper, yellowtail snapper, gray snapper, and mutton snapper and were assumed to represent the dynamics of the Florida commercial reef fisheries. Over 90% of these species’ nationwide landings were from Florida, and they were within the top 15 reef fish landings over the last two decades (Table 2.10). Yellowtail snapper and red grouper commercial landings exceeded recreational landings; gag grouper, gray snapper, and mutton snapper landings were primarily recreational; and black grouper landings were approximately equally distributed between the commercial and recreational sectors (Figure 2.7)

Florida grouper landings were dominated by red grouper, followed by gag grouper, then black grouper (Figure 2.8). The ‘other’ category of groupers included 16 species identified in the MFTT database. Dominant snapper landings in Florida were split between yellowtail snapper, red snapper, and vermilion snapper, but the majority of the landings for the latter two were not in Florida (Figure 2.9, Table 2.10). Gray snapper commercial landings exceeded mutton snapper, and the ‘other’ snapper category represented 12 species identified in the MFTT database. There were three primary gears that landed over 90% of commercial catch biomass 1995-2016 for the subset of snappers and groupers covered here: hook-and-line (HL), longline (LL), and spearfishing (SP) gears (Table 2.11). HL was the primary gear for gag grouper, black grouper, gray snapper, mutton snapper, and yellowtail snapper; LL was the primary gear for red grouper landings. Yellowtail and gray snappers were almost never landed

Table 2.10: Total commercial reef fish landings in Florida and all of the United States from 1992-2011, sorted in descending Florida biomass. Species chosen for this analysis were abundant in total Florida landings, representative of their respective stocks from Florida commercial fishery data, and annotated in bold font.

Species	**Florida (lbs)**	**USA (lbs)**	**FL/USA %**
Red grouper	**128,404,033**	**133,671,786**	**96.06%**
Gag grouper (Gulf)	**42,869,394**	**53,493,783**	**98.64%**
Yellowtail snapper	**36,638,757**	**36,655,774**	**99.95%**
Vermilion snapper	31,711,613	69,698,918	45.50%
Greater amberjack	27,831,392	32,976,928	84.40%
Red snapper	18,923,201	90,184,888	20.98%
Yellowedge grouper	14,303,342	20,289,509	70.50%
Golden tilefish	13,331,118	17,547,595	75.97%
Grunts	11,760,261	14,177,486	82.95%
Black grouper	**8,095,198**	**8,321,942**	**97.28%**
Scups/Porgies	7,860,254	8,444,791	93.08%
Gray snapper	**7,762,116**	**8,559,452**	**90.68%**
Black sea bass	6,149,884	22,741,713	27.04%
Scamp	6,050,602	12,628,971	47.91%
Mutton snapper	**5,508,212**	**5,585,245**	**98.62%**
Snowy grouper	5,394,385	9,492,956	56.83%

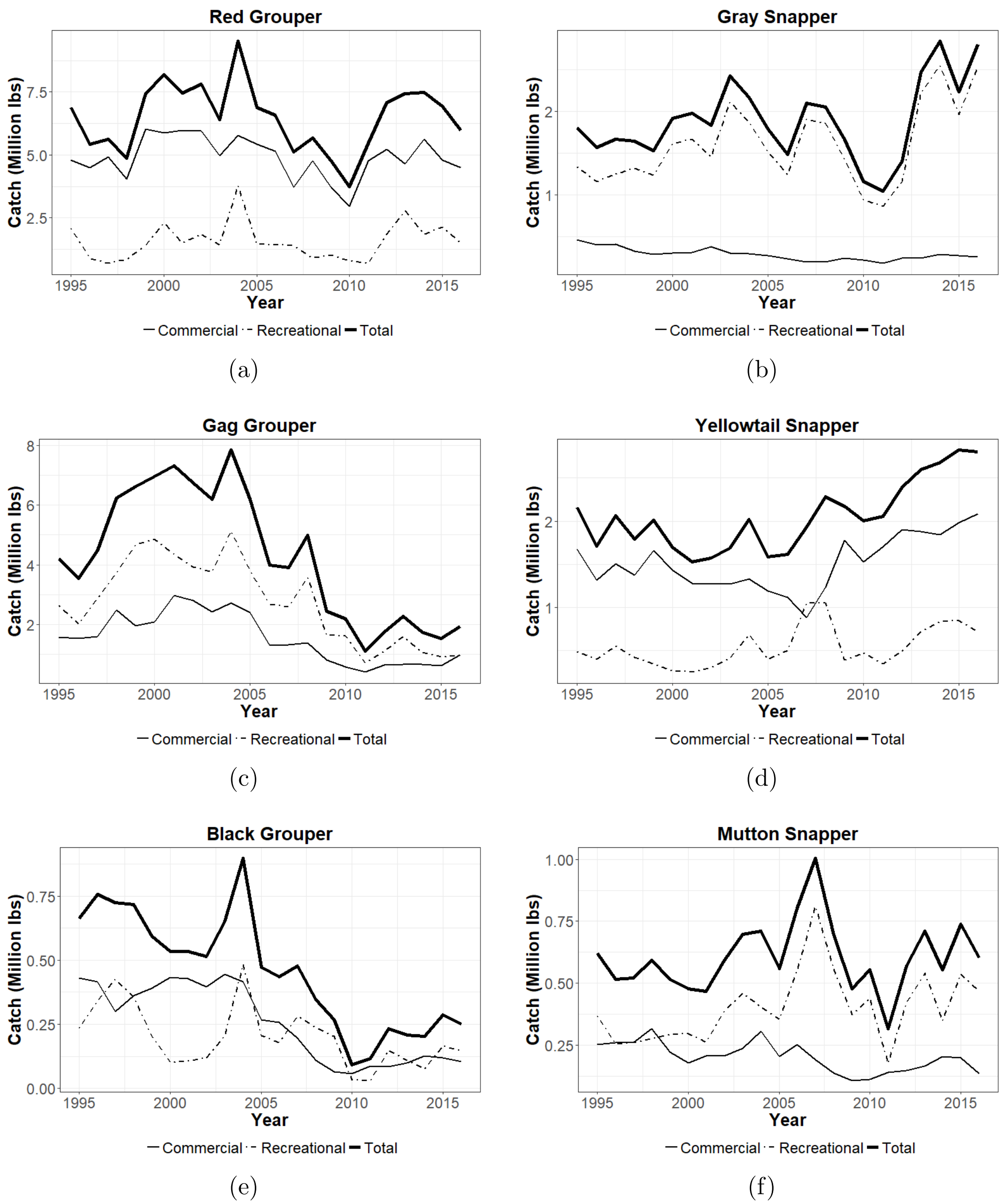


Figure 2.7: Commercial, recreational, and total catch biomass of commercially important groupers (left) and snappers (right) from 1995-2016 with catch biomass in millions of pounds (unequal y-axis ranges).

Table 2.11: Percentage of ‘principal’ reef fish commercial landings by hook-and-line (HL), longline (LL), spearfishing (SP), other (OT), and unknown (UN) gears from 1995-2016 in the MFTT database. Gag grouper and yellowtail snapper (bolded) were selected as ‘analysis’ species.

Species	**HL**	**LL**	**OT**	**SP**	**UN**
Red Grouper	33.2%	58.7%	7.2%	0.4%	0.5%
Gag Grouper	**67.1%**	**26.5%**	**0.7%**	**5.2%**	**0.4%**
Black Grouper	48.8%	38.4%	3.4%	8.4%	0.9%
Gray Snapper	86.1%	3.7%	1.7%	7.8%	0.7%
Mutton Snapper	50.2%	39.5%	7.2%	2.4%	0.7%
Yellowtail Snapper	**98.3%**	**0.3%**	**0.6%**	**0.1%**	**0.8%**

on LL gear, but approximately one-third of gag grouper, black grouper, and mutton snapper landings were caught on LL gear.

From the subset of ‘principal’ species, gag grouper and yellowtail snapper were identified as ‘analysis’ species due to their many contrasting points: grouper *vs.* snapper, primarily recreational *vs.* commercial landings, multiple *vs.* single gear fisheries, and complex *vs.* simple regulatory histories. Gag grouper regulatory history was summarized in Table 2.12. Yellowtail snapper minimum size limit has remained 12” since 1983 with no major changes in commercial or recreational regulations, aside from a commercial closure November-December 2015 when the annual catch limit was reached.

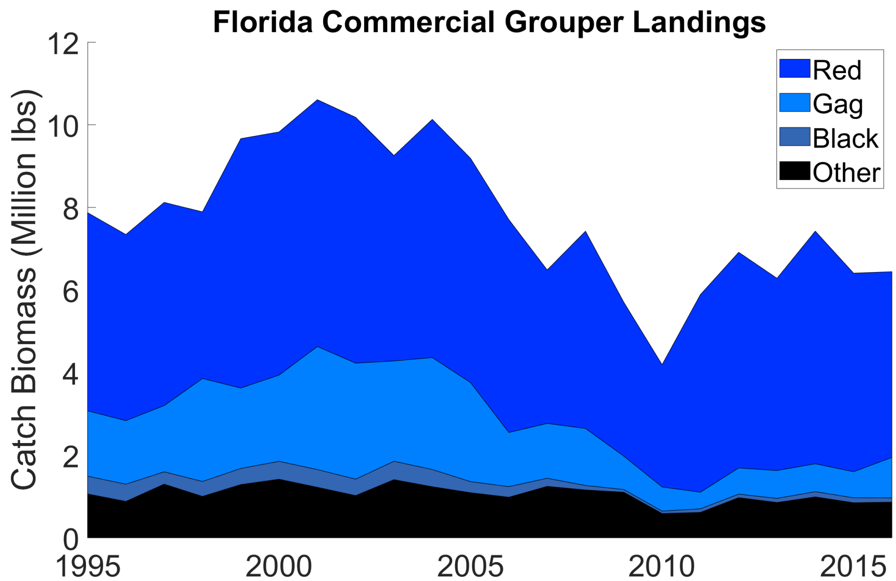


Figure 2.8: Florida commercial grouper landings from top species 1995-2016.

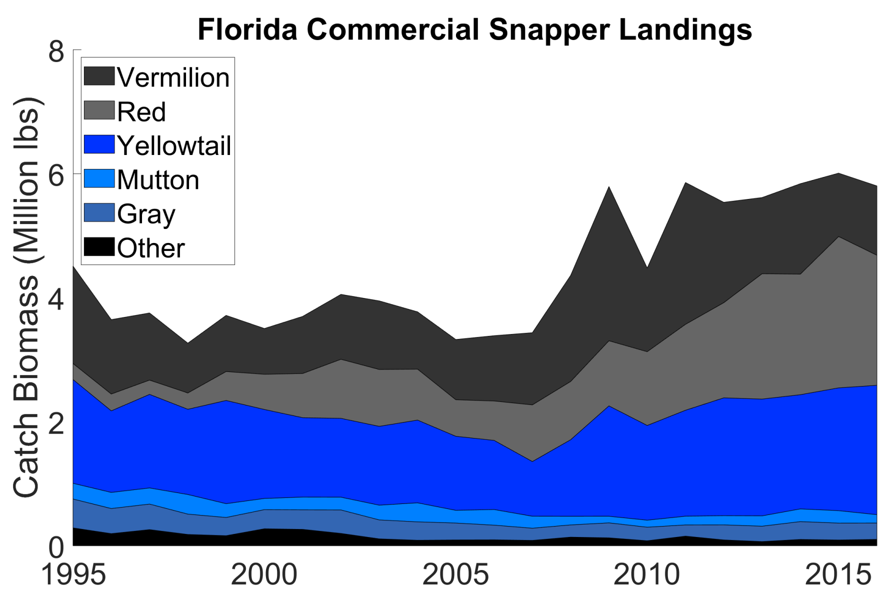


Figure 2.9: Florida commercial snapper landings from top species 1995-2016.

Table 2.12: Gag grouper South Atlantic (SA) and Gulf of Mexico (GM) 2009 regulations were implemented in 1999 for SA and in 2000 for GM; the prior size limit was 20” TL. Annual Catch Limits (ACLs) and Annual Catch Targets (ACTs) were set for gag grouper in 2009, when gag was split from the shallow water grouper complex. Privilege-based fishing was was implemented in the Gulf of Mexico in 2010 with Individual Fishing Quotas (IFQs), when closed seasons were no longer required.

	Gulf of Mexico					**South Atlantic**				
					Commercial					
Year	L_c	Closed Season	Reported	ACT	ACT%	L_c	Closed Season	Reported	ACL	ACL%
2016	22	–	910,996	939,000	97.0	24	Jan 1-Apr 30	230,519	297,882	77.4
2015	22	–	542,774	939,000	57.8	24	Jan 1-Apr 30	278,969	295,459	94.4
2014	22	–	586,377	835,000	70.2	24	Jan 1-Apr 30	334,095	326,722	102.3
2013	22	–	575,335	708,000	81.3	24	Jan 1-Apr 30	366,939	326,722	112.3
2012	22	–	523,138	567,000	92.3	24	Jan 1-Apr 30	355,603	352,940	100.8
2011	24	–	318,663	430,000	74.1	24	Jan 1-Apr 30	428,482	352,940	121.4
2010	24	–	496,826	1,410,000	35.2	24	Jan 1-Apr 30	411,630	352,940	116.6
2009	24	Feb 15-Mar 15	715,814	1,320,000	54.2	24	Mar 1- Apr 30	442,758	352,940	125.4
					Recreational					
Year	L_c	Closed Season	Reported	ACT	ACT%	L_c	Closed Season	Reported	ACL	ACL%
2016	22	Jan 1-May 31	796,430	1,708,000	46.6	24	Jan 1-Apr 30	151,456	312,351	48.5
2015	22	Jan 1-May 31	823,940	1,708,000	48.2	24	Jan 1-Apr 30	58,348	310,023	18.8
2014	22	Jan 1-Jun 30	862,101	1,519,000	56.8	24	Jan 1-Apr 30	169,447	340,060	49.8
2013	22	Jan 1-Jun 30	1,435,421	–	–	24	Jan 1-Apr 30	78,472	340,060	23.1
2012	22	Jan 1-Jun 30 Nov 1-Dec 31	938,547	–	–	24	Jan 1-Apr 30	177,097	340,060	52.1
2011	22	Jan 1-Jun 30 Nov 1-Dec 31	660,287	–	–	24	Jan 1-Apr 30	169,854	340,060	49.9
2010	22	Jan 1-Jun 30 Nov 1-Dec 31	1,664,257	–	–	24	Jan 1-Apr 30	171,841	340,060	50.5
2009	22	(Feb 1-Mar 31)	–	–	–	24	–	–	–	–

2.2.3 Florida Commercial Fleet Validations

Yellowtail snapper and gag grouper required identical federal commercial permits, but their distributions and regulations varied significantly. The federal boundary between the Gulf of Mexico and South Atlantic Fishery Management Councils was ecologically obsolete for yellowtail snapper because it split the stock down the middle. On the other hand, gag grouper was distributed in more temperate waters separated by the tropical waters of south Florida, and the jurisdictional boundary between the Gulf of Mexico and South Atlantic made ecological sense (Figure 2.10). From 1992-2011, 98.6% of the gag grouper Gulf of Mexico commercial catch was landed in Florida, while only 31.0% of the South Atlantic commercial catch was landed in Florida. Furthermore, regulations including size limits, quotas, closed seasons, and definitions of fishing privileges have differed between management jurisdictions (Table 2.12). Yellowtail snapper regulations, on the other hand, have remained consistent across jurisdictions throughout the simulation time frame. Considering all of these points, yellowtail snapper was modeled as a single stock, and gag grouper was modeled as two separate stocks.

Each fleet was comprised of a set of vessels that had landed the 'analysis' species in a given year and had a unique combination of permit type (GM, SA1, SA2) and gear (HL, LL, SP). Fleets were defined for each species individually to ensure the vessels were operating in the 'analysis' species' habitat (Figure 2.10). The MFTT dataset did not have vessel identifying information for approximately half of the vessels that had landed reef fishes, meaning if permit type was utilized (which required vessel information to link to MFTT), data would be lost. Because gag grouper GM/SA

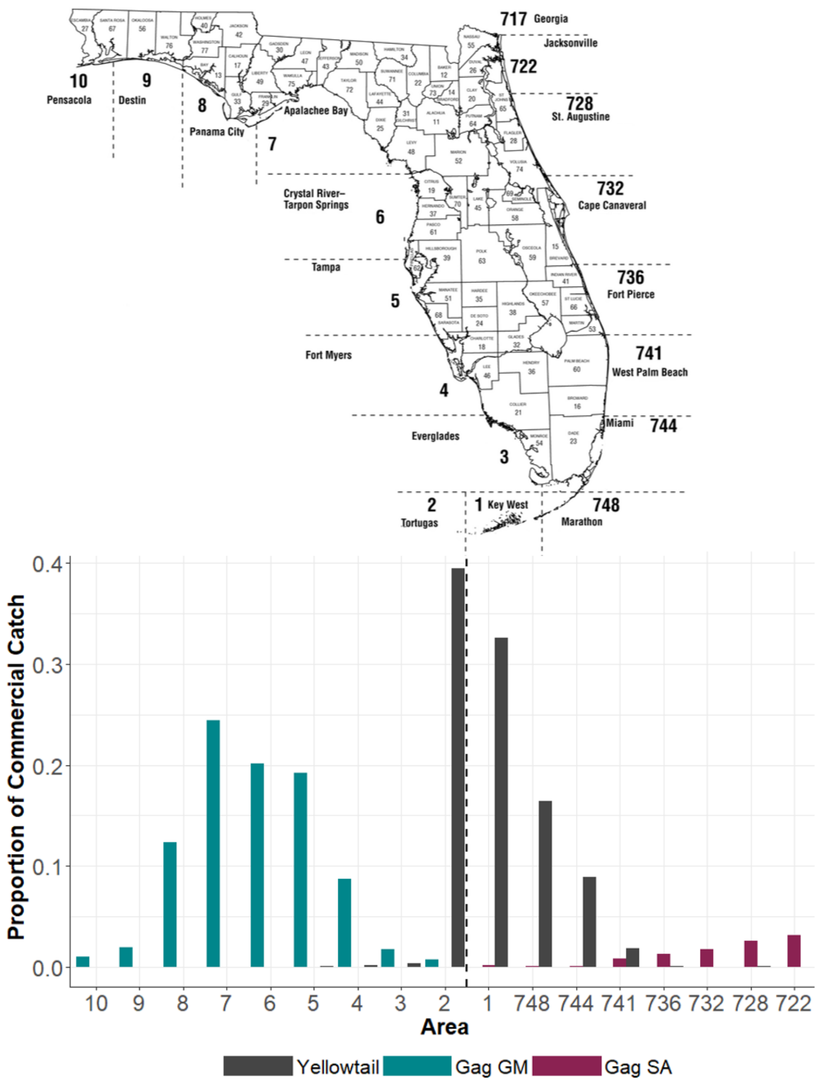


Figure 2.10: Yellowtail snapper and gag grouper catch distribution. Areas along the x-axis run west to east around the state of Florida. Area 10 was the northwestern most part of Florida, Key West was area 1, and area 722 was the northeastern most part of Florida. The Gulf of Mexico (GM) jurisdiction included all areas northwest of the Dry Tortugas (areas 2–10), and the South Atlantic (SA) included all areas northeast of Key West (areas 1–722).

stocks were caught well within their respective jurisdictional boundaries, fishing area was used to infer permit type (i.e. a fish caught in the Gulf of Mexico required a GM permit). Yellowtail snapper were primarily caught where federal jurisdictions overlapped in south Florida, and required the identification of permit type to appropriately define these fleets. Total revenue of yellowtail snapper that could be attributed to a permit type was shown relative to the total statewide revenue (Figure 2.11). Four fleets were identified for yellowtail snapper, all with Hook-and-Line gear: South Atlantic Snapper-Grouper Unlimited Trip Limit (YT/HL/SA1), South Atlantic Snapper-Grouper 225lb Trip Limit (YT/HL/SA2), Gulf of Mexico Reef Fish (YT/HL/GM), and South Atlantic Unlimited Trip in combination with Gulf of Mexico Reef Fish (YT/HL/GMSA1). Six fleets were identified for gag grouper: Gulf of Mexico Reef Fish Hook-and-Line (GAG/HL/GM), Longline (GAG/LL/GM), and Spear (GAG/SP/GM); South Atlantic Snapper-Grouper Hook-and-Line (GAG/HL/SA), Longline (GAG/LL/SA), and Spear (GAG/SP/SA) (Figure 2.12). Data were not sufficient in the FLS database to build a cost function for LL/SA, but was substantial enough in the MFTT to require a fleet definition. LL data was investigated and displayed adequately similar characteristics that all LL data was combined to build a cost function that was applied to the gag grouper Gulf of Mexico and South Atlantic fleets.

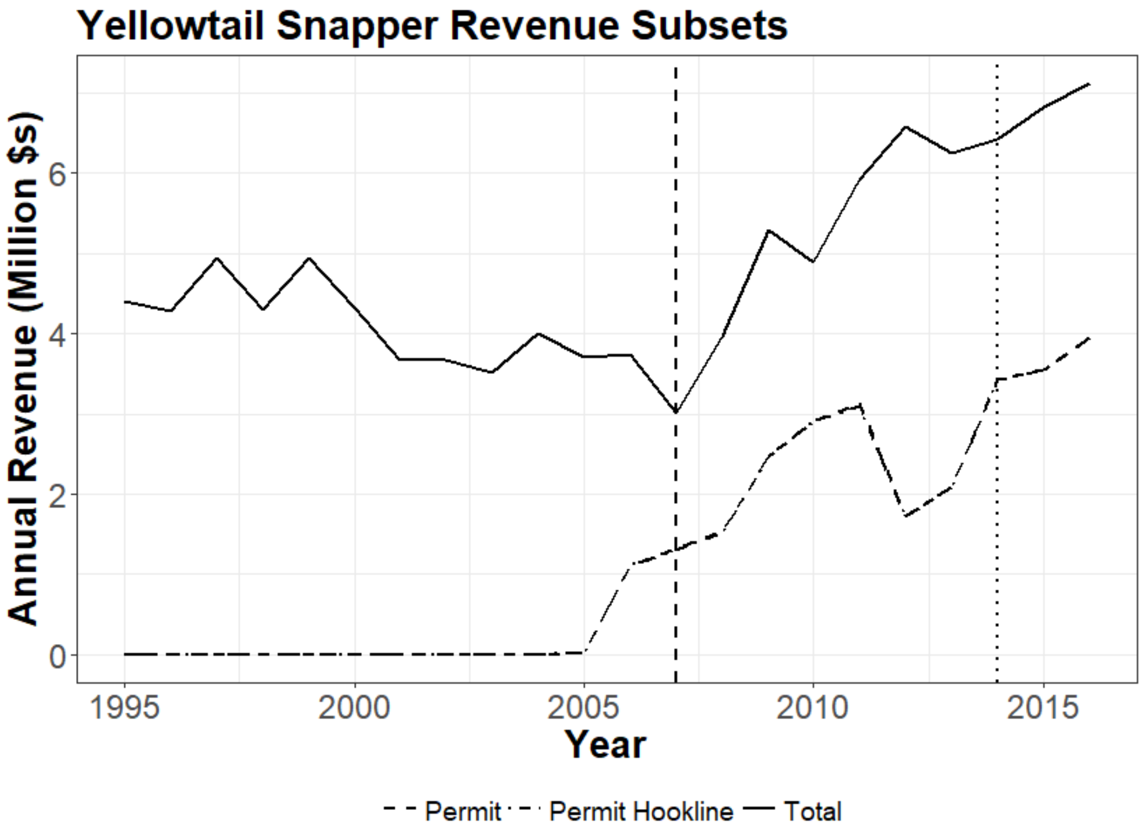
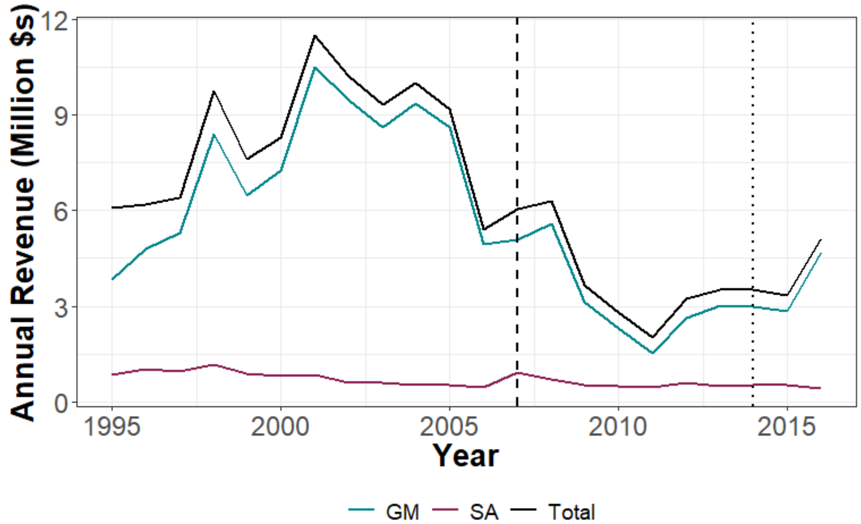
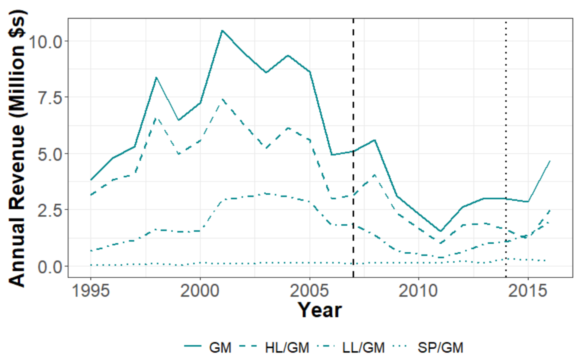


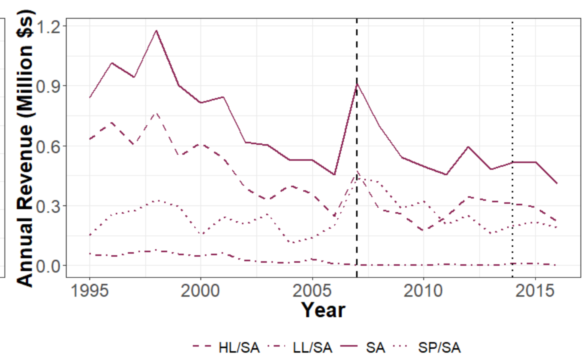
Figure 2.11: Total revenue generated by yellowtail snapper where total landings by active permitted vessels was considered valid in 2007 (dashed lines), and the primary gear was defined from cost samples beginning in 2014 (dotted lines). Nearly all of the yellowtail catch was landed with vertical hook-and-line gear.



(a) Florida



(b) Gulf of Mexico



(c) South Atlantic

Figure 2.12: Gag grouper commercial revenue was dominated by the Gulf of Mexico (a). Across both jurisdictions, hook-and-line was the primary gear targeting gag grouper (b-c).

Commercial fleets were defined for the 'analysis' species yellowtail snapper and gag grouper, and the subsets of vessels and associated trips that comprised these fleets were validated across the MFTT and FLS databases to ensure representative fleet definitions for 2014-2016 (variable costs sampling range) and 2007-2016 (variable costs

Table 2.13: Gag grouper and yellowtail snapper fleet compositions of catch and trip within time periods (2014-2016, 2007-2016), and across datasets (FLS, MFTT). This was created to check for equality between FLS and MFTT trip statistics for the fleets defined within each time period, ensuring representative subsets.

Yellowtail Snapper

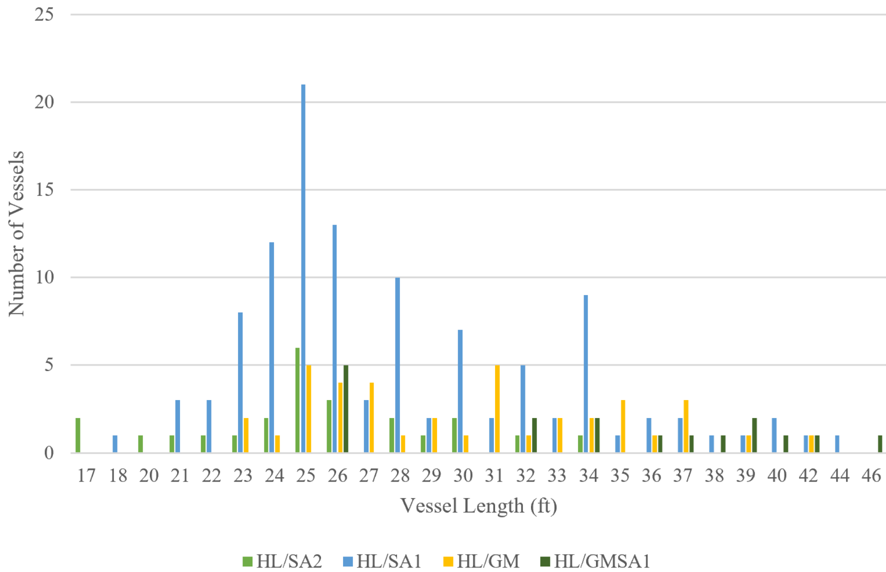
	FLS 2014-2016		MFTT 2014-2016		FLS 2007-2016		MFTT 2007-2016	
Gear	Catch	Trips	Catch	Trips	Catch	Trips	Catch	Trips
HL	99.9%	97.3%	99.9%	99.4%	99.8%	98.2%	99.7%	99.4%
LL	0.0%	1.3%	0.0%	0.0%	0.1%	0.7%	0.0%	0.0%
OT	0.0%	0.7%	0.1%	0.1%	0.0%	0.6%	0.1%	0.1%
SP	0.0%	0.7%	0.1%	0.5%	0.0%	0.5%	0.2%	0.4%
UN	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.0%

Gag Grouper

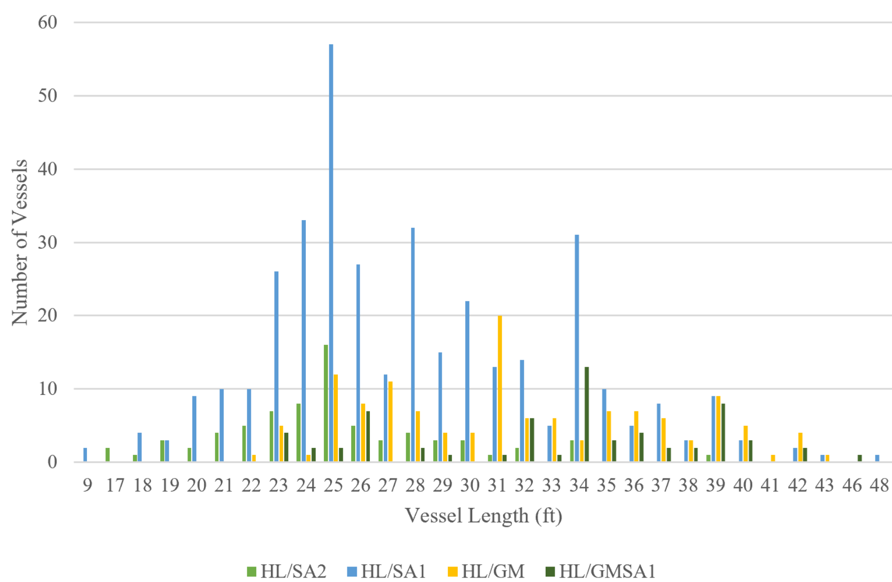
	FLS 2014-2016		MFTT 2014-2016		FLS 2007-2016		MFTT 2007-2016	
Gear	Catch	Trips	Catch	Trips	Catch	Trips	Catch	Trips
HL	51.6%	70.5%	50.8%	68.6%	59.0%	72.2%	61.9%	73.8%
LL	40.0%	17.0%	38.6%	16.6%	32.8%	16.5%	28.3%	13.8%
OT	0.1%	0.4%	0.1%	0.2%	0.4%	1.1%	0.1%	0.3%
SP	8.3%	12.1%	10.5%	14.6%	7.8%	10.3%	9.8%	12.4%
UN	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.3%

applied). For all time periods and datasets, over 99% of yellowtail snapper landings were attributed to HL gear (Table 2.13). Gag grouper catch was more dispersed between gears, but the distributions were approximately equal within defined time frames for each dataset. An emerging trend was an increase in SP and LL catch relative to the total, resulting in a decrease in the percentage of gag catch attributed to HL gear.

Vessel length was hypothesized to influence costs to fish, therefore, vessel length distributions between the FLS 2014-2016 and MFTT 2007-2016 were analyzed to ensure representative subsets of fleets from both databases. Yellowtail snapper subsets displayed a larger number of vessels within each length class for the longer time interval, but approximately equal ranges and medians for each fleet (Figure 2.13). Vessel lengths within fleets defined for yellowtail snapper and gag grouper were nearly identical between FLS 2014-2016 and MFTT 2007-2016.



(a) 2014-2016 Florida Logbook System Vessels, $n = 192$



(b) 2007-2016 Marine Fisheries Trip Ticket Vessels, $n = 635$

Figure 2.13: Yellowtail snapper vessel length distributions for the hook-and-line fleets in the Florida Logbook System 2014-2016 were approximately equal to the Marine Fisheries Trip Ticket 2007-2016.

Trip characteristics were also compared between fleet subsets within the two datasets. Variables compared included average vessel length, average number of days per trip, proportion of trips, and proportion of vessels within each fleet. These characteristics were nearly identical across all metrics for the yellowtail snapper fleets (Table 2.14). Gag grouper trip characteristics were similar, but some fleets, including LL gear, were insufficiently sampled and many vessels did not include any information on length. Overall, the fleet definitions applied to FLS and MFTT represented the same subsets of vessels targeting yellowtail snapper and gag grouper, allowing for the construction of bioeconomic functions using cost data in FLS and catch/revenue data in MFTT.

2.2.4 Florida Reef Fish Market Description

Regional ex-vessel prices for snappers and groupers were driven by statewide domestic production. For domestic Florida grouper production, the regional prices had the highest correlation with the region with the highest landings (Table 2.15). For example, high gag grouper landings in west Florida were the driving force for gag grouper prices in all regions around the state. Similar patterns were observed in black and red groupers. All grouper species produced better results when the Florida market was considered one unit (Table 2.16). Gag and black grouper prices were driven by their own landings, while red grouper, a less desirable species, had more substitute goods in the form of other groupers, so was driven by total domestic landings around the state. For domestic snapper production, adding a regional component reversed the direction of the supply-demand relationship in some instances. When treating the state as a single market, the relationship between domestic landings and

Table 2.14: Comparison of fleet characteristics used to define the cost function for the fleets (FLS 2014-2016), and the fleets this function was applied to (MFTT 2007-2016). Vessel Length and Days/Trip were the averages of vessel lengths and days per trip, respectively, weighted by number of trips. Proportion of trips ran by each fleet type and proportion of vessels within each fleet were also reported.

Fleet		**FLS (2014-16)**		
$n = 8,158$	Vessel Length	Days/Trip	p(Trips)	p(Vessels)
YT/HL/SA2	25.24	1.02	0.13	0.13
YT/HL/SA1	26.74	1.35	0.66	0.58
YT/HL/GM	30.81	3.54	0.10	0.20
YT/HL/GMSA1	30.97	1.63	0.10	0.09
Fleet		**MFTT (2007-16)**		
$n = 50,654$	Vessel Length	Days/Trip	p(Trips)	p(Vessels)
YT/HL/SA2	25.48	1.05	0.14	0.11
YT/HL/SA1	27.57	1.35	0.67	0.58
YT/HL/GM	30.43	2.81	0.07	0.21
YT/HL/GMSA1	31.60	1.77	0.11	0.10
Fleet		**FLS (2014-16)**		
$n = 4,457$	Vessel Length	Days/Trip	p(Trips)	p(Vessels)
GAG/HL/GM	32.4	4.83	0.61	0.54
GAG/LL/GM	37.0	11.1	0.14	0.11
GAG/SP/GM	31.2	1.53	0.07	0.09
GAG/HL/SA	29.9	2.85	0.11	0.19
GAG/SP/SA	26.4	1.06	0.06	0.07
Fleet		**MFTT (2007-16)**		
$n = 86,676$	Vessel Length	Days/Trip	p(Trips)	p(Vessels)
GAG/HL/GM	36.8	3.91	0.41	0.43
GAG/LL/GM	45.2	7.68	0.11	0.12
GAG/SP/GM	30.9	1.83	0.03	0.07
GAG/HL/SA	29.8	1.60	0.38	0.25
GAG/SP/SA	26.9	1.71	0.08	0.13

Table 2.15: Regional grouper price correlations were shown with the top two correlations among all regions and types of landings. Top landings in a given region drove the prices for all other regions.

Gag	Gag (lbs, W)	Groupers (lbs, W)
Price (W)	-0.72079	-0.65688
Price (S)	-0.49058	-0.5695
Price (E)	-0.63824	-0.61614
Red	Red (lbs, S)	Groupers (lbs, S)
Price (W)	-0.52576	-0.50728
Price (S)	-0.57604	-0.5438
Price (E)	-0.4439	-0.45658
Black	Black (lbs, S)	Groupers (lbs, S)
Price (W)	-0.55855	-0.65661
Price (S)	-0.67389	-0.64144
Price (E)	-0.6281	-0.56196

Table 2.16: Statewide estimates of price were improved by removing regionality and considering Florida as a single market.

	Gag (lbs)	Groupers (lbs)
Gag price	**-0.73068**	-0.58391
	Red (lbs)	Groupers (lbs)
Red price	-0.39156	**-0.55355**
	Black (lbs)	Groupers (lbs)
Black price	**-0.7021**	-0.57323

price cleaned up significantly. This finding makes intuitive sense due to Florida's narrow shape and ease with which product can be shipped between the west and east coasts. Therefore, Florida was modeled as a single market influencing ex-vessel prices of grouper and snapper domestic production.

Seafood imports into Tampa and Miami comprised a over half of the available snapper and grouper biomass in the state from 1995-2016. Total available biomass was defined here as commercial landings and imports into Florida; no information was obtained on the domestic transport of seafood. Approximately half of the total grouper biomass in Florida was imported (Figure 2.14). By 2016, snapper imports

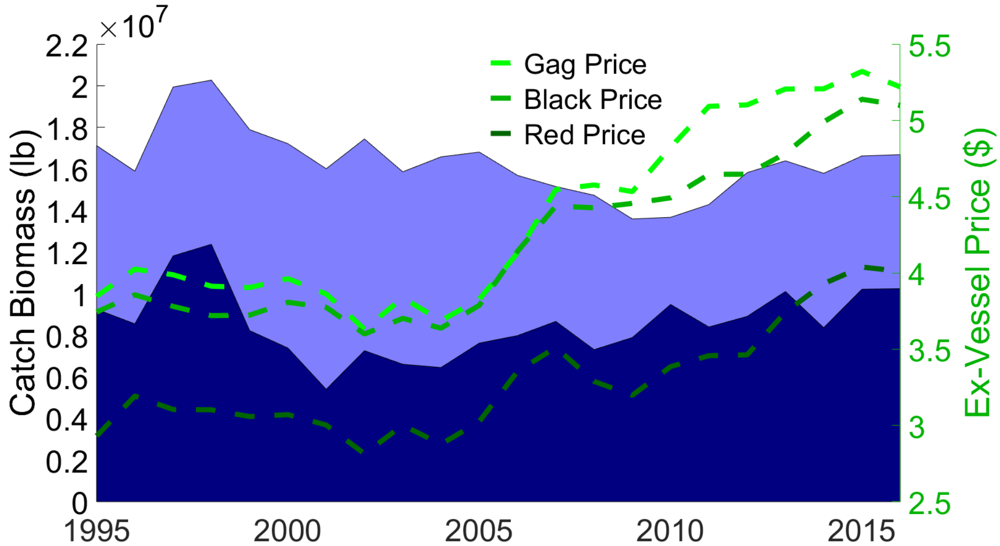


Figure 2.14: Total grouper imports (dark purple) into Florida and domestic grouper landings (light purple) from 1995-2016 with gag, black, and red grouper ex-vessel prices standardized to 2016 dollars shown.

represented 90% of the total available snapper biomass in Florida (Figure 2.15). Snapper imports in Florida exceeded grouper imports by a factor of over 2.5 throughout the 22 year period (Table 2.17). The top 4 importers into Florida of both snappers and groupers were Mexico, followed by Panama, then Brazil and Nicaragua. Grouper imports have remained fairly low and stable through time, while snapper imports have dominated the Florida seafood market.

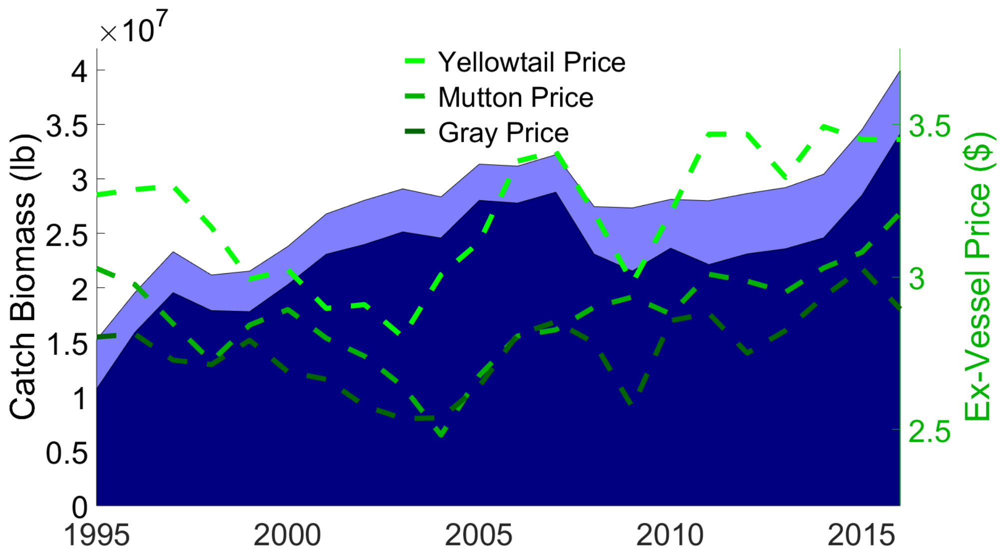


Figure 2.15: Total imports (dark purple) into Florida and domestic landings (light purple) from 1995-2016 with standardized yellowtail, gray, and mutton snapper prices shown.

Table 2.17: Top countries importing snappers and groupers into Florida sorted in descending order. All imports from countries throughout 1995-2016 were reported, and totals represented all imports of snappers and groupers into Florida from 1995-2016 from all countries, even those not listed here. Total snapper imports into Florida exceeded grouper imports by a factor of over 2.5.

Country	**Groupers (lbs)**	**Country**	**Snappers (lbs)**
Mexico	133,771,864	Mexico	102,167,128
Panama	26,305,876	Panama	93,911,805
Brazil	6,513,210	Brazil	81,296,388
Nicaragua	2,843,649	Nicaragua	75,313,148
India	2,651,082	Suriname	50,238,629
Trinidad & Tobago	2,418,730	Honduras	22,319,037
Ecuador	2,396,153	Trinidad & Tobago	18,406,890
Colombia	2,216,210	Costa Rica	17,967,117
Costa Rica	1,577,950	Guyana	12,701,133
Honduras	1,314,439	Indonesia	9,765,803
TOTAL	**188,520,515**	**TOTAL**	**507,847,872**

CHAPTER 3

Biological and Economic Parameterization

Demographics and fisheries data assimilated in the previous Chapter were used to parameterize biological and economic models (Figure 3.1).

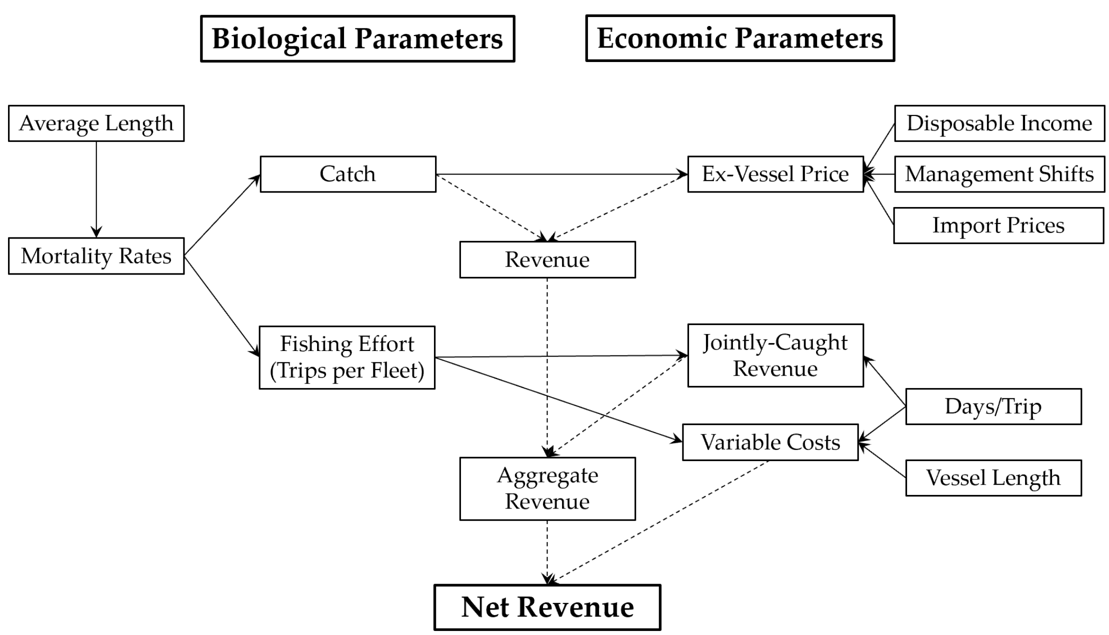


Figure 3.1: In this Chapter, mortality rates, variable costs, ex-vessel prices, jointly-caught revenue, and net revenue were estimated under the current management regime.

3.1 Methods

3.1.1 Demand Functions for Snappers and Groupers

Demand functions estimate willingness to pay for a good given the quantity available in the market. Within fisheries markets, inverse demand functions are utilized to estimate fish price variations as a function of the variations in its landings (Barten & Bettendorf, 1989). The quantity of available snappers and groupers in the market was approximately set and assumed to drive ex-vessel prices (as opposed to prices driving the quantity produced) because these natural resources have a set carrying capacity constrained by life history parameters. The MFTT database recorded landings and associated ex-vessel prices that fishermen received at the dock for every trip, allowing for an accurate representation of total monthly landings entering the market and average ex-vessel prices for all 'principal' species. Ex-vessel price, $p(t)$ was estimated within each monthly time step, t, using j explanatory variables,

$$p(t) = \beta_0 + \sum\nolimits_j \beta_j \cdot x_j(t) + \varepsilon(t) \tag{3.1}$$

where β_0 was the estimated intercept, $x_j(t)$ were explanatory variables with coefficients β_j, and $\varepsilon(t)$ was the error associated with the model.

The best model for each species was built by progressively adding explanatory variables to the model that explained the most variation in a stepwise fashion until the variables no longer significantly contributed to the model. This method was validated using a stepwise selection procedure in SAS, PROC GLMSELECT. Potential j explanatory variables tested for groupers included monthly landings of the species being analyzed, total monthly domestic landings of all groupers, total monthly grouper imports into Florida, monthly average price of grouper imports, U.S. disposable income, Florida population size, and management interventions as dichotomous variables. Management interventions tested for groupers included major regulatory changes such as the implementation of the privilege-based IFQ program for the shallow-water grouper fishery in the Gulf of Mexico in 2010, quota closures of the shallow-water grouper fishery from October-December 2000 and November-December 2005 in the Gulf of Mexico, and seasonal closures January-April 2010-2016 in the South Atlantic. Deepwater grouper closures were not tested for its effect on the ex-vessel prices of shallow water groupers due to their comparatively minimal landings. Potential j explanatory variables tested for snappers included monthly landings of the species being analyzed, total monthly domestic landings of all snappers, total monthly snapper imports into Florida, monthly average price of snapper imports, U.S. disposable income, Florida population size, and management interventions as dichotomous variables. The only management intervention tested for snappers was the closure of the mutton snapper fishery May-June in the South Atlantic since 2006 (commercial fishermen were still allowed to land the recreational bag limit, 10 snappers, the entire time period). Commercial closures of one month or less within any jurisdiction for any species were not considered.

Ex-vessel prices, $p(t)$, have been documented to respond to changes in predictor variables with an unknown lag. In other words, predictor variables such as Florida commercial landings or imported products received at the processor's level may have a delayed effect on current market conditions for fishermen. This was investigated by calculating the cross-covariance between monthly ex-vessel price, $p(t)$, individually with j explanatory variables, $x_j(t)$. Covariances, s_{p,x_j}, were calculated between predictor variables x_j and ex-vessel price where lags of k_j months up to n_t total months were examined against the price in the current month, $p(t+k_j)$.

$$s_{p,x_j}(k_j) = \frac{1}{n_t} \cdot \sum_{t=1}^{n_t-k_j} [p(t+k_j) - \bar{p}] \cdot [x_j(t) - \bar{x}_j] \tag{3.2}$$

For ease of interpretation, the cross-covariances, $s_{p,x_j}(k_j)$, were converted to cross-correlations, $r_{p,x_j}(k_j)$, using the variance of the response variable p and explanatory variable x_j with no lags, $s_{p,p}(0)$ and $s_{x_j,x_j}(0)$, respectively.

$$r_{p,x_j}(k_j) = \frac{s_{p,x_j}(k_j)}{\sqrt{s_{x_j,x_j}(0) \cdot s_{p,p}(0)}} \tag{3.3}$$

Lags, k_j, were confirmed if there was a single, significant peak of higher correlation not equivalent to zero, $k_j \neq 0$. If significant lags were found that made biological/intuitive sense, then they were applied to the regression Equation 3.1, resulting in the general form.

$$p(t) = \beta_0 + \sum\nolimits_j \beta_j \cdot x_j(t+k_j) + \varepsilon(t) \tag{3.4}$$

When there was high correlation among explanatory variables, as seen in many economic indicators, remedial measures were taken. The explanatory variables were regressed upon each other to remove the correlation among them, then the final inverse demand function was fit with the residuals of the regression between explanatory variables. The effect of this procedure was to include all of the variation from the second explanatory variable except that which was already explained by the first explanatory variable. In the case where the second significant explanatory variable was correlated with the first significant explanatory variable, the regression took the form

$$x_1(t) = b_0 + b_1 \cdot x_2(t) + \phi_2(t) \tag{3.5}$$

where $x_1(t)$ was the explanatory variable that accounted for most of the standalone variation in the ex-vessel price estimation model, $x_2(t)$ was the next most significant, albeit correlated, explanatory variable. Therefore, the residuals, defined here as $\phi_2(t)$, were incorporated in the final regression model instead of $x_2(t)$. The residuals, $\phi_2(t)$, incorporated the same variability as $x_2(t)$, minus the portion that was correlated and already explained by $x_1(t)$. When this method was applied, the estimate of the coefficient for the correlated variable did not change, but precision of the estimates for coefficients of the variables already in the model was increased. Correlations between explanatory variables were removed and incorporated into Equation 3.4, and the final inverse demand model was defined generally by the form

$$p(t) = \beta_0 + \sum\nolimits_j \beta_j \cdot x_j(t + k_j) + \beta_{(j+1)} \cdot \phi_{(j+1)}(t) + \varepsilon(t) \tag{3.6}$$

with $(j+1)$ total explanatory variables. No correlated variables displayed a significant lag, therefore the general form with the lag of k_j on month t was not applied to the model residuals, $\phi_{(j+1)}(t)$, from Equation 3.5. These methods were applied to build inverse demand functions for gag grouper, black grouper, red grouper, yellowtail snapper, mutton snapper, and gray snapper.

Quantity of landings was assumed to be a primary explanatory variable of ex-vessel price demanded, and ex-vessel price elasticity was estimated relative to the changes in the quantity (in weight) of landings. Sensitivity of ex-vessel prices demanded throughout the time frame, $\bar{p}$, was measured with respect to associated changes in the quantity of landings, $\bar{x}_q$ through elasticity (Wessels & Anderson, 1992). Elasticity, μ_q, of ex-vessel price was estimated with respect to changes in the quantity of landings as

$$\mu_q = \beta_q \cdot \frac{\bar{x}_q}{\bar{p}} \tag{3.7}$$

where β_q was the parameter associated with the quantity of significant landings estimated in Equation 3.6, $\bar{x}_q$ was the average landings 1995-2016, and $\bar{p}$ was the average ex-vessel price.

3.1.2 Mortality Rate Estimation

Length-based assessment methods require samples of length composition and demographics information to estimate population exploitation rates (Ehrhardt & Ault, 1992; Gedamke & Hoenig, 2006; Ault *et al.* , 2008; Nadon *et al.* , 2015). With representative sampling of populations throughout the assessment region, statistical catch-at-age and length-based models have been proven to yield the same estimates (Ault *et al.* , 2014).

Oldest age a_λ was an input parameter for lifespan estimators of the instantaneous rate of natural mortality M (Alagaraja, 1984; Hewitt & Hoenig, 2005). The parameters L_∞, K, and L_λ are inputs for length-based estimators of the instantaneous rate of total mortality Z (Beverton & Holt, 1957; Ehrhardt & Ault, 1992).

Mortality rates of the two 'analysis' species, gag grouper and yellowtail snapper, were estimated annually using the average length of the population above the size at first capture. Total mortality rates estimated through average length in the exploited phase of fish populations has firm groundings in modern fisheries science (Beverton & Holt, 1957; Ehrhardt & Ault, 1992; Ault *et al.* , 2008; Nadon *et al.* , 2015). Average length in the exploited phase was calculated following methodology defined by Ehrhardt & Ault (1992), an adaptation of Beverton & Holt (1957), using length composition from the TIP database and accounting for biases associated with an infinite maximum age.

Average length during year y, $\bar{L}(y)$, was defined as

$$\bar{L}(y) = \frac{Z(y) \cdot \int_{a_c(y)}^{a_\lambda} N(a,y) \cdot L(a,y)da}{Z(y) \cdot \int_{a_c(y)}^{a_\lambda} N(a,y)da} \tag{3.8}$$

where a and y referred to age and year, respectively; $a_c(y)$ was the age at first capture during year y, a_λ was the maximum age, $Z(y)$ was the total mortality rate, $N(a,y)$ was the abundance of age a fish during year y at length $L(a,y)$ (Ault *et al.* , 2008). To solve Equation 3.8, $L(a,y)$ was substituted with the annual von Bertalanffy growth equation (Equation 2.1), and $N(a,y)$ was substituted with the exponential mortality model

$$N(a + \Delta a, y + \Delta y) = N(a,y) \cdot e^{-Z(y)} \tag{3.9}$$

where $N(a + \Delta a, y + \Delta y)$ represented the progression of fish to the next age class for every time step. Integration between age at first capture $a_c(y)$ and maximum age a_λ resulted in the model

$$\left(\frac{L_\infty - L_\lambda}{L_\infty - L_c(y)}\right)^{Z(y)/K} = \frac{Z(y) \cdot [L_c(y) - \bar{L}(y)] + K \cdot [L_\infty - \bar{L}(y)]}{Z(y) \cdot [L_\lambda - \bar{L}(y)] + K \cdot [L_\infty - \bar{L}(y)]} \tag{3.10}$$

which cannot be analytically solved, so the difference was minimized via an iterative numeric algorithm to estimate $Z(y)$, the total instantaneous mortality rate of the population in the exploited phase.

All lengths below the annual legal length at first capture (L_c) were deleted to remove the biases associated with sampling before full selection to the gear and methodology. The TIP length compositions were assumed to be representative of the entire Florida commercial reef fishery and ecosystem in which they operated. The length structure in year y was dependent on the current mortality rate and the length structure in previous years. Therefore, 3- and 5-year moving averages of lengths in the exploited phase were calculated to account for inter-annual dependency of these samples while retaining observable trends. The reported year y referred to the most recent year in the moving average and was most representative of that year's mortality rate (i.e. 3-year moving average for 1986 included data 1984-1986). The exploited phase was defined as fish measuring greater than the length at first capture, L_c. The average length of the exploited phase was calculated by first computing the sum of lengths of exploited phase fish, $N(y)L(y)$, and the abundance of exploited phase fish, $N(y)$,

$$N(y) \cdot L(y) = \sum_{\eta=1}^{f(y)} L(\eta) \qquad\qquad N(y) = \sum_{\eta=1}^{f(y)} N(\eta) \tag{3.11}$$

where $f(y)$ was the total number of η trips in year y (where y encompassed years $[y-2:y]$ or $[y-4:y]$ for 3- and 5-year moving averages, respectively), $L(\eta)$ were the individual lengths of fish on trip η (where $L(\eta) \geq L_c$), and $N(\eta)$ was the number of fish sampled on trip η. The ratio of the sum of lengths of exploited phase fish over the density of exploited phase fish results in the statewide estimate of average length, $\bar{L}(y)$,

$$\bar{L}(y) = \frac{N(y) \cdot L(y)}{N(y)} \tag{3.12}$$

Sample variances of average length in year y, $s^2[\bar{L}(y)]$, were defined as

$$s^2[\bar{L}(y)] = \frac{\left(N(y) \cdot L(y) - N(y) \cdot \bar{L}(y)\right)^2}{N(y) - 1} \tag{3.13}$$

where $N(y)$ were the total number of fish sampled in year y (where y encompassed years $[y-2:y]$ or $[y-4:y]$ for 3- and 5-year moving averages, respectively). The sample variance was used to calculate the variance of average length within each year, $var[\bar{L}(y)]$,

$$var[\bar{L}(y)] = \frac{s^2[\bar{L}(y)]}{[\bar{N}(y)]^2 \cdot f(y)} \tag{3.14}$$

where $f(y)$ was the total number of trips and $\bar{N}(y)$ was the average number of fish sampled per trip in year y (where y encompassed years $[y-2:y]$ or $[y-4:y]$ for 3- and 5-year moving averages, respectively). Number of trips was used to estimate variance as opposed to number of fish because fish caught on a single trip were not considered independent units. Finally, variance of average length of the exploited phase was converted to standard error.

$$SE[\bar{L}(y)] = \sqrt{var[\bar{L}(y)]} \tag{3.15}$$

The estimates of average length and standard error were used to estimate 95% confidence intervals $\left(\bar{L}(y) - 1.96 \cdot SE[\bar{L}(y)], \bar{L}(y) + 1.96 \cdot SE[\bar{L}(y)]\right)$.

Natural mortality rate, M, was defined with 5% survivorship to the maximum age in months, a_λ,

$$M = \frac{-ln(0.05)}{a_\lambda} \tag{3.16}$$

and assumed a constant rate of decline through all age classes for all years (Alagaraja, 1984; Ault *et al.* , 1998; Nadon & Ault, 2016). Fishing mortality rate, $F(y)$, was calculated through the fundamental principle

$$F(y) = Z(y) - M \tag{3.17}$$

where $Z(y)$ was the total mortality rate and M was the natural mortality rate.

3.1.3 Jointly-Caught Revenue

Jointly-caught revenue was defined as revenue generated from any species other than the 'analysis' species and contributed to the aggregate revenue for the defined 'fleets.' Jointly-caught revenue, $R_{g\psi}$, was calculated for five species categories requiring federal permits, ψ = Reef Fish, Cobia & King/Spanish Mackerel, Dolphin/Wahoo, Tuna/Shark/Swordfish, and Other Nontarget. Monthly jointly-caught revenues were represented by $R_{g\psi}(t)$ where actual prices p were multiplied by the jointly-caught yield Y_ψ on each trip η for fleet g that fished $f_g(t)$ total trips in month t.

$$R_{g\psi}(t) = \sum_{\eta_g=1}^{f_g(t)} p_{g\psi}(\eta) \cdot Y_{g\psi}(\eta) \tag{3.18}$$

The total jointly-caught revenue calculated in Equation 3.18 was estimated using the time each fleet spent at sea over the entire month. Days fished per month, $d_g(t)$, was calculated by summing the days fished on each trip η by fleet g over month t.

$$\hat{R}_{g\psi}(t) = \beta_{g0} + \beta_{g1} \cdot \sum_{\eta_g=1}^{f_g(t)} d_{g\eta}(t) \tag{3.19}$$

$\hat{R}_{g\psi}(t)$ was the predicted jointly-caught revenue in month t, β_{g0} and β_{g1} were the estimated intercepts and coefficients, and there were $d_g(t)$ total days fished in month t across all trips $f_g(t)$. Transformations were utilized to linearize functions and normalize error, meeting the assumptions of linear regression.

3.1.4 Variable Cost Index

Variable costs to fish were influenced by fleet specific components analogous to vessel characteristics and effort. The FLS dataset included 53 variables comprising total variable costs, but many were sparsely or never recorded by fishermen. Following exploratory analyses, trip-level non-labor variable costs most reliably recorded included fuel, bait, ice, tackle, groceries, and miscellaneous expenses. Trips with missing fuel costs were excluded from analyses, because fuel was assumed a mandatory expense to operate a fishing vessel. Any other missing cost variable was assumed to be no cost for that trip. Average amount spent on each cost category was calculated and compared between fleets. Total variable costs for fleet g, c_g, were defined per trip η as

$$c_g(\eta) = \sum_{l=1}^{6} y_{gl}(\eta) \tag{3.20}$$

where y_{gl} were the fleet specific explanatory variables $l =$ fuel, bait, ice, tackle, groceries, and miscellaneous expenses, where fuel cost > 0, and $c_g(\eta)$ was total variable cost per trip. Given the high variability in trip costs and revenue within and between fleets, these co-variances were analyzed further.

An index was explored to characterize the differences in fleets' decisions to continue investing in fishing on a single trip. Because the FLS data did not include reliable information on revenue, and the MFTT data did not include cost information, weekly totals of revenue and cost were calculated for each fleet from 2014-2016 via the MFTT and FLS, respectively. The total revenue for each fleet g in week w, $R_g(w)$, was divided by the total costs in that same week, $c_g(w)$, correcting for discrepancies in total days fished per week, $d_g(w)$, reported on MFTT and FLS datasets, resulting in $\iota_g(w)$, a weekly cost index.

$$\iota_g(w) = \frac{R_g(w)/d_{g,MFTT}(w)}{c_g(w)/d_{g,FLS}(w)} \tag{3.21}$$

The weekly cost index, $\iota_g(w)$, represented the anticipated daily revenue per unit cost throughout each week w. Weekly averages of costs per trip, $\bar{c}_{g\eta}(w)$, were calculated by dividing costs, $c_g(w)$, by total number trips, $f_g(w)$, for each week (where individual trips $\eta_g \in f_g$).

$$\bar{c}_{g\eta}(w) = \frac{c_g(w)}{f_g(w)} \tag{3.22}$$

These variables were transformed to attain linearity then regressed to investigate if total average costs spent on a trip, $\bar{c}_{g\eta}(w)$, were influenced by the anticipated daily revenue in that week, $\iota_g(w)$.

$$\bar{c}_{g\eta}(w) = \beta_{g0} + \beta_{g1} \cdot \iota_g(w) + \varepsilon(w) \tag{3.23}$$

The goal of this cost index analysis was to investigate differences in relationships between anticipated revenue and willingness to spend among fleets. The slopes and intercepts of transformed costs on indices were compared between fleets using an ANalysis of COVAariance (ANCOVA), and significance levels were reported.

3.1.5 Variable Cost Function

Variable costs to target reef fishes was the amount of money spent running individual trips and did not include fixed costs (e.g. annual permit fees, insurance, slip fees, etc.) or opportunity costs. Variable costs were sampled on a subset of vessels in the FLS database and needed to be related to variables also available in the MFTT database. Vessel characteristics available in both datasets included vessel length, hull material, and year built. Effort data was recorded on both datasets, but MFTT effort had to be converted from hours to days to match the FLS format. MFTT hours were converted to days via fleet specific averages of hours per trip less than 24 hours. Total costs per trip, $\hat{c}_g(\eta)$, were estimated using the FLS data for each fleet g under the form

$$\hat{c}_g(\eta) = \beta_{g0} + \sum\nolimits_r \beta_{gr} \cdot \varphi_{gr}(\eta) \tag{3.24}$$

where $\varphi_{gr}(\eta)$ were r significant trip-level explanatory variables out of those tested including vessel length, hull material, year built, and days fished per trip; and appropriate transformations were made on the independent and dependent variables to attain linearity for regression. Linear transformations required to fit the FLS data within Equation 3.24 were assumed representative of the functional forms and were applied to estimate trip-level costs using the MFTT, $\hat{C}_g(\eta)$,

$$\hat{C}_g(\eta) = \beta_{g0} + \sum\nolimits_r \beta_{gr} \cdot \Phi_{gr}(\eta) \tag{3.25}$$

where β_{g0} and β_{gr} were parameters estimated in Equation 3.24 and $\Phi_{gr}(\eta)$ were the identical r significant explanatory variables available in the MFTT database. Total monthly variable costs, $\hat{C}_g(t)$, were calculated by summing up all $\eta \in f_g$ trips in month t for each function.

$$\hat{C}_g(t) = \sum_{\eta=1}^{f_g(t)} \hat{C}_g(\eta) \tag{3.26}$$

These were the total variable costs expended to attain revenue from both the'analysis' and jointly-caught species.

3.1.6 Net Revenue of Commercial Reef Fisheries

Net revenue for the Florida commercial reef fishery was defined here as the aggregate revenue generated from a trip minus the variable costs to fish. Aggregate revenue, $R_g(t)$, included revenue from the 'analysis' species s and jointly-caught species ψ. These revenues were calculated by multiplying the yield in weight of 'analysis' species, $Y_{gs}(\eta)$, and jointly-caught species, $Y_{g\psi}(\eta)$, with their respective ex-vessel prices, $p_{gs}(\eta)$ and $p_{g\psi}(\eta)$, for each trip $\eta \in f(t)$.

$$R_g(t) = \sum_{\eta_g=1}^{f_g(t)} p_{gs}(\eta) \cdot Y_{gs}(\eta) + \sum_{\eta_g=1}^{f_g(t)} \sum_{\psi=1}^{5} p_{g\psi}(\eta) \cdot Y_{g\psi}(\eta) \tag{3.27}$$

The aggregate revenue, $R_g(t)$, by fleet g in month t was generated from the variable costs, $C_g(t)$, estimated in Equation 3.26. Net revenue, $\Pi_g(t)$, was calculated as the difference of these values for each fleet g.

$$\Pi_g(t) = R_g(t) - C_g(t) \tag{3.28}$$

Net revenue, $\Pi_g(t)$, of the permitted vessels targeting the species of interest with the primary gears was allocated to the federal jurisdiction of the associated permits: Gulf of Mexico or South Atlantic. Parameters within Equations 3.1–3.28 are defined in Table 3.1.

Table 3.1: Variables parameterizing the biological and economic models fit within this Chapter were compiled and defined.

Variable	Definition	Units	Equation
y	Year	years	
t	Month	months	
w	Week	weeks	
a	Age	years/months	
g	Fishing fleet index		
η	Trip		
β_0	Intercepts		
β	Coefficients		
$p(t)$	Ex-vessel price at time t	2016$	3.1
$x_j(t)$	Explanatory ex-vessel price variables		3.1
s_{p,x_j}	Covariance between $p(t)$ and $x_j(t)$		3.2
k_j	Lags between $p(t)$ and $x_j(t)$	months	3.2
$s_{p,x_j}(k_j)$	Cross-covariance of $p(t)$ and $x_j(t)$		3.2
$r_{p,x_j}(k_j)$	Cross-correlation of $p(t)$ and $x_j(t)$		3.3
$\phi_{j+1}(t)$	Residuals of $x_{(j+1)}$ x_j at time t		3.5
$\bar{x}_q$	Average quantity of landings	pounds	3.7
$\bar{p}$	Average ex-vessel price	2016$	3.7
μ	Ex-vessel price elasticity		3.7
$N(y)$	Abundance in year y	fish	3.11
$L(y)$	Length of fish in year y	mm FL	3.11
$f(y)$	Total number of trips in year y	trips	3.11
$N(\eta)$	Number sampled on trip η	fish	3.11
$L(\eta)$	Length of fish sampled on trip η	mm FL	3.11
$L_c(y)$	Length at first capture in year y	mm FL	
$a_c(y)$	Age at first capture in year y	years	

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Variable	Definition	Units	Equation
$\bar{L}(y)$	Average length of the exploited phase	mm FL	3.12
$s^2[\bar{L}(y)]$	Sample variance of $\bar{L}(y)$	mm FL	3.13
$var[\bar{L}(y)]$	Annual variance of $\bar{L}(y)$	mm FL	3.14
$SE[\bar{L}(y)]$	Standard error of $\bar{L}(y)$	mm FL	3.15
$N(a, y)$	Abundance of age a fish in year y	fish	3.8
$L(a, y)$	Length of fish age a in year y	mm FL	3.8
$Z(y)$	Total mortality rate in year y		3.8
M	Natural mortality rate		3.16
$F(y)$	Fishing mortality rate in year y		3.17
$f_g(t)$	Total trips by fleet g in month t		3.18
$Y_{g\psi}(\eta)$	Yield from jointly-caught species ψ	pounds	3.18
$p_{g\psi}(\eta)$	Ex-vessel prices of jointly-caught species	2016$	3.18
$R_{g\psi}(\eta)$	Jointly-caught species ψ trip revenue	2016$	3.18
$R_{g\psi}(t)$	Jointly-caught species ψ monthly revenue	2016$	3.18
$d_g(t)$	Days fished in month t	days	3.19
$\hat{R}_{g\psi}(t)$	Estimated Monthly revenue from species ψ	2016$	3.19
$c_g(\eta)$	Total costs on trip η (FLS)	2016$	3.20
$y_{gl}(\eta)$	Variable costs	2016$	3.20
$R_g(w)$	Revenue per week	2016$	3.21
$c_g(w)$	Cost per week	2016$	3.21
$\iota(w)$	Cost index		3.21
$d_g(w)$	Days fished in week w	days	3.21
$f_g(w)$	Trips per week	2016$	3.22
$\bar{c}_{g\eta}$	Money spent per trip η	2016$	3.23
$\hat{c}_g(\eta)$	Estimated costs on trip η (FLS)	2016$	3.24
$\varphi_{gr}(\eta)$	r trip-level explanatory cost variables (FLS)		3.24
$\hat{C}_g(\eta)$	Estimated costs on trip η (MFTT)	2016$	3.25
$\Phi_{gr}(\eta)$	r trip-level explanatory cost variables (MFTT)		3.25
$\hat{C}_g(t)$	Estimated costs of fleet g in month t	2016$	3.26
$\Pi_g(t)$	Net revenue of fleet g in month t	2016$	3.28

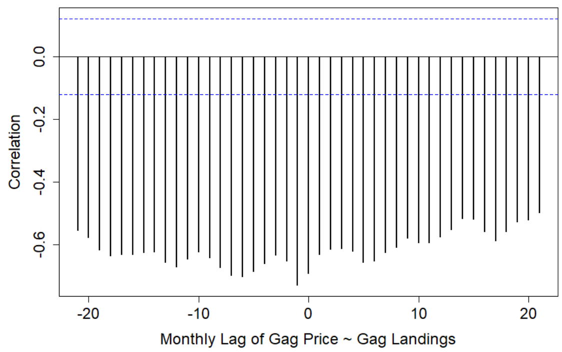
3.2 Results

3.2.1 Final Inverse Demand Functions

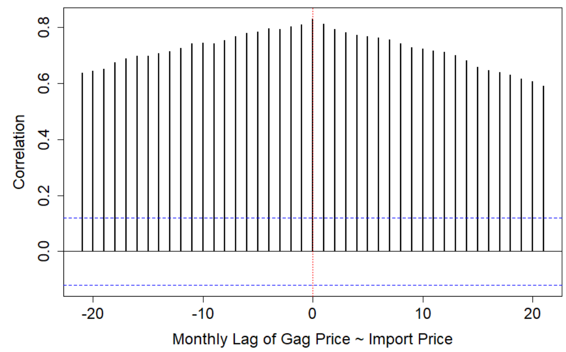
Demand models estimating willingness to pay followed similar functional forms and explanatory variables within grouper and snapper species (Table 3.2). The variable that explained the highest amount of variation for all species was their species-specific landings, except red grouper which was driven by total domestic grouper landings. Regression coefficients and R^2 are provided in Table 3.3. Ex-vessel prices of grouper species (black, gag, and red) all displayed a lagged response with monthly landings, but were most highly correlated with the import price in the current month (Figure 3.2). In other words, the domestic grouper landings led the market conditions affecting ex-vessel prices. Yellowtail snapper, the snapper species with the highest ex-vessel price, was also significantly correlated with lagged landings ($k_j = -1$), indicating it could be a market driver for snappers.

Table 3.2: Explanatory variables in the inverse demand function estimating ex-vessel prices. Significant lags, k, were noted, as well as when residuals of a correlated variable, ϕ, were used instead of the variable itself.

Species	β_0	β_1	β_2	β_3	β_4
Gag Grouper	Int	Gag (lbs, k=-1)	Imp price (ϕ)	IFQ	–
Black Grouper	Int	Black (lbs, k=-1)	Imp price (ϕ)	IFQ	–
Red Grouper	Int	Grouper (lbs, k=-1)	Imp price	IFQ	Disp inc (ϕ)
Yellowtail Snapper	Int	Yellowtail (lbs, k=-1)	Imp price	–	Disp inc (ϕ)
Mutton Snapper	Int	Mutton (lbs)	Imp price	–	Disp inc (ϕ)
Gray Snapper	Int	Gray (lbs)	Imp price	–	Disp inc (ϕ)



(a) Gag grouper landings, lag $k_j = -1$



(b) Import prices, lag $k_j = 0$

Figure 3.2: Cross-correlograms for gag grouper ex-vessel price with (a) gag grouper landings and (b) grouper import prices. A lag of $k_j = -1$ for gag grouper landings means the ex-vessel price this month is driven by landings from last month. No lag means the import price of groupers is in sync with the price of gag grouper in Florida. Red and black groupers displayed these trends as well.

Import prices of substitute goods accounted for the next most variation in all inverse demand functions (Table 3.2). Substitute goods here refers to grouper import prices for all grouper species and snapper import prices for all snapper species. Import price of groupers was highly correlated with lagged gag and black grouper landings; therefore, the residuals of import price on lagged landings was included in these models instead of import price to avoid autocorrelation in the models. These two highly valuable species' ex-vessel prices were also the only models that were not significantly affected by disposable income. High correlation of their ex-vessel prices with grouper import prices and no correlation with disposable income could be indicators for these species as market drivers unaffected by typical economic influences.

Implementation of privilege-based fishing regulations increased the ex-vessel prices of groupers between 5-10%. In 2010 when the IFQ system was implemented, gag grouper prices jumped \$0.495, black grouper prices jumped \$0.307, and red grouper prices jumped \$0.195 (Table 3.3). No other regulatory changes were influential on Florida ex-vessel prices. The final inverse demand models explained between 86% of the variation in ex-vessel price for gag grouper and 53% of the variation in mutton snapper ex-vessel price.

Table 3.3: Inverse demand function coefficients for all species where explanatory variables were defined in Table 3.2.

Species	β_0	β_1	β_2	β_3	β_4	R^2
Gag Grouper	4.682	-3.52E-06	0.299	0.495	0	0.8636
Black Grouper	4.523	-1.96E-05	0.39	0.307	0	0.8272
Red Grouper	2.885	-5.21E-07	0.252	0.195	-1.50E-04	0.7000
Yellowtail Snapper	2.298	-5.48E-06	0.694	0	-1.08E-04	0.6762
Mutton Snapper	2.709	-1.56E-05	0.201	0	-1.18E-04	0.5312
Gray Snapper	2.655	-7.55E-06	0.166	0	-8.32E-05	0.6295

These species' ex-vessel prices were relatively inelastic, but overall, species with higher value tended to have more elastic prices (with the exception of yellowtail snapper, which had the highest elasticity). The higher elasticity of yellowtail snapper ex-vessel price could be due to its restricted range in southern Florida, creating a more local market, combined with the relatively large proportion of snapper landings in all of Florida (Figure 2.9), allowing smaller percent changes in landings to have a larger impact on the percent change in price.

3.2.2 Species Mortality Rates

Average lengths of the exploited phase from the TIP database were used to estimate annual mortality rates of yellowtail snapper, Gulf of Mexico (GM) gag grouper, and South Atlantic (SA) gag grouper from 1995-2016. TIP length data were available from 1984, but data quality did not improve until the early 1990's, when annual sam-

Table 3.4: Principal Florida grouper and snapper species were sorted in descending order of mean ex-vessel price from 1995-2016. The significance of elasticities was estimated using the delta method and p-values were reported, where all were highly significant estimates.

Species	Ex-Vessel Price ($)	Elasticity	p-value
Gag Grouper	4.386	-0.113	1.11E-27
Black Grouper	4.205	-0.107	4.36E-10
Red Grouper	3.338	-0.105	2.15E-04
Yellowtail Snapper	3.287	-0.221	7.89E-19
Mutton Snapper	2.933	-0.093	4.40E-15
Gray Snapper	2.874	-0.067	6.75E-27

ple sizes more than quadrupled (Figure 3.3). Average length was calculated 1995–2016 when catch data was considered more reliable.

Annual average length of the exploited phase was dependent on both previous size structures and current mortality rates. Yellowtail snapper fit the trends observed in the catch data using a 3-year running average to estimate mortality rates (Figure 3.4), while gag grouper fit the catch data best with a 5-year running average (Figure 3.5). The longer running average required for gag could be explained by the longer lifespan and time to reach equilibrium or the greater variability in regulations, particularly size limits. The standard error of the average length estimate for yellowtail snapper was consistently low for the entire dataset, and estimates of fishing mortality rates exceeded natural mortality rates (Table 3.5). Gulf of Mexico gag grouper $\bar{L}(y)$ precision increased through time, and the estimated total mortality rate indicated that fishing mortality rate was always at least twice the rate of natural mortality (Table 3.6). South Atlantic gag grouper had the lowest number of samples in the TIP database (Figure 3.3), which resulted in the highest standard errors throughout 1995-2016 (Table 3.7). The fishing mortality rate estimates were initially less than

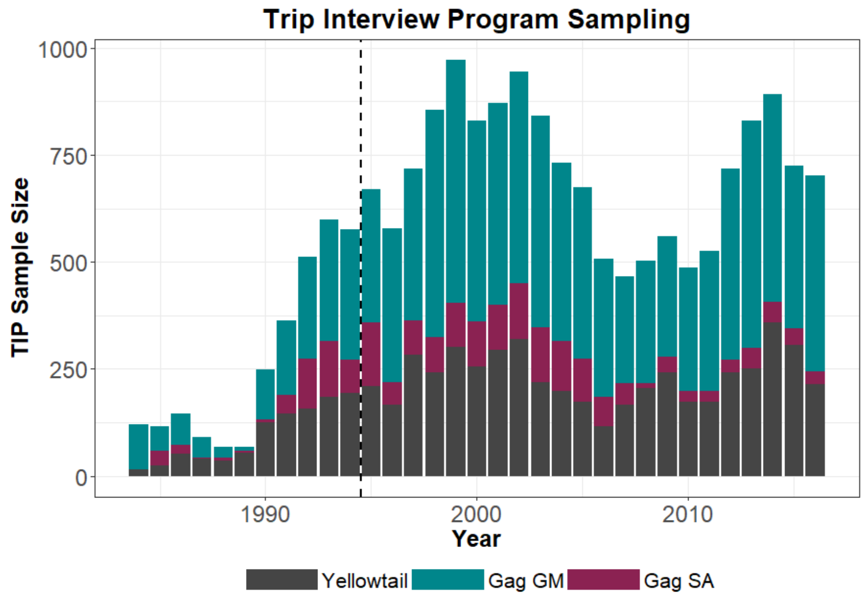


Figure 3.3: Trip Interview Program (TIP) samples sizes (i.e. number of trips) for yellowtail snapper, gag grouper in the Gulf of Mexico (GM), and gag grouper in the South Atlantic (SA). Reliable catch and effort data from the Marine Fisheries Trip Ticket (MFTT) was available in Florida since 1995, which was marked with a dashed line.

natural mortality at the start of the dataset but exceeded natural mortality rates within the first 5 years of the dataset.

3.2.3 Jointly-Caught Revenue

Jointly-caught revenue was dependent on fishing mortality rates (more specifically, fishing effort) attributed to each fleet type. Financial dependence of commercial reef fisheries on jointly-caught revenue varied by fleet type and fishing region. Over half of the South Atlantic yellowtail snapper fleets' aggregate revenue was due to yellowtail snapper alone over the 2007–2016 time period (Figure 3.6). Yellowtail snapper fleets' jointly-caught species composition was more diverse in the South Atlantic compared to the Gulf of Mexico. The South Atlantic yellowtail snapper fleets primarily rely on other reef fishes, cobia, and king & spanish mackerels. The Gulf of Mexico yellowtail

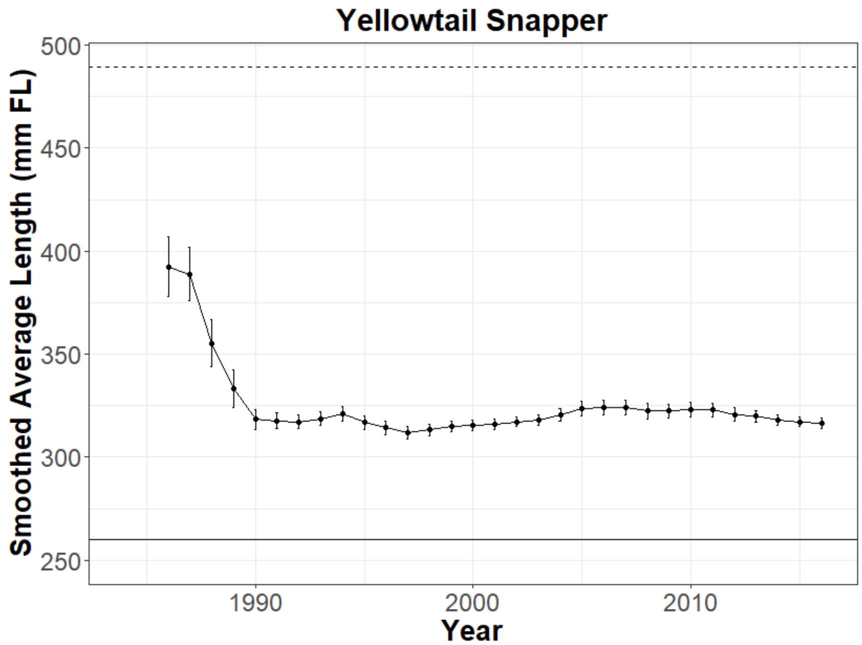


Figure 3.4: Average length of the exploited phase of yellowtail snapper with a 95% confidence interval shown for each 3 year running average from 1986-2016. With increasing sample sizes, precision of the estimates increased markedly.

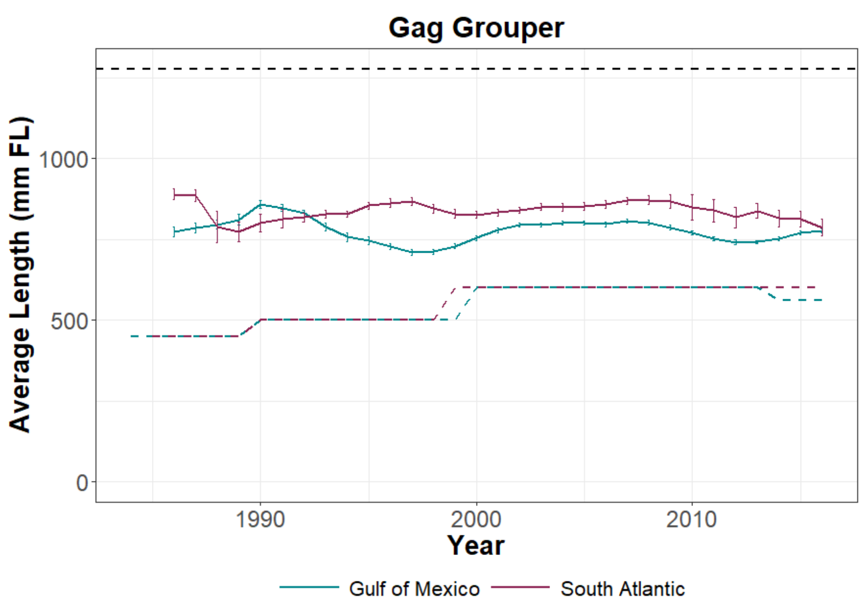


Figure 3.5: Average length of the exploited phase of gag grouper in the Gulf of Mexico and South Atlantic with a 95% confidence interval shown for each 5 year running average from 1986-2016. Overall, mortality rates were lower in the South Atlantic, resulting in larger annual average lengths in this region.

Table 3.5: Yellowtail snapper moving average length of the exploited phase, $\bar{L}(y)$, within the TIP database ($L \geq L_c$) was used to estimate the total mortality rate, $Z(y)$. Natural mortality rate, M, assumed 5% survivorship to the maximum age and was subtracted from $Z(y)$ to estimate fishing mortality rate, $F(y)$.

	Yellowtail Snapper					
Year (y)	TIP $\bar{L}(y)$	$SE[\bar{L}(y)]$	$L_c(y)$	$Z(y)$	M	$F(y)$
1995	316.77	1.61	260	0.404	0.130	0.274
1996	314.23	1.65	260	0.429	0.130	0.299
1997	311.99	1.56	260	0.454	0.130	0.323
1998	313.30	1.47	260	0.439	0.130	0.309
1999	314.94	1.35	260	0.422	0.130	0.292
2000	315.66	1.31	260	0.415	0.130	0.284
2001	316.07	1.24	260	0.411	0.130	0.280
2002	317.12	1.19	260	0.401	0.130	0.270
2003	317.86	1.27	260	0.394	0.130	0.263
2004	320.70	1.54	260	0.369	0.130	0.239
2005	323.66	1.79	260	0.345	0.130	0.215
2006	324.04	1.84	260	0.342	0.130	0.212
2007	324.31	1.83	260	0.340	0.130	0.210
2008	322.43	1.88	260	0.355	0.130	0.225
2009	322.48	1.70	260	0.354	0.130	0.224
2010	323.22	1.76	260	0.349	0.130	0.218
2011	322.87	1.71	260	0.351	0.130	0.221
2012	320.77	1.66	260	0.368	0.130	0.238
2013	319.89	1.44	260	0.376	0.130	0.245
2014	318.00	1.28	260	0.392	0.130	0.262
2015	317.08	1.17	260	0.401	0.130	0.271
2016	316.60	1.26	260	0.406	0.130	0.275

Table 3.6: Gulf of Mexico Gag Grouper moving average length of the exploited phase, $\bar{L}(y)$, within the TIP database ($L \geq L_c$) was used to estimate the total mortality rate, $Z(y)$. Natural mortality rate, M, assumed 5% survivorship to the maximum age and was subtracted from $Z(y)$ to estimate fishing mortality rate, $F(y)$.

	Gulf of Mexico Gag Grouper					
Year (y)	TIP $\bar{L}(y)$	$SE[\bar{L}(y)]$	$L_c(y)$	$Z(y)$	M	$F(y)$
1995	766.1	4.90	508.0	0.266	0.097	0.169
1996	747.1	4.29	508.0	0.298	0.097	0.201
1997	726.6	3.94	508.0	0.338	0.097	0.242
1998	722.9	3.27	508.0	0.347	0.097	0.250
1999	732.5	3.19	508.0	0.326	0.097	0.229
2000	744.5	3.25	609.6	0.531	0.097	0.434
2001	758.2	3.09	609.6	0.590	0.097	0.494
2002	775.8	2.86	609.6	0.669	0.097	0.572
2003	800.2	2.78	609.6	0.512	0.097	0.415
2004	816.9	2.21	609.6	0.436	0.097	0.339
2005	820.3	2.11	609.6	0.423	0.097	0.326
2006	830.1	1.96	609.6	0.387	0.097	0.291
2007	834.1	2.10	609.6	0.374	0.097	0.278
2008	830.9	2.25	609.6	0.385	0.097	0.288
2009	831.1	2.22	609.6	0.384	0.097	0.287
2010	825.0	2.16	609.6	0.405	0.097	0.309
2011	812.0	2.18	609.6	0.457	0.097	0.360
2012	783.3	2.54	558.8	0.295	0.097	0.199
2013	767.8	2.53	558.8	0.327	0.097	0.231
2014	761.5	2.49	558.8	0.342	0.097	0.245
2015	764.3	2.49	558.8	0.335	0.097	0.239
2016	765.1	2.37	558.8	0.334	0.097	0.237

Table 3.7: South Atlantic Gag Grouper moving average length of the exploited phase, $\bar{L}(y)$, within the TIP database ($L \geq L_c$) was used to estimate the total mortality rate, $Z(y)$. Natural mortality rate, M, assumed 5% survivorship to the maximum age and was subtracted from $Z(y)$ to estimate fishing mortality rate, $F(y)$.

	South Atlantic Gag Grouper					
Year (y)	TIP $\bar{L}(y)$	$SE[\bar{L}(y)]$	$L_c(y)$	$Z(y)$	M	$F(y)$
1995	843.9	5.11	508.0	0.170	0.097	0.073
1996	846.7	4.88	508.0	0.167	0.097	0.071
1997	855.3	4.76	508.0	0.159	0.097	0.063
1998	855.8	5.60	508.0	0.159	0.097	0.062
1999	850.6	5.79	609.6	0.237	0.097	0.140
2000	835.9	4.83	609.6	0.261	0.097	0.165
2001	834.7	4.43	609.6	0.264	0.097	0.167
2002	834.2	4.44	609.6	0.265	0.097	0.168
2003	840.6	4.37	609.6	0.253	0.097	0.157
2004	845.7	4.17	609.6	0.245	0.097	0.148
2005	852.0	4.67	609.6	0.235	0.097	0.138
2006	856.9	4.96	609.6	0.227	0.097	0.130
2007	858.3	5.21	609.6	0.225	0.097	0.128
2008	860.0	5.47	609.6	0.222	0.097	0.126
2009	869.6	5.85	609.6	0.209	0.097	0.112
2010	868.2	7.72	609.6	0.211	0.097	0.114
2011	852.5	10.68	609.6	0.234	0.097	0.137
2012	837.7	13.48	609.6	0.258	0.097	0.162
2013	846.8	10.88	609.6	0.243	0.097	0.146
2014	819.0	10.90	609.6	0.294	0.097	0.197
2015	817.5	9.63	609.6	0.297	0.097	0.200
2016	816.1	9.69	609.6	0.300	0.097	0.203

snapper fleets were always heavily reliant upon other reef fishes covered under their federal Reef Fish permit.

Gag grouper fleets relied almost exclusively upon jointly-caught revenue sources. Due to closed seasons, groupers have been unavailable for capture for a portion of the year since 1999 in the South Atlantic and from 2000-2009 in the Gulf of Mexico. The majority of year-round revenue sources from gag grouper commercial fleets were jointly-caught reef fishes (Figure 3.6). As seen in yellowtail snapper fleets, South Atlantic gag grouper fleets were characterized by more diverse revenue sources. The majority of both fleets relied upon reef fishes as their primary revenue sources.

The yellowtail snapper fleets' total revenue from reef fishes 2007–2016 was dominated by yellowtail snapper, with greater amberjack and gray snapper trailing far behind in the South Atlantic, and red grouper and red snapper also significant contributors in the Gulf of Mexico (Figure 3.7). In the South Atlantic, vermilion snapper generated more revenue than gag grouper, and in the Gulf of Mexico, red grouper, red snapper, and vermilion snapper generated more revenue than gag grouper (Figure 3.7). The Gulf of Mexico gag grouper fisheries generated the most reef fish revenue by far, with red grouper landings alone exceeding $150 million over this 10 year period.

Jointly-caught revenue sources were estimated relative to total number of days fished per month for each fleet. These data were distinct for each fleet, and patterns were particularly discernible for jointly-caught reef fishes between regions and gear types (Figure 3.8). Appropriate transformations were made to linearize data, and estimated coefficients for jointly-caught revenue from ψ = Reef Fishes (REEF), Cobia & King/Spanish Mackerel (CKSM), and Other (OTHR) were reported for all yellowtail snapper fleets and all gag grouper fleets in Tables 3.8 & 3.9. Regressions

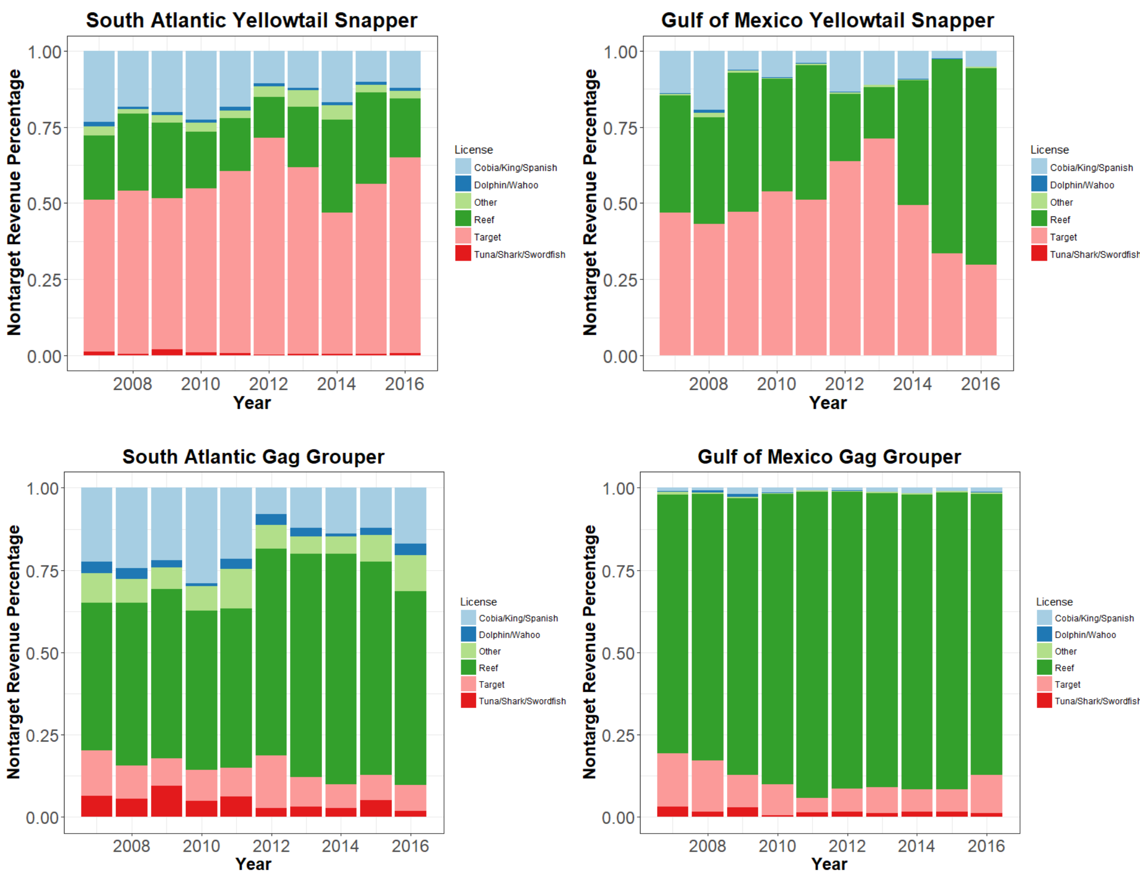
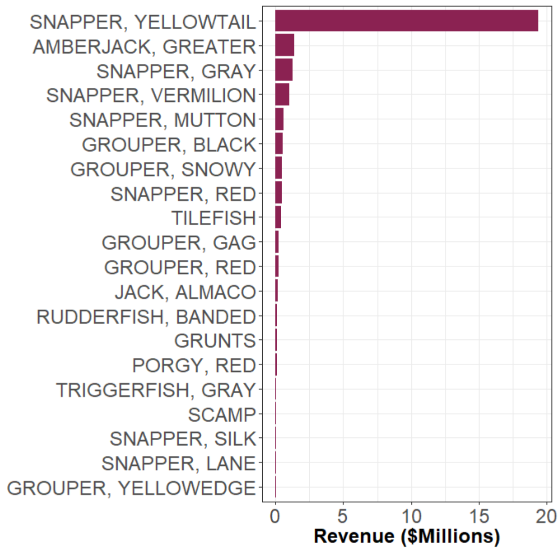
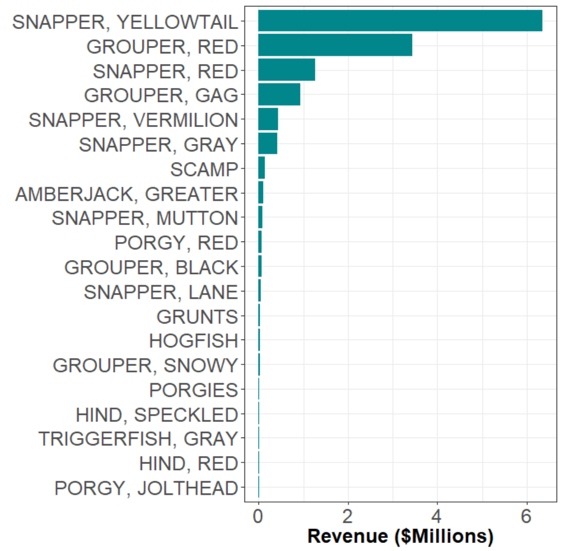


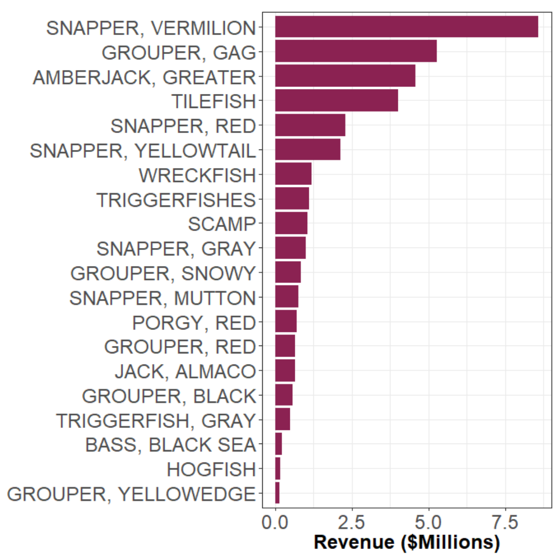
Figure 3.6: Annual revenue from jointly-caught species were grouped into categories based on required federal commercial permits: Cobia/King/Spanish (Coastal Migratory Pelagic, King Mackerel), Dolphin/Wahoo (Atlantic Dolphin-Wahoo), Reef (Reef Fish and Snapper-Grouper), Tuna/Shark/Swordfish (Highly Migratory Species, Swordfish, and Atlantic Tuna), and Other.



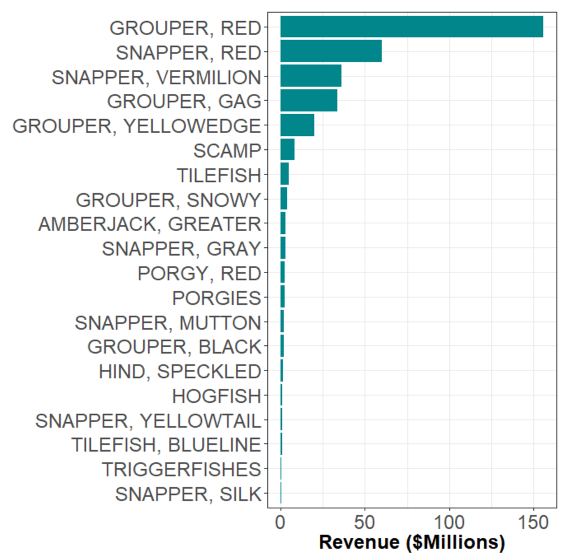
(a) **Yellowtail Snapper, South Atlantic**



(b) **Yellowtail Snapper, Gulf of Mexico**



(c) **Gag Grouper, South Atlantic**



(d) **Gag Grouper, Gulf of Mexico**

Figure 3.7: Species composition of total reef fish revenue sources from 2007–2016. *Top row:* Yellowtail snapper fleets' revenue was dominated by yellowtail snapper in both the South Atlantic and the Gulf of Mexico. *Bottom row:* Gag grouper fleets relied much more heavily upon other reef fishes within the South Atlantic and Gulf of Mexico, namely red grouper, red snapper, and vermilion snapper.

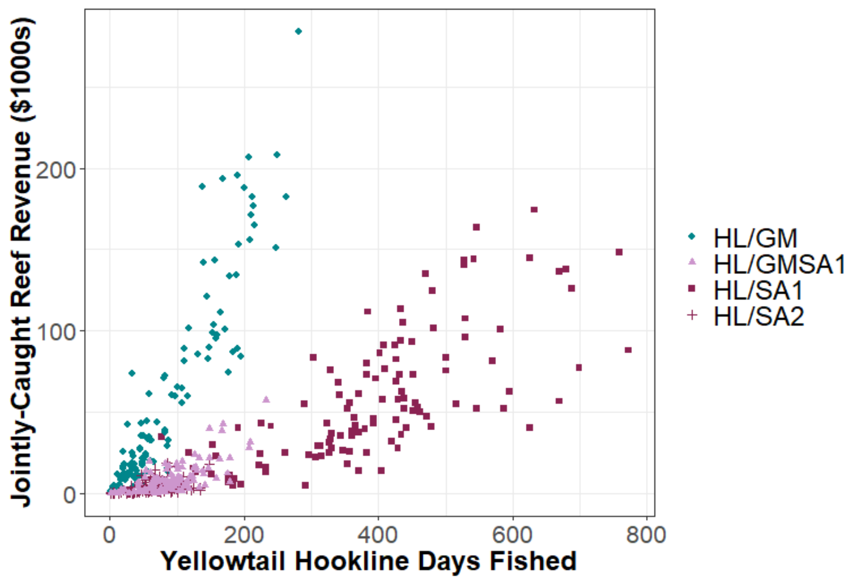
Table 3.8: Yellowtail snapper jointly-caught revenue coefficients were estimated by fleet for each species group ψ = Reef Fishes (REEF), Cobia & King/Spanish Mackerel (CKSM), and Other Nontarget (OTHR). In these regressions, Dolphin/Wahoo and Tuna/Shark were included in the OTHR category. Goodness of fit, R^2 was reported, and transformations required to attain linearity and normality were reported for the nontarget revenue, ψ, and the number of days fished d.

Yellowtail Snapper						
	Fleet	β_0	β_1	R^2	ψ	d
REEF	HL/SA1	7.45	0.17	0.609	Log	Sqrt
	HL/GM	-43.88	29.76	0.861	Sqrt	Sqrt
	HL/GMSA1	4.87	0.39	0.555	Log	Sqrt
	HL/SA2	4.34	0.42	0.458	Log	Sqrt
CKSM	HL/SA1	6.47	0.21	0.500	Log	Sqrt
	HL/GM	3.98	0.49	0.188	Log	Sqrt
	HL/GMSA1	2.82	0.62	0.228	Log	Sqrt
	HL/SA2	5.68	0.45	0.587	Log	Sqrt
OTHR	HL/SA1	7.25	0.11	0.390	Log	Sqrt
	HL/GM	3.29	0.40	0.247	Log	Sqrt
	HL/GMSA1	3.39	0.49	0.533	Log	Sqrt
	HL/SA2	5.03	0.29	0.291	Log	Sqrt

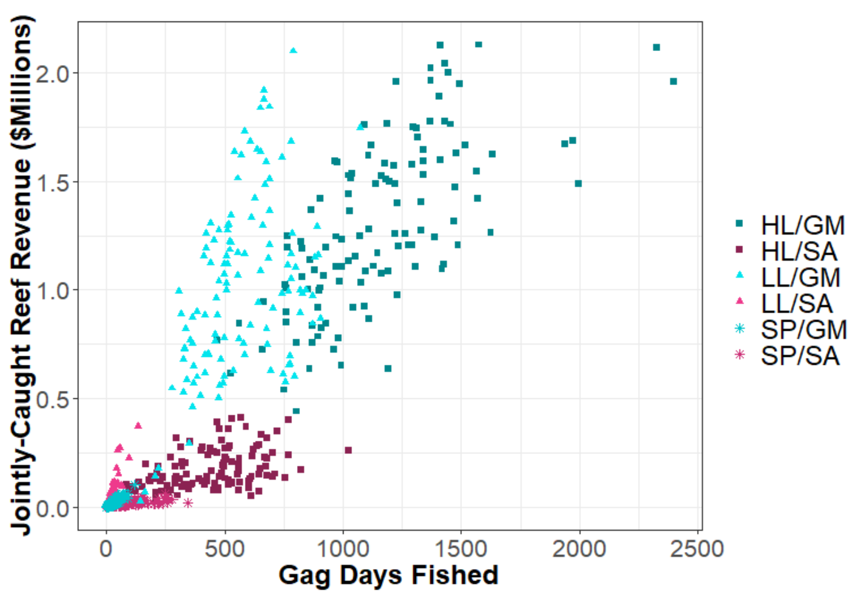
estimating jointly-caught reef fish revenue were plotted with the transformed data in Figure 3.9.

3.2.4 Variable Costs of Fleets Targeting Reef Fishes

Daily revenue earned per unit cost spent was investigated with its effect on total costs per trip, and for all fleets, there was a diminishing return to continue fishing on a single trip. In other words, if a fisherman decided to spend money to continue fishing on a single trip, the approximated daily profitability index decreased. The daily anticipated revenue/cost indices were transformed to attain linearity for parametric ANCOVA. Yellowtail snapper indices were significant for all fleets in estimating total cost per trip and displayed significant interactions between fleet and total cost for all

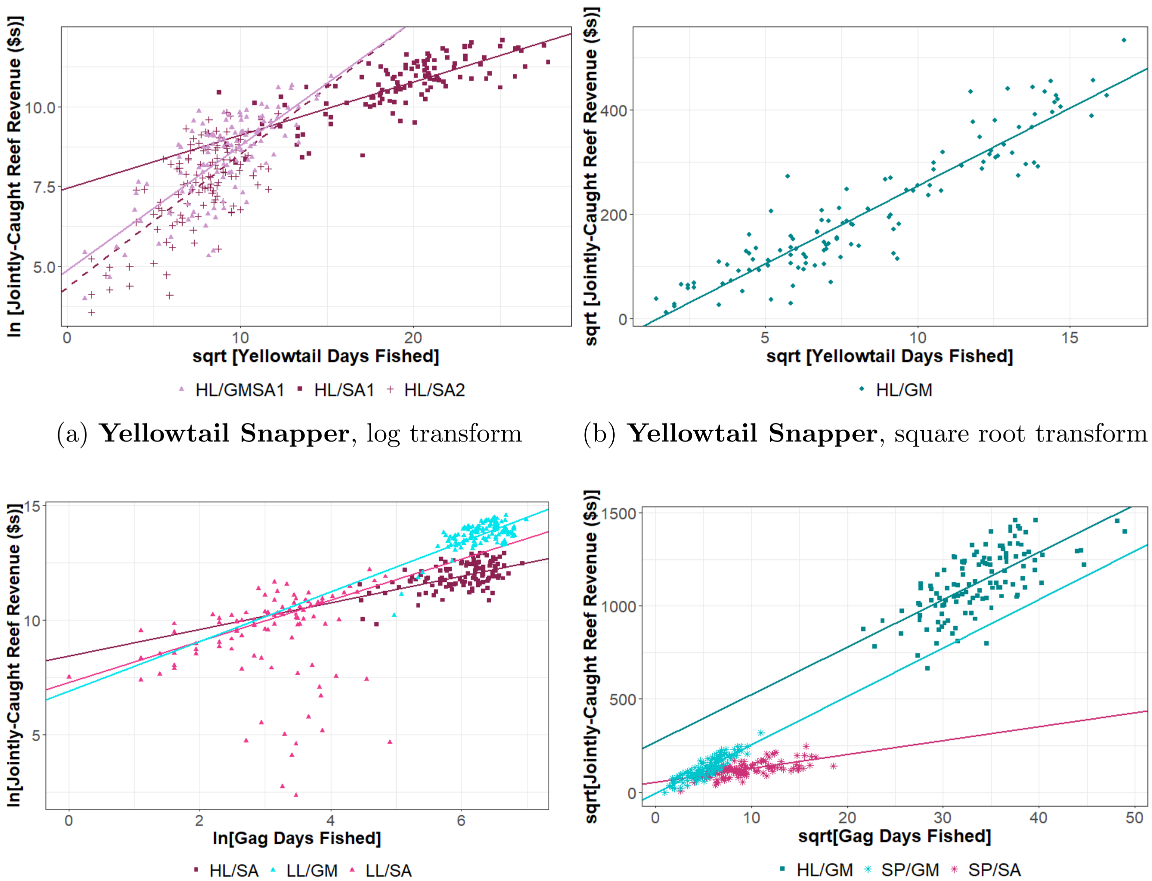


(a) **Yellowtail Snapper Fleets**



(b) **Gag Grouper Fleets**

Figure 3.8: Yellowtail snapper and gag grouper fleets' jointly-caught reef fish revenue was clustered by region and gear type over days fished, indicating different catch rates.



(a) **Yellowtail Snapper**, log transform

(b) **Yellowtail Snapper**, square root transform

(c) **Gag Grouper**, log transform

(d) **Gag Grouper**, square root transform

Figure 3.9: Jointly-caught reef fish revenue was predicted by days fished. The linearly transformed data were shown with the linear regression.

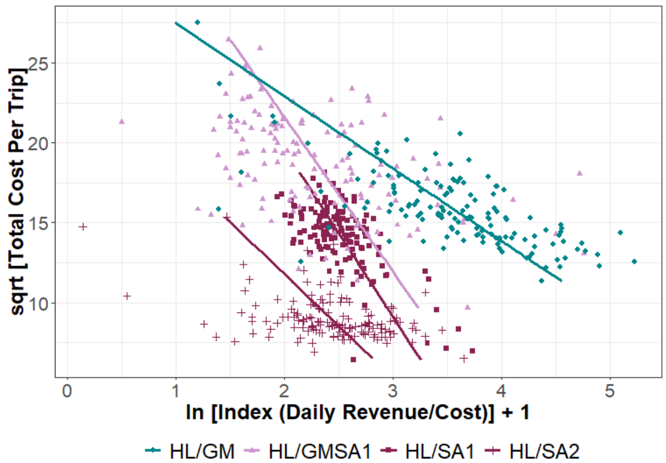
Table 3.9: Gag grouper jointly-caught revenue coefficients were estimated by fleet for each species group ψ = Reef Fishes (REEF), Cobia & King/Spanish Mackerel (CKSM), and Other Nontarget (OTHR). In these regressions, Dolphin/Wahoo and Tuna/Shark were included in the OTHR category. Goodness of fit, R^2 was reported, and transformations required to attain linearity and normality were reported for the nontarget revenue, ψ, and the number of days fished d.

Gag Grouper						
	Fleet	β_0	β_1	R^2	ψ	d
REEF	HL/GM	269.81	25.46	0.463	Sqrt	Sqrt
	LL/GM	6.89	1.09	0.431	Log	Log
	SP/GM	-6.68	26.03	0.778	Sqrt	Sqrt
	HL/SA	8.45	0.58	0.280	Log	Log
	LL/SA	7.27	0.90	0.113	Log	Log
	SP/SA	53.21	7.47	0.379	Sqrt	Sqrt
CKSM	HL/GM	4.42	0.73	0.010	Log	Log
	LL/GM	-4.32	1.82	0.240	Log	Log
	SP/GM	4.28	0.07	-0.016	Log	Log
	HL/SA	-603.86	143.08	0.439	Sqrt	Log
	LL/SA	4.35	0.18	-0.001	Log	Log
	SP/SA	27.32	0.89	0.031	Sqrt	Sqrt
OTHR	HL/GM	1.70	1.01	0.171	Log	Log
	LL/GM	4.54	0.84	0.015	Log	Log
	SP/GM	4.66	0.37	0.153	Log	Sqrt
	HL/SA	7.60	0.09	0.392	Log	Sqrt
	LL/SA	52.09	17.74	0.155	Sqrt	Sqrt
	SP/SA	7.32	0.24	0.526	Log	Sqrt

fleets except HL/SA2, which was nearly significant (Figure 3.10). Yellowtail snapper Gulf of Mexico fleets had a higher daily revenue/cost index compared to the South Atlantic, and the dual license holders displayed trends characteristic of both regions. Gag grouper indices were all highly significant in estimating total cost per trip and did not display any interaction among fleets and total costs (Figure 3.11). Gag grouper fleets clustered by gear type as well, with longline gears appearing the most profitable per unit of cost. hook-and-line gears across yellowtail snapper and gag grouper fleets were also analyzed, with significance of the index and interaction with cost influencing some of the fleets (Figure 3.12). The only fleet with a cost index not significantly different from the Gulf of Mexico gag reef fish hook-and-line fleet was the Gulf of Mexico yellowtail snapper hook-and-line fleet. This was likely due to the overlap in vessels targeting gag grouper and yellowtail snapper with hook-and-line gear in the Gulf of Mexico. Fleets from the same jurisdiction, despite targeting different species, displayed similar trends. All fleets appear to be appropriately defined allowing for identification of differences in expected profits and total costs to target the species of interest.

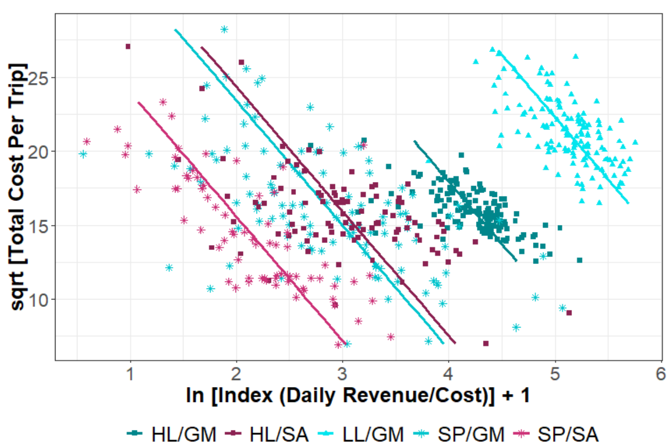
3.2.5 Final Variable Cost Functions

Commercial reef fishery fleets were defined by region, license type, and gear. These fleets spent, on average, different amounts towards each of the six variable costs sampled on FLS: bait, ice, miscellaneous, tackle, grocery, and fuel (Table 3.10). Yellowtail snapper fleets were all operating hook-and-line gear, but financial expenditures towards each category still varied between fleets based on trip characteristics. Fleets with longer trips required more funds towards groceries and fuel, and the HL/SA2



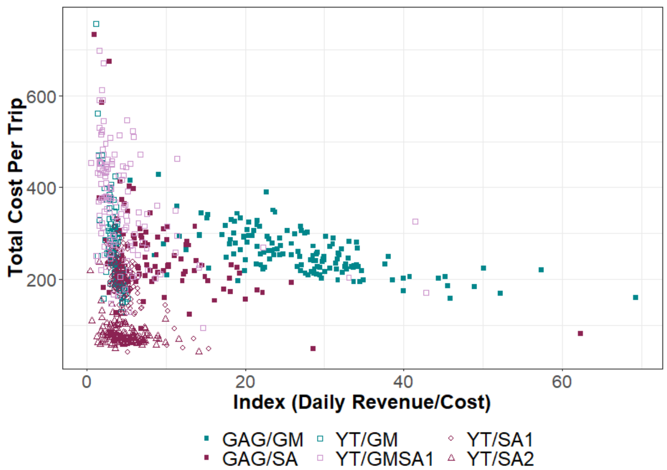
Fleet	Yellowtail
Int (HL/GM)	7.059***
HL/GMSA1	-2.820***
HL/SA1	-3.190***
HL/SA2	-3.244***
Cost	-0.221***
HL/GMSA1·Cost	0.117***
HL/SA1·Cost	0.125***
HL/SA2·Cost	0.067•

Figure 3.10: ANCOVA results for yellowtail hook-and-line gears showing the significant interactions between fleets' total trip cost and the index, daily revenue/cost, explained 61.16% of the variation in the model and significance levels were noted ($p < 0.0001$*** , $p < 0.001$** , $p < 0.05$* , $p < 0.10$•). These lines were plotted against the linearly transformed data. Int and Cost refer to the HL/GM fleet's intercept and slope, respectively, and all other fleets' estimates are added to these values to obtain their intercepts and slopes, respectively.



Fleet	Gag
Int (HL/GM)	6.134***
LL/GM	1.518***
SP/GM	-1.350***
HL/SA	-1.239***
LL/SA	-2.279***
Cost	-0.119***

Figure 3.11: ANCOVA results for gag fleets' total trip cost and the index daily revenue/cost where significance levels ($p < 0.0001$***) were shown ($R^2 = 0.8530$). These lines were plotted against the linearly transformed data. Int and Cost refer to the HL/GM fleet's intercept and slope, respectively, and all other fleets' intercepts were obtained by adding their value to the reported intercept. The interaction between fleet and cost was not significant; therefore, all gag fleet slopes were identical.



Fleet	hook-and-line
Int (GAG/GM)	6.855***
GAG/SA	-1.721***
YT/GM	0.204
YT/GMSA1	-2.616***
YT/SA1	-2.987***
YT/SA2	-3.040***
Cost	-0.164***
(GAG/SA)·Cost	0.0230
(YT/GM)·Cost	-0.0567•
(YT/GMSA1)·Cost	0.0603***
(YT/SA1)·Cost	0.0686***
(YT/SA2)·Cost	0.0102

Figure 3.12: ANCOVA results for hook-and-line fleets showing the significant interactions between fleets' total trip cost and the index of daily revenue/cost where significance levels ($p < 0.001$***, $p < 0.01$**, $p < 0.05$*, $p < 0.10^{\bullet}$) were shown ($R^2 = 0.7403$).

fleet with a trip catch limit had the lowest spending in all categories. Gag grouper fleets were primarily segregated by gears, and spearfishing fleets spent less than $20 on bait, while longline fleets spent over $1000 on bait (Table 3.10).

Costs to target each species of interest were defined based on fleet characteristics available in both the FLS and MFTT databases. Significant predictors of cost included vessel length and days per trip. Transformations required to linearize the cost functions per fleet were defined in Table 3.11, and the estimated coefficients from Equation 3.24 were compiled in Table 3.12. Most functions obtained a good fit, with R^2 values ranging from 0.653-0.693, but the HL/SA2 fleet had a poor fit, with only 4.1% of the variation explained by the model. Days fished was not significant in predicting cost for the HL/SA2 fleet because this variable had little to no contrast for this fleet, as most of their trips were only 1-2 days. The models for gag grouper fleets ranged from explaining 19.0%-75.1% of the variation in costs to fish, with HL

Table 3.10: Average variable costs per trip for each fleet. These six economic variables ($y_{gl}(\eta)$) were summed up for each trip η in Equation 3.20 to represent total cost, where all values are in 2016 dollars.

Fleet	Bait	Ice	Misc	Tackle	Grocery	Fuel	Total
YT/HL/SA2	24.21	10.51	3.28	0.02	12.17	36.28	86.47
YT/HL/SA1	135.94	19.36	19.17	1.34	21.48	93.14	290.44
YT/HL/GM	192.27	63.80	51.15	0.00	167.45	370.62	845.29
YT/HL/GMSA1	238.75	55.79	57.73	0.89	47.35	195.95	596.46
GAG/HL/GM	259.37	112.14	141.00	2.00	233.00	479.75	1227.68
GAG/LL/GM	1315.38	457.83	809.00	9.00	988.00	1444.87	5023.30
GAG/SP/GM	14.05	26.92	70.00	1.00	49.00	251.60	412.53
GAG/HL/SA	141.93	64.30	70.00	1.00	112.00	299.97	689.41
GAG/SP/SA	17.67	20.65	59.00	4.00	26.00	142.86	270.88

gears obtaining the best fits. Vessel length was not a significant predictor of cost to fish for LL fleets because length was rarely available in the FLS LL fleet.

3.2.6 Net Revenue of Commercial Reef Fisheries

Commercial annual revenue from yellowtail snapper exceeded \$7,000,000 in 2016, but just over half of this revenue was allocated to a vessel allowing for modeling here due to incomplete vessel information within MFTT (Figure 2.11). The only licenses that were held in conjunction of any significance were the Gulf of Mexico Reef Fish permit (GM) and the South Atlantic Snapper Grouper Unlimited Trip Limit permit (SA1). For these trips, the revenue and cost were added to the jurisdiction where the trip occurred (Figure 3.13). Net revenue in the South Atlantic exceeded the Gulf of Mexico, until recently when Gulf of Mexico fleets likely began reaping the benefits of the Dry Tortugas reserve (Figure 3.15).

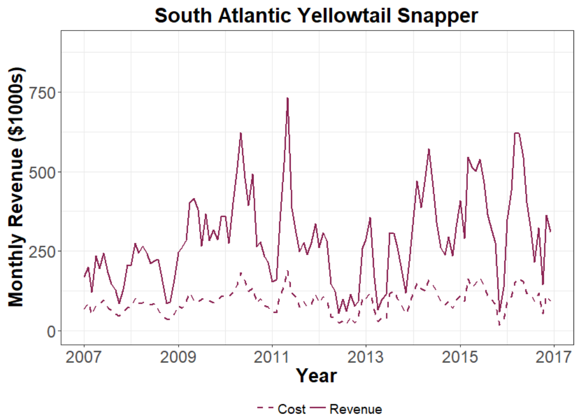
Commercial annual revenue from gag grouper peaked at nearly \$12,000,000 in 2001, but has decreased to as low as just over \$2,000,000 in 2011 (Figure 2.12).

Table 3.11: Transformed variables for predicting variable costs for the various single gear fleets where Days refers to days per trip, and Length refers to vessel length.

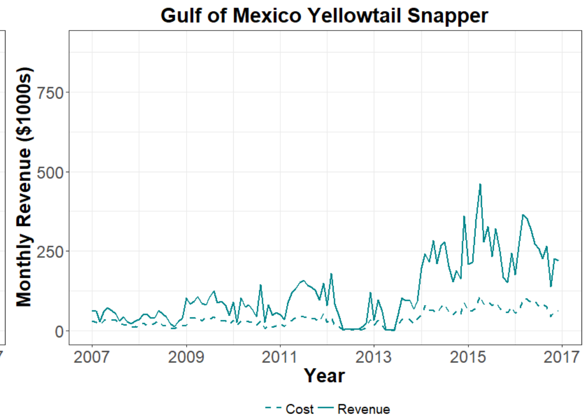
Fleet	y	β_0	β_1	β_2
YT/HL/SA2	sqrt(cost)	Int	–	ln(Length)
YT/HL/SA1	sqrt(cost)	Int	ln(Days)	Length
YT/HL/GM	sqrt(cost)	Int	ln(Days)	Length
YT/HL/GMSA1	sqrt(cost)	Int	ln(Days)	–
GAG/HL/GM	sqrt(cost)	Int	ln(Days)	Length
GAG/LL/GM	log(cost)	Int	sqrt(Days)	–
GAG/SP/GM	sqrt(cost)	Int	ln(Days)	Length
GAG/SA/HL	sqrt(cost)	Int	sqrt(Days)	Length
GAG/SA/SP	sqrt(cost)	Int	–	Length

Table 3.12: Estimated coefficients for predicting variable costs for the various single gear fleets. Required transformations for these functions were reported in Table 3.11

Fleet	β_0	β_1	β_2	R^2
YT/HL/SA2	24.7	0	-4.89	0.042
YT/HL/SA1	0.857	14.7	0.419	0.656
YT/HL/GM	7.66	8.46	0.386	0.653
YT/HL/GMSA1	18.4	15.9	0	0.693
GAG/HL/GM	7.80	6.91	0.34	0.553
GAG/LL/GM	6.96	0.44	0	0.265
GAG/SP/GM	9.57	4.00	0.26	0.190
GAG/HL/SA	-12.93	14.04	0.42	0.751
GAG/SP/SA	-12.34	0	1.08	0.359



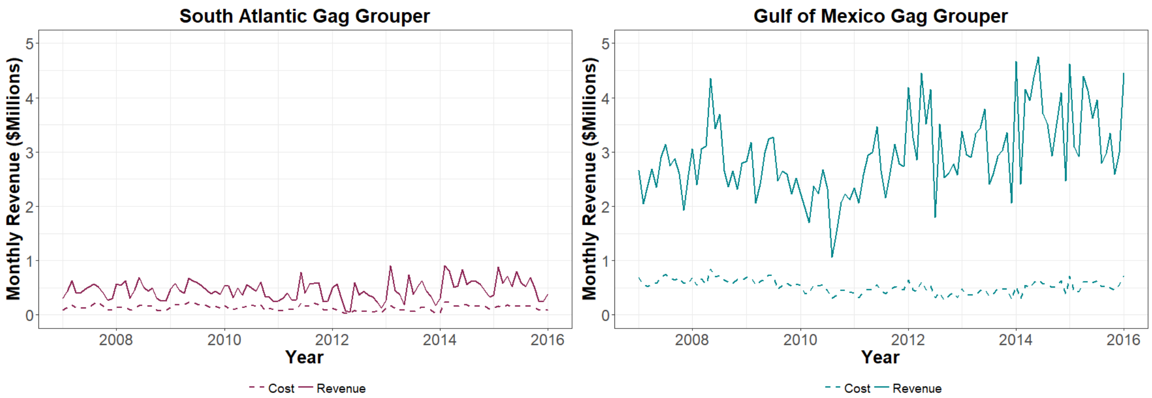
(a) South Atlantic



(b) Gulf of Mexico

Figure 3.13: Yellowtail snapper aggregate revenue and costs from actively permitted vessels in the South Atlantic and Gulf of Mexico for single gear hook-and-line line trips by vessels that landed at least one yellowtail snapper in the time frame.

Monthly revenue and costs in the Gulf of Mexico exceeded that observed in the South Atlantic throughout the entire time period (Figure 3.14). Net revenue from gag grouper fleets in the Gulf of Mexico reached over $4,000,000 in some months (Figure 3.16). The parameter estimates from this Chapter were used as inputs within an age-structured bioeconomic simulation model and validated using observed data in the following Chapter.



(a) South Atlantic

(b) Gulf of Mexico

Figure 3.14: Gag grouper aggregate revenue and costs from actively permitted vessels in the South Atlantic and Gulf of Mexico for all single gear trips fishing hook-and-line, longline, or spearfishing gears.

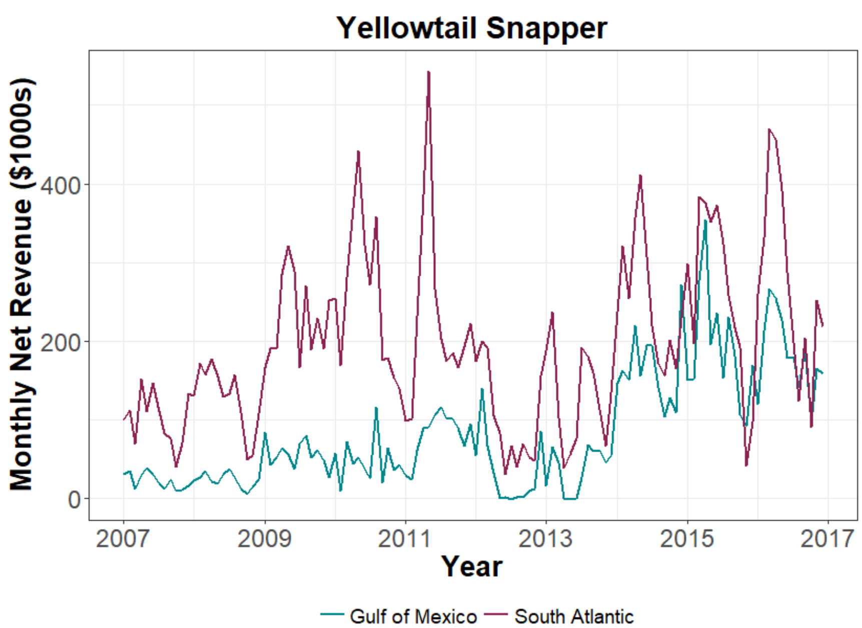


Figure 3.15: Net revenue generated by yellowtail snapper from vertical hook-and-line gears in South Atlantic and Gulf of Mexico from all species landed on directed trips.

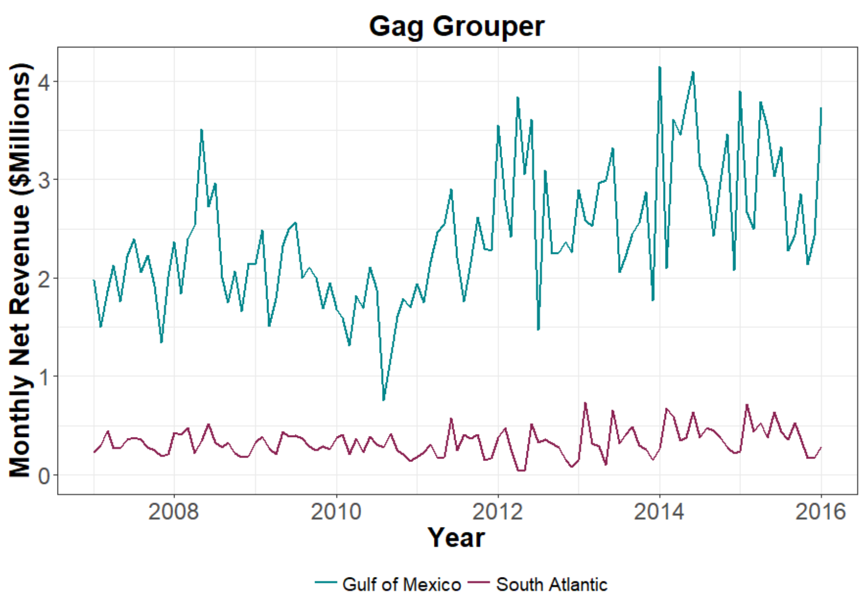


Figure 3.16: Net revenue generated by gag grouper fleets in South Atlantic and Gulf of Mexico from all species landed on directed trips.

CHAPTER 4

Bioeconomic Simulation of Florida's Commercial Reef Fish Fleets

Biological and economic functions defined in the previous Chapter were integrated and validated, resulting in a dynamic model with the flexibility to simulate regulations and observe bioeconomic impacts (Figure 4.1).

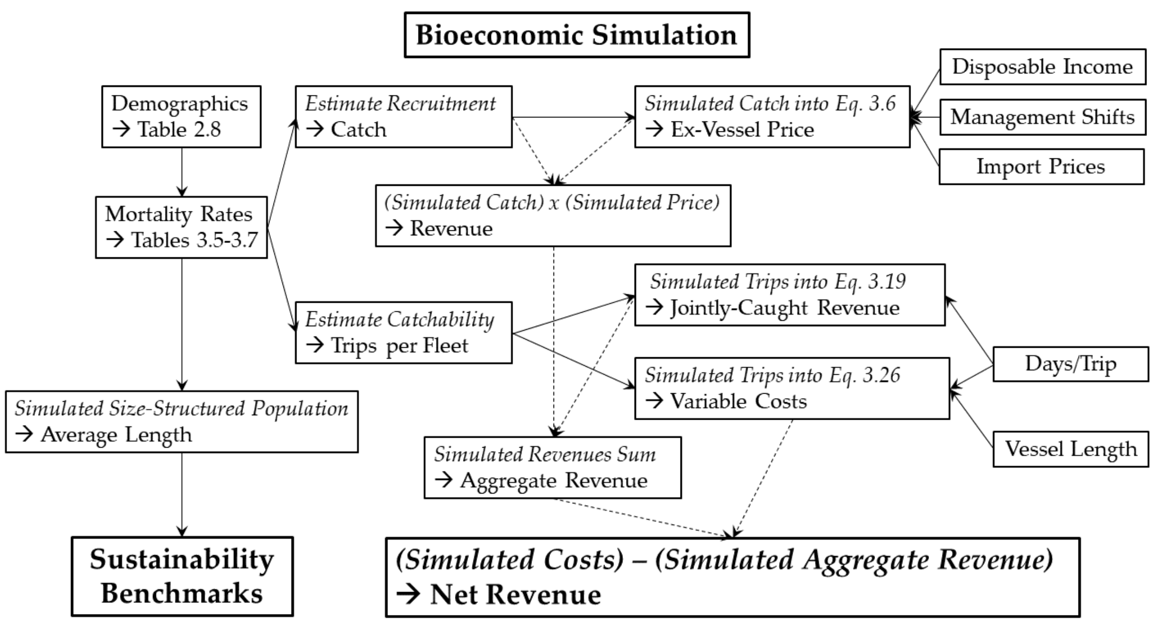


Figure 4.1: Demographics and mortality rates were used as inputs in a numerical cohort model. Italicized text represents what was estimated or simulated in this model, and text preceded by an arrow shows what was produced and validated using observed data.

4.1 Methods

4.1.1 Assessment of Current Status of Reef Fishes

Age-structured production models were parameterized by species-specific demographics information described in Chapter 2 and used as inputs into Equations 2.1–2.2 to define growth. Annual mortality rate estimates reported in Tables 3.5–3.7 were converted to monthly rates and used to describe exponential population decline. Mortality rates in months t within year y were assumed constant for fish in the exploited phase, $Z(a,t) \equiv Z(a,t)|a(t) \geq a_c(t), t \in y$, allowing for annual changes in fishing mortality rates and minimum size limits. Error structure of these estimates were incorporated in the simulation models through uncertainty in average length, $\bar{L}(y)$, estimated in Equation 3.8 and subsequently total mortality rates, $Z(y)$, where $Z(y) = M + F(y)$, the sum of natural and fishing mortality rates. The error was modeled by randomly sampling from the normal distribution $\bar{L}(y) \sim N(\bar{L}(y), 1.96 \cdot SE[\bar{L}(y)])$ (following the notation $X \sim N(\mu, \sigma)$) which was used to estimate total mortality, $Z(y)$, and a 95% confidence interval around all estimates by running the model described below in its entirety 1000 times.

Population abundances, $N(a,t)$, were defined for each species, monthly time step t, and incoming monthly cohort a. Recruitment, $N(0,t)$, was assumed to be constant for each incoming monthly cohort. The initial population structure, $N(a,0)$ was first defined under no fishing mortality, $F(a,0) = 0$, where

$$N(a + \Delta a, 0) = N(a, 0) \cdot e^{-M(a)} \tag{4.1}$$

represented the initial cohort decline, and age steps Δa were one month. This was defined in order to simulate population decline from an unfished state through to the population structure that would produce the average catch for each species in the first month t of available data by adjusting recruitment, $N(0,t)$. Population declines were adapted from the exponential mortality model introduced in Equation 3.9 to parse out natural and fishing mortality rates

$$N(a+\Delta a, t+\Delta t) = \begin{cases} N(a,t)\cdot e^{-M(a)} & \text{if } a(t) < a_c(t) \\ N(a,t)\cdot e^{-[F(a,t)+M(a)]} & \text{if } a(t) \geqslant a_c(t) \end{cases} \tag{4.2}$$

where fishing mortality rate at time t was only imposed on species that were above the age at first capture, and time steps Δt were one month. The minimum ages at first capture, $a_c(t)$, were synced to minimum size limits, converted to millimeters fork length, and matched actual fishing regulations through time. The simulated population was run to equilibrium levels that matched those realized under the first month of data, t. The equilibrium population structure then replaced $N(a,t)$ in Equation 4.2 for future simulations under actual conditions. Average population sizes throughout monthly intervals for each age class were calculated as

$$\bar{N}(a,t) = \frac{N(a,t)}{Z(a,t)}\cdot(1-e^{-Z(a,t)}) \tag{4.3}$$

which estimated abundance at the midpoint of the interval. Age-structured population abundance was converted to biomass utilizing age-specific weights, $\bar{W}(a)$, from Equation 2.2.

$$\bar{B}(a,t) = \bar{N}(a,t)\cdot\bar{W}(a) \tag{4.4}$$

Total yield in numbers, $Y_N(a,t)$, was estimated by applying the instantaneous fishing mortality rate, $F(a,t)$, on the population abundance susceptible to exploitation.

$$Y_N(a,t) = F(a,t) \cdot \bar{N}(a,t) \qquad \text{if } a(t) \geqslant a_c(t) \tag{4.5}$$

The estimate of total yield in numbers (of recreational and commercial fisheries) was then related to yield in weight, $Y_W(a,t)$.

$$Y_W(a,t) = F(a,t) \cdot \bar{B}(a,t) \qquad \text{if } a(t) \geqslant a_c(t) \tag{4.6}$$

The simulated yield in weight, $Y_W(a,t)$, was summed across age classes above the age at first capture, $a_c(t)$, to represent total yield.

$$\hat{Y}_W(t) = \sum_{a=a_c(t)}^{a_\lambda \cdot 12} Y_W(a,t) \tag{4.7}$$

The difference between the simulated yield in weight, $\hat{Y}_W(t)$, and observed yield in weight, $Y_W(t)$, was minimized by iteratively adjusting recruitment, $N(0,t)$, which was assumed to be constant throughout the simulation period. The model was validated by comparing time series of simulated catch to total reported catch and simulated average length to average length calculated using the TIP data.

4.1.2 Sustainability Benchmarks

Sustainability benchmarks have been developed to allow for comparison of population status estimates for any species relative to a pre-defined threshold. Spawning potential ratio (SPR), a common biological benchmark, was an indicator of the stock's ability to reproduce and replenish the population. SPR was defined as the ratio of the current spawning stock biomass (SSB) to the SSB of an unfished stock. SSB was calculated as the total population biomass above the age at 50% maturity, a_m, defined in Chapter 2.

$$SSB(t) = \sum_{a=a_m}^{a_\lambda} \bar{B}(a,t) \tag{4.8}$$

The age-structured population model built in Section 4.1.1 was run with no fishing mortality, $F(a,t) = 0$, to estimate the unfished biomass, $\bar{B}_0$, which was then plugged into Equation 4.8 to calculate the unfished spawning stock biomass, SSB_0. The ratio of current to unfished SSB,

$$SPR(t) = \frac{SSB(t)}{SSB_0} \tag{4.9}$$

defined SPR, which should be above approximately 40% to maximize yields, and was considered overfished if it dropped below 30%.

Fishing rates that promoted long-term sustainability were equal to the natural mortality rates of the populations, $F \approx M$. In a surplus production model, this rate produces the maximum sustainable yield, MSY. The fishing mortality rate producing MSY, F_{MSY}, and associated population biomass, B_{MSY}, were often used as benchmarks to determine if a population was undergoing overfishing or was currently overfished, respectively. The age-structured model described in Equations 4.2–4.7 was run with $F_{MSY} = M$ under identical initial conditions to determine the associated population biomass, $\bar{B}_{MSY}$, and estimate overfishing and overfished status throughout the model time frame. Overfishing was defined as $F(t)/F_{MSY} > 1$, and overfished populations were defined as $\bar{B}(t)/\bar{B}_{MSY} < 1$.

4.1.3 Validating Age-Structured Bioeconomic Models

Age-structured bioeconomic models were created by linking the numerical cohort models developed in Section 4.1.1 with the functions defining commercial fleet revenue and cost relative to catch and effort, respectively, in Chapter 3. Outputs from the age-structured biological production models were used as inputs to the economic models then plotted with raw fisheries economics data to observe the bioeconomic models' ability to track reality. First, the population-wide assessment had to be translated to commercial productivity. The commercial yield, $Y_c(t)$, was estimated through the proportion of commercial yield to the total yield in weight, $Y_W(t)$.

$$\frac{Y_c(t)}{Y_W(t)} = \tau_c(t) \tag{4.10}$$

The proportion of commercial landings, $\tau_c(t)$, was then used to estimate the proportion of fishing mortality rate attributed to the commercial fleets. Equations 4.2-4.7 were combined into an expanded form with the inclusion of proportionality constants $\tau_c(t)$ and $\nu_c(t)$,

$$\tau_c(t) \cdot \hat{Y}_W(a,t) = \nu_c(t) \cdot F(a,t) \cdot \frac{N(a,t) \cdot W(a)}{[\nu_c(t) \cdot F(a,t) + M]} \cdot [1 - e^{-[\nu_c(t) \cdot F(a,t) + M]}] \tag{4.11}$$

where $\nu_c(t)$ was defined as the proportion of instantaneous fishing mortality rate $F(a,t)$ allocated to the commercial fleet. Due to the nonlinear nature of age-structured production models, it was impossible to analytically solve for the proportion of the fishing mortality rate due to the primary fleets, so the difference between the left- and right-sides of this equation were minimized by iteratively solving for ν_c. The simulated monthly commercial catch biomass by age, $\hat{Y}_c(a,t)$, was defined as

$$\hat{Y}_c(a,t) = \nu_c(t) \cdot F(a,t) \cdot \bar{B}(a,t) \qquad \text{if } a(t) \geqslant a_c(t) \tag{4.12}$$

then summed over age classes to estimate total monthly commercial catch biomass, $\hat{Y}_c(t)$,

$$\hat{Y}_c(t) = \sum_{a=a_c(t)}^{a_\lambda} \hat{Y}_c(a,t) \tag{4.13}$$

where length at first capture throughout the simulation time frame, $L_c(t)$, was compiled in Table 2.12 and converted to age at first capture, $a_c(t)$. These simulated landings were used as inputs in the demand function defined in Equation 3.6 to estimate ex-vessel prices.

Variation in ex-vessel price was primarily explained by commercial landings, so ex-vessel price was estimated with the simulated landings assuming all other inputs remained unchanged (import prices of substitute goods, implementation of the IFQ program, and disposable income). The simulated commercial landings were lagged one month ($k = -1$) per results in Table 3.3 to estimate ex-vessel price, $p[t, \hat{Y}_c(t+k)]$, and input into Equation 3.6. The predicted ex-vessel prices were multiplied by the simulated commercial landings in month t, $\hat{Y}_c(t)$, which resulted in estimation of total monthly commercial revenue from the 'analysis' species.

$$\hat{R}_{cs}(t) = p[t, \hat{Y}_c(t+k)] \cdot \hat{Y}_c(t) \tag{4.14}$$

Total commercial revenue from the 'analysis' species was allocated to the primary fleets based on proportions of yield generated through time. Revenue generated by the target species for each primary fleet, $\hat{R}_{gs}(t)$, was defined as

$$\hat{R}_{gs}(t) = \tau_{gs}(t) \cdot \hat{R}_{cs}(t) \tag{4.15}$$

where $\tau_{gs}(t)$ was the monthly proportion of commercial yield attributed to permitted vessels operating the primary gears targeting each species by fleet. To define financial benefits to the South Atlantic and Gulf of Mexico exclusively, revenue was allocated to the permit-issuing region, and revenue generated by vessels with permits in both regions (e.g. YT/HL.GMSA1), was allocated by applying the average proportion of revenue attained from South Atlantic and Gulf of Mexico waters 2012-2016.

Commercial fleets incurred variable costs at different rates, requiring identification of fishing effort by fleet in the simulation model to more accurately estimate costs (Table 3.10, Figures 3.10–3.12). The annual mortality rate estimated in Equation 3.17 was allocated to the individual fleets based on the proportion of each fleets' landings to the total yield following Equation 4.11. The proportion of yield attributed to the primary fleets, $T_g(t)$, was defined as

$$T_g(t) = \tau_c(t) \cdot \tau_g(t) \tag{4.16}$$

where τ_c was the proportion of total landings that were commercial and τ_g was the proportion of commercial landings to each primary fleet. The proportion of total yield landed by each fleet, $T_g(t)$, was used to estimate the proportion of fishing mortality rate attributed to each fleet, $\nu_g(t)$,

$$T_g(t) \cdot \hat{Y}(a,t) = \nu_g(t) \cdot F(a,t) \cdot \frac{N(a,t) \cdot W(a)}{[\nu_g(t) \cdot F(a,t) + M]} \cdot [1 - e^{-[\nu_g(t) \cdot F(a,t) + M]}] \tag{4.17}$$

where $\nu_g(t)$ was defined as the proportion of instantaneous fishing mortality rate $F(a,t)$ allocated to each g fleet. This allowed for the estimation of the fishing mortality rate attributed to each fleet g, $F_g(a,t)$.

$$F_g(a,t) = \nu_g(t) \cdot F(a,t) \tag{4.18}$$

Fishing mortality rates from the commercial fleets, $F_g(a,t)$, were converted to number of trips per month of these fleets, $f_g(t)$, through estimation of the catchability coefficient, q_g.

$$q_g = \frac{F_g}{f_g} \tag{4.19}$$

F_g and f_g were monthly averages from 2012-2016 and were considered representative of the dataset. Catchability, q_g, was assumed to be constant over the time period, 1995–2016, and was validated for each fleet by independently estimating population biomass through the fundamental equation

$$\bar{B}_g(t) = \frac{Y_g(t)}{q_g \cdot f_g(t)} \tag{4.20}$$

where $Y_g(t)$ and $f_g(t)$ were yield and number of trips, respectively, attributed to each fleet g throughout the simulation time frame. Once validated, the catchability coefficient q_g was used to estimate the fishing effort required by each fleet to attain the estimated landings via $F_g(t)$. With the rearrangement of Equation 4.19, the number of trips per month for each fleet, $\hat{f}_g(t)$, was estimated from the simulated fishing mortality rate, $F_g(t)$.

$$\hat{f}_g(t) = \frac{F_g(t)}{q_g} \tag{4.21}$$

Jointly-caught revenue was revenue generated by any species other than the single target species on all trips $\hat{f}_g(t)$ by vessels in each fleet g. This revenue was divided into $\psi = 3$ categories based on federal permit delineations and results shown in Tables 3.8 & 3.9: (1) Reef fish ; (2) Cobia, King & Spanish Mackerel ; and (3) Other. Monthly jointly-caught revenue for each fleet, $\hat{R}_{g\psi}(t)$, was estimated as

$$\hat{R}_{g\psi}(t) = \sum_{\psi_g=1}^{3} [\beta_{g0} + \beta_{g1} \cdot \hat{f}_g(t) \cdot d_g(t)] \tag{4.22}$$

where $\hat{f}_g(t)$ was the estimated number of trips per month t by each fleet g and $d_g(t)$ was the average number of days per trips by fleet g in month t. Coefficient estimates and required transformations of data were reported in Tables 3.8 & 3.9. Shifts in production possibility frontiers, or the tendency of fishermen to shift target species under different management or environmental conditions, were explored but ignored here. In other words, it was assumed that fishermen would land jointly-caught species at the same rates observed under the current management regime.

Trip-level costs for each fleet were fit in Equation 3.25 using explanatory variables r = days fished and vessel length. Monthly averages of these trip-level fleet characteristics, $\Phi_{gr}(t)$, were input into Equation 3.25, transformed into real space, then multiplied by the estimated number of trips.

$$\hat{C}_g(t) = [\beta_{g0} + \beta_{gr} \cdot \Phi_{gr}(t)]^2 \cdot \hat{f}_g(t) \tag{4.23a}$$

$$\hat{C}_g(t) = exp[\beta_{g0} + \beta_{gr} \cdot \Phi_{gr}(t)] \cdot \hat{f}_g(t) \tag{4.23b}$$

Transformations were applied based on the reversal of those required to obtain linearity defined in Table 3.11. Costs for vessels with permits in both the South Atlantic and Gulf of Mexico (GMSA1) were divided using the proportion of trips these permit holders operated in each respective region 2012–2016.

Monthly variable costs were subtracted from the revenue generated by target and nontarget species for fleet g. Net revenue, $\hat{\Pi}_g(t)$, was estimated monthly as

$$\hat{\Pi}_g(t) = \hat{R}_{gs}(t) + \hat{R}_{g\psi}(t) - \hat{C}_g(t) \tag{4.24}$$

where $\hat{R}_{gs}(t)$ was revenue from the 'analysis' species s estimated in Equation 4.15, $\hat{R}_{g\psi}(t)$ was revenue from jointly-caught species ψ estimated in Equation 4.22, and $\hat{C}_g(t)$ was the cost to operate these trips estimated in Equation 4.23. All results in this section were compared to actual catch, ex-vessel prices, revenue, number of trips, costs, and net revenue to validate the models under actual conditions before proceeding with simulations of different management strategies.

Table 4.1: Biological and economic variables for functions parameterizing age-structured bioeconomic simulation models.

Variable	Definition	Units	Equation
a	Age	months	
t	Time	months	
$Z(a,t)$	Total mortality rate		
$F(a,t)$	Fishing mortality rate		
$N(a,0)$	Initial, unfished population structure	count	4.1
$N(0,t)$	Number of recruits to the population	count	
$a_c(t)$	Age at first capture	months	4.2
$N(a,t)$	Abundance	count	4.2
$\bar{N}(a,t)$	Average abundance	count	4.3
$\bar{B}(a,t)$	Average biomass	pounds	4.4
$\hat{Y}_N(a,t)$	Simulated yield in numbers	count	4.5
$\hat{Y}_W(a,t)$	Simulated yield in weight of age a fish	pounds	4.6
$\hat{Y}_W(t)$	Simulated yield in weight	pounds	4.7
$Y_W(t)$	Observed yield in weight	pounds	
$\bar{B}_0$	Average biomass of unfished population	pounds	
$SSB(t)$	Spawning Stock Biomass	pounds	4.8
SSB_0	Spawning Stock Biomass ($F=0$)	pounds	
$SPR(t)$	Spawning Potential Ratio		4.9
F_{MSY}	Fishing mortality rate at MSY		
B_{MSY}	Population biomass at MSY	pounds	
$\tau_r(t)$	Proportion of yield to recreational fleet r		4.11
$\nu_r(t)$	Proportion of F to recreational fleet r		4.11
$\hat{Y}_c(a,t)$	Commercial yield of age a fish	pounds	4.12
$\hat{Y}_c(t)$	Commercial yield	pounds	4.13
$p[t,\hat{Y}_c(t+k)]$	Ex-vessel price	2016\$	4.15
k_j	Lag between $p(t)$ and $x_(t)$	months	4.15
$\hat{\bar{R}}_{cs}(t)$	Revenue of the target species	2016\$	4.15
$\tau_g(t)$	Proportion of commercial yield to fleet g		4.15
$\hat{\bar{R}}_{gs}(t)$	Simulated revenue of target species s	2016\$	4.15
$T_g(t)$	Proportion of total yield to fleet g		4.16
$\nu_g(t)$	Proportion of F to fleet g		4.17
$F_g(a,t)$	F attributed to fleet g		4.18
q_g	Catchability of fleet g		4.19

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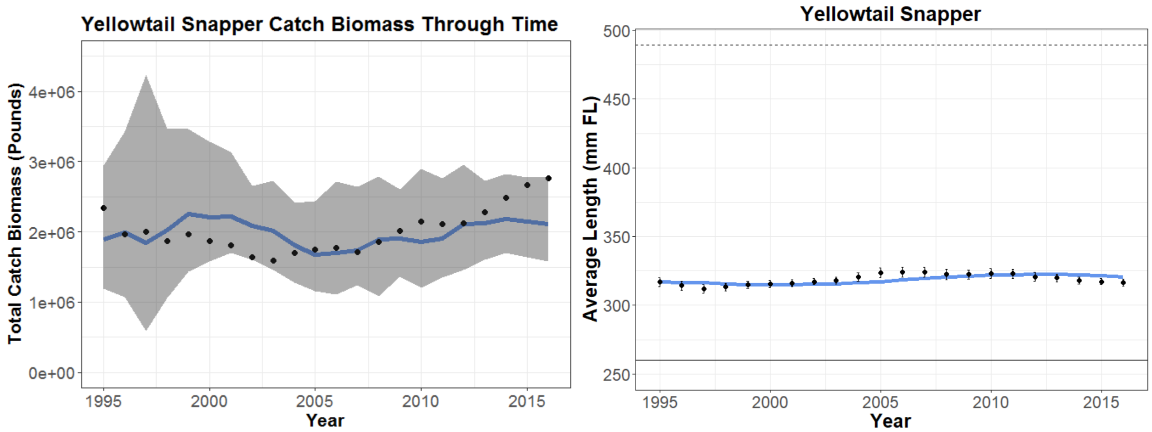
Variable	Definition	Units	Equation
$\hat{f}_g(t)$	Simulated number of trips	trips	4.21
$\hat{R}_{g\psi}(t)$	Simulated revenue from jointly-caught species ψ	2016$	4.22
$d_g(t)$	Number of days fished by fleet g in month t	days	4.22
$\hat{C}_g(t)$	Total costs of fleet g in month t	2016$	4.23
$\Phi_{gr}(\eta)$	r trip-level explanatory cost variables		4.23
$\Pi_g(t)$	Net revenue of fleet g in month t	2016$	4.24

4.2 Results

4.2.1 Total Mortality Rates and Assessment Results

The bioeconomic model simulations began in 1995 because catch data reliability improved, allowing for validation of the simulated catch from the numerical cohort model. The yellowtail snapper model was run to equilibrium under 1995 conditions using average lengths from 1993-1995, scaled with constant recruitment to match observed commercial and recreational catches, $N(0,t) = 249,500$. The simulated catch tracked the same space as the observed catch, with only slight divergences, and the simulated average length of the exploited phase approximately matched the calculated average length from the TIP database (Figure 4.2). From 2012-2016, there was a decrease in the average size of the TIP catch composition, indicating an increase in fishing mortality rate, that resulted in an increased observed catch in recent years. The yellowtail snapper simulation model captured the dynamics of catch through the numerical cohort model.

Detailed population length structures of gag grouper were shown at the start of the simulation in 1995 with the unfished population structure to illustrate population truncation under increasing mortality rates (Figure 4.3). In 1995, Gulf of Mexico



(a) Catch (b) Average Length

Figure 4.2: (a) Simulated yellowtail snapper catch (blue line) with the 95% confidence interval (CI) shaded compared to the actual catch (black dots). (b) Average length of the simulated population (blue line) compared to the average length of the TIP data (black dots with 95% CI error bars).

gag grouper were experiencing a total mortality rate of $Z_{95} = 0.266$, while South Atlantic gag grouper $Z_{95} = 0.170$. The lower mortality rate in the South Atlantic was synonymous with a larger average size in the exploited phase. The average length of the unfished population was largest because it was only subjected to natural mortality, $Z = 0.097$. As fishing rates increased, the population was truncated more severely, resulting in a decreased average length of the exploited phase.

The gag grouper models were run to equilibrium under 1995 conditions using average lengths 1991-1995, scaled with constant recruitment by region to match the observed total catches in the Gulf of Mexico ($N(0,t) = 43,000$) and South Atlantic ($N(0,t) = 4,200$). The difference in recruitment between stocks was likely because the South Atlantic model was only capturing Florida landings, approximately one-third of the catch that was landed throughout the South Atlantic, while nearly all of the catch from the Gulf of Mexico stock was landed in Florida. Gag grouper

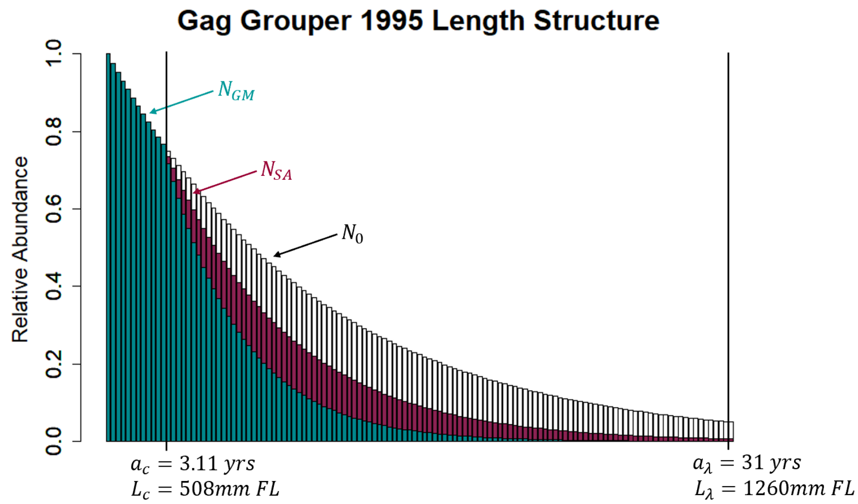
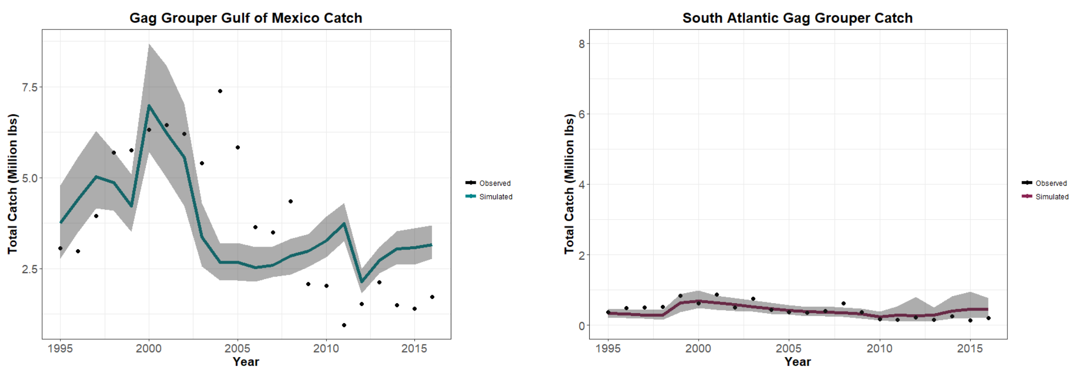


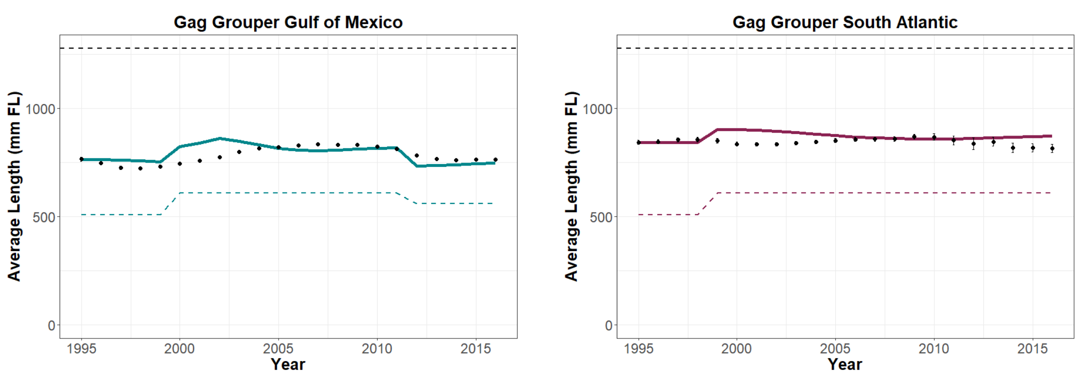
Figure 4.3: Length structure of gag grouper abundance in 1995 in the Gulf of Mexico, N_{GM}, and South Atlantic, N_{SA}. These were truncated from the unfished population, N_0, through their relative fishing mortality rates, $F_{GM} = 0.169$ and $F_{SA} = 0.073$, imposed on the population above the minimum size limit, L_c. The average length of the exploited phase between the age at first capture and the maximum age, $a_c \leq a \leq a_\lambda$, was larger for N_{SA} than N_{GM} due to the lower fishing mortality rate.

simulation results did not track the observed data as well as yellowtail snapper results. Yellowtail snapper has had the same minimum size limit in place since 1983, no closed seasons, and limited closures due to quotas. The gag grouper population experienced multiple regulatory changes throughout the simulation time frame including different minimum size limits and seasonal closures between regions and sectors, muddling the results from the average length mortality estimator. The simulation model captured the general trends of catch in both the Gulf of Mexico and South Atlantic, but the increased minimum size limit in the year 2000 (508mm FL to 610mm FL) appeared to not be enforced immediately, resulting in lagged responses in the actual data (Figure 4.4). The Gulf of Mexico reduced the commercial minimum size limit to 559mm FL in 2012, likely in a response to reduced catch rates (size limit changes reflect 20in, 24in, 22in TL). The simulated average length of the exploited phase matched the calculated TIP average length for both regions, but the simulation model was faster to reach equilibrium than the actual data, which appeared to have lagged transitions in average length following changes in size limit regulations (Figure 4.5). Overall, the simulation model appeared to capture the dynamics of the Florida gag grouper populations in the Gulf of Mexico and South Atlantic.



(a) Gulf of Mexico (b) South Atlantic

Figure 4.4: Simulated gag grouper catch in the (a) Gulf of Mexico and (b) South Atlantic with 95% confidence intervals shaded.



(a) Gulf of Mexico (b) South Atlantic

Figure 4.5: Gag grouper average length from the simulation (solid lines) compared to the calculated average length from the TIP data (black dots) in the (a) Gulf of Mexico and (b) South Atlantic. The length at first capture L_c was shown for each region with dashed lines.

4.2.2 Current Sustainability Status

The SPR of yellowtail snapper was 25.06% in 1995, increased to a peak of 29.31% in 2011, then decreased to 26.94% in 2016 (Table 4.2). This agreed with the indicators measuring overfishing and overfished status, F/F_{MSY} and B/B_{MSY}, respectively, which followed a path towards sustainability then back towards more overfished in recent years (Figure 4.6). Yellowtail snapper ended the simulation approximately where it started despite rebuilding efforts in the mid-late 2000s: $F/F_{MSY} = 2.11$ and $B/B_{MSY} = 0.66$. Gag grouper in the Gulf of Mexico experienced its highest SPR at the beginning of the simulation time frame, when SPR was only 20.86%. The SPR dropped to a low of 7.01% in 2004 and has slowly increased to 12.25% in 2016 (Table 4.2). GM gag had been reducing fishing mortality rates but was still experiencing overfishing rates and severe overfished status at the end of the simulation timeframe, $F/F_{MSY} = 2.45$ and $B/B_{MSY} = 0.38$ (Figure 4.6). Gag grouper SPR in the South Atlantic peaked in 1998 at 45.60%, then dropped to a low of 27.93% in 2008, and has stabilized around 32.24% in 2016 (Table 4.2). SA gag grouper began the simulation not overfished and overfishing not occurring, but fishing mortality rate has steadily increased resulting in $F/F_{MSY} = 2.10$ and $B/B_{MSY} = 0.74$ in 2016 (Figure 4.6).

Table 4.2: Sustainability benchmarks were calculated annually for all populations modeled throughout the simulation period. Gulf of Mexico gag grouper experienced the highest fishing mortality rates throughout all simulations, which resulted in the lowest spawning potential ratio (SPR) out of any population modeled here. By the end of the simulation time frame, all stocks were undergoing overfishing and in an overfished state.

	Yellowtail Snapper			**GM Gag Grouper**			**SA Gag Grouper**		
Year	$\frac{F}{F_{MSY}}$	$\frac{B}{B_{MSY}}$	SPR	$\frac{F}{F_{MSY}}$	$\frac{B}{B_{MSY}}$	SPR	$\frac{F}{F_{MSY}}$	$\frac{B}{B_{MSY}}$	SPR
1995	2.10	0.62	25.0%	1.75	0.60	20.9%	0.76	1.22	44.7%
1996	2.30	0.61	24.8%	2.08	0.59	20.5%	0.73	1.22	44.8%
1997	2.48	0.60	24.1%	2.50	0.56	19.5%	0.65	1.23	45.1%
1998	2.37	0.59	23.6%	2.59	0.53	18.2%	0.64	1.24	45.6%
1999	2.24	0.58	23.4%	2.37	0.50	17.2%	1.45	1.21	44.3%
2000	2.18	0.59	23.6%	4.49	0.45	15.0%	1.70	1.12	41.0%
2001	2.15	0.59	23.8%	5.11	0.36	11.6%	1.73	1.03	37.5%
2002	2.08	0.60	24.1%	5.92	0.28	8.7%	1.74	0.95	34.4%
2003	2.02	0.61	24.4%	4.30	0.24	7.2%	1.62	0.89	32.0%
2004	1.83	0.62	25.1%	3.51	0.24	7.0%	1.53	0.85	30.3%
2005	1.65	0.64	26.0%	3.37	0.25	7.3%	1.43	0.82	29.2%
2006	1.63	0.66	27.1%	3.01	0.26	7.8%	1.35	0.80	28.5%
2007	1.61	0.68	27.9%	2.87	0.28	8.4%	1.33	0.79	28.1%
2008	1.72	0.69	28.5%	2.98	0.29	8.9%	1.30	0.78	27.9%
2009	1.72	0.70	28.8%	2.97	0.30	9.4%	1.16	0.79	28.0%
2010	1.68	0.70	29.0%	3.19	0.31	9.6%	1.18	0.80	29.3%
2011	1.70	0.71	29.3%	3.72	0.30	9.4%	1.42	0.81	30.5%
2012	1.83	0.70	29.2%	2.06	0.31	9.8%	1.67	0.80	31.3%
2013	1.88	0.70	28.9%	2.39	0.34	10.7%	1.51	0.80	32.6%
2014	2.01	0.69	28.3%	2.54	0.35	11.3%	2.04	0.79	33.1%
2015	2.08	0.67	27.6%	2.47	0.37	11.8%	2.07	0.77	33.0%
2016	2.11	0.66	26.9%	2.45	0.38	12.2%	2.10	0.74	32.2%

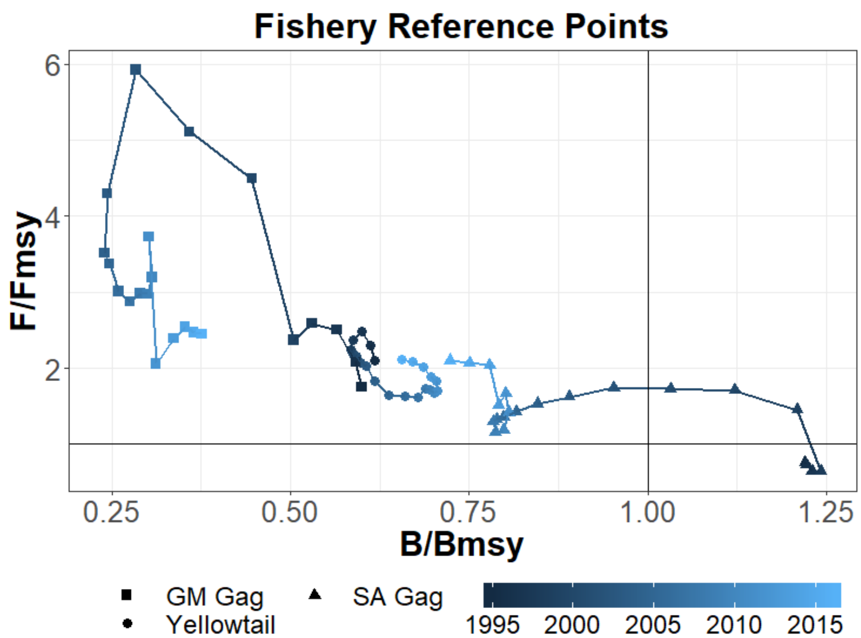


Figure 4.6: Fishery reference points for yellowtail snapper (circle), Gulf of Mexico (GM) gag grouper (square), and South Atlantic (SA) gag grouper (triangle) from 1995-2016. The yellowtail snapper stock approached sustainability, then the annual catch limits were increased allowing for further overexploitation. SA gag grouper steadily approached overfished status throughout the years, and GM gag grouper increased then decreased fishing mortality rates, but remained in severe overfished status 1995-2016.

4.2.3 Validation of Age-Structured Bioeconomic Model

The simulated catch and fishing mortality rates were validated in the previous section and used as inputs to economic functions fit in Chapter 3, resulting in the bioeconomic simulation model sought to be validated here. Commercial fishing mortality rates were estimated using the proportionality constant, τ_c, to simulate commercial yields (Equation 4.11, Table 4.3). The yellowtail snapper commercial sector represented a majority of the fishing mortality rate, while Gulf of Mexico gag grouper and South Atlantic gag grouper were overall dominated by the recreational sector. Simulated commercial yield was used to estimate ex-vessel price and tracked the data well, but did not entirely capture the seasonality displayed from the inverse demand function using real data as an input (Figure 4.7). The simulated ex-vessel price was multiplied by the simulated landings to generate revenue for the commercial fishery. Revenue was allocated to each fleet based on historical distribution of revenue between fleets then distributed to the Gulf of Mexico and South Atlantic based on permit type jurisdiction (Table 4.4). For vessels with a permit in both regions, the YT/HL/GMSA1 fleet, 42.0% of the trips were operated in the Gulf of Mexico, but 80.2% of their revenue was generated from this region, and the costs and revenue simulated for this fleet were allocated accordingly (Figure 4.8). The relatively high percentage of catch from the Gulf of Mexico despite the lower percentage of trips operating in this region could be explained by the Dry Tortugas, which was highly productive and distant, yielding higher catches but requiring longer run times per trip to reach.

Table 4.3: Proportion of the fishing mortality rate attributed to the commercial fleet, ν_c, was estimated in Equation 4.11. Yellowtail snapper fishing mortality rates were primarily commercial, and both Gulf of Mexico and South Atlantic gag grouper mortality rates were mostly recreational.

Year	**Yellowtail Snapper**	**GM Gag Grouper**	**SA Gag Grouper**
1995	0.75	0.11	0.45
1996	0.78	0.37	0.53
1997	0.76	0.28	0.49
1998	0.75	0.30	0.55
1999	0.76	0.22	0.17
2000	0.80	0.11	0.19
2001	0.82	0.31	0.12
2002	0.83	0.27	0.20
2003	0.81	0.31	0.10
2004	0.75	0.19	0.15
2005	0.72	0.25	0.26
2006	0.70	0.24	0.26
2007	0.66	0.24	0.36
2008	0.58	0.17	0.12
2009	0.58	0.28	0.31
2010	0.66	0.15	0.64
2011	0.75	0.30	0.63
2012	0.80	0.48	0.76
2013	0.78	0.46	0.73
2014	0.75	0.49	0.59
2015	0.72	0.54	0.68
2016	0.71	0.57	0.57

Table 4.4: Proportion of the catch biomass each fleet landed, τ_g, was shown throughout the simulation time frame. All catch from yellowtail snapper and gag grouper was primarily landed with hook-and-line gear, and yellowtail snapper hook-and-line catch was dominated by the Snapper Grouper Unlimited Trip Limit (SA1) fleet. Both the South Atlantic and the Gulf of Mexico gag grouper spearfishing gears became more prevalent throughout the simulation time frame. These annual proportions were utilized to allocate simulated revenue and estimate the proportion of fishing mortality rate attributed to each fleet.

	Yellowtail Snapper, HL				**Gag Grouper, GM**			**Gag Grouper, SA**		
Year	SA2	SA1	GM	GMSA1	HL	LL	SP	HL	LL	SP
1995	0.04	0.63	0.03	0.30	0.82	0.18	0.00	0.75	0.08	0.17
1996	0.04	0.63	0.03	0.30	0.79	0.20	0.01	0.71	0.05	0.24
1997	0.04	0.63	0.03	0.30	0.77	0.21	0.02	0.65	0.07	0.28
1998	0.04	0.63	0.03	0.30	0.79	0.20	0.01	0.66	0.07	0.27
1999	0.04	0.63	0.03	0.30	0.76	0.23	0.01	0.61	0.07	0.32
2000	0.04	0.63	0.03	0.30	0.77	0.22	0.02	0.76	0.06	0.18
2001	0.04	0.63	0.03	0.30	0.70	0.29	0.01	0.64	0.08	0.28
2002	0.04	0.63	0.03	0.30	0.65	0.34	0.01	0.64	0.04	0.32
2003	0.04	0.63	0.03	0.30	0.60	0.38	0.01	0.55	0.03	0.42
2004	0.04	0.62	0.03	0.30	0.65	0.34	0.01	0.77	0.03	0.20
2005	0.04	0.62	0.03	0.30	0.65	0.34	0.02	0.68	0.06	0.25
2006	0.04	0.65	0.03	0.28	0.60	0.38	0.02	0.55	0.02	0.44
2007	0.05	0.59	0.03	0.33	0.61	0.38	0.02	0.52	0.00	0.48
2008	0.04	0.67	0.03	0.27	0.71	0.26	0.02	0.41	0.00	0.59
2009	0.02	0.69	0.03	0.26	0.74	0.22	0.04	0.48	0.00	0.52
2010	0.02	0.78	0.09	0.12	0.71	0.23	0.05	0.34	0.00	0.65
2011	0.01	0.72	0.13	0.13	0.67	0.25	0.09	0.54	0.01	0.45
2012	0.02	0.81	0.10	0.11	0.69	0.23	0.08	0.59	0.00	0.41
2013	0.01	0.73	0.11	0.17	0.64	0.32	0.05	0.67	0.00	0.33
2014	0.02	0.55	0.18	0.24	0.55	0.36	0.09	0.60	0.02	0.38
2015	0.02	0.54	0.14	0.31	0.43	0.48	0.09	0.56	0.02	0.42
2016	0.01	0.69	0.07	0.23	0.53	0.42	0.05	0.54	0.00	0.46

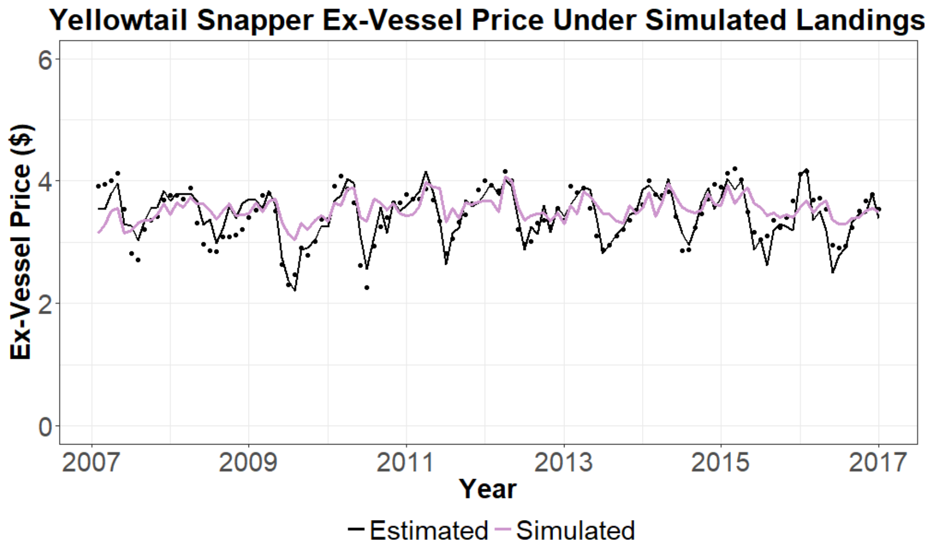
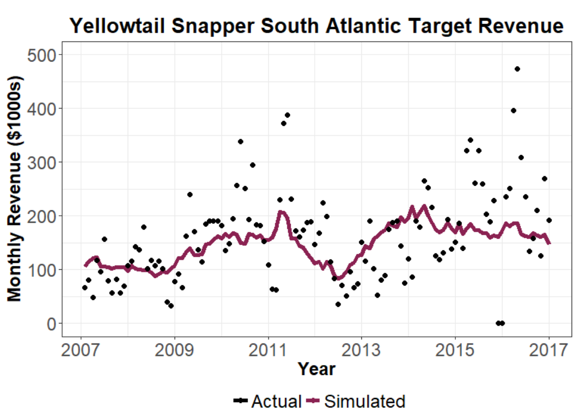
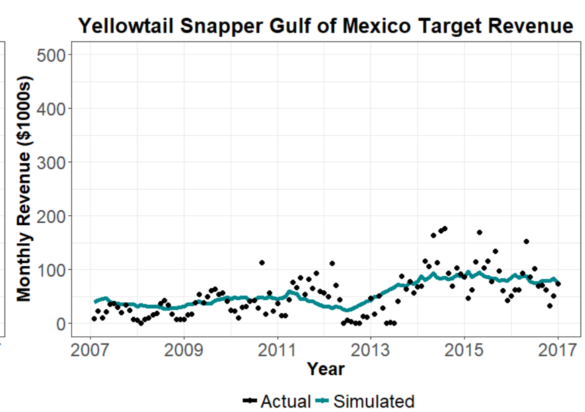


Figure 4.7: Actual ex-vessel price (black dots) were used as a reference to compare estimated prices and simulated prices. Estimated prices (black lines) were modeled using actual landings in the inverse demand function, while simulated prices (purple lines) were modeled using simulated landings in the inverse demand function.



(a) South Atlantic



(b) Gulf of Mexico

Figure 4.8: Simulated revenue of yellowtail snapper was estimated by multiplying the landings from the age-structured production model with the simulated ex-vessel price, then compared to actual revenue from each management jurisdiction.

Number of trips per month by each fleet was simulated from the proportion of fishing mortality rate attributed to each fleet annually divided by a constant catchability coefficient estimated in Equation 4.19. Out of the yellowtail hook-and-line fleets, the South Atlantic Snapper Grouper Unlimited Trip Limit (SA1) fleet had the highest number of trips per month, f_g (Table 4.5). Vessels with both the Gulf of Mexico Reef Fish and South Atlantic Unlimited (GMSA1) had the highest catchability, q_g, out of any other yellowtail snapper hook-and-line fleet, despite the fact that the GM fleet ran longer trips. This could be explained by the assumed experience of fishermen in the industry long enough to obtain permits for both federal jurisdictions. Another explanation could be that vessels with GMSA1 permits were almost exclusively operating out of Monroe county, the center of the yellowtail snapper distribution. Hook-and-line gear comprised the majority of the trips targeting gag grouper in both the Gulf of Mexico and the South Atlantic. Out of the Gulf of Mexico gag grouper fleets, longline gear had the highest catchability and the longest trips by far, giving them more time to catch more gag grouper on any given trip, resulting in a higher catchability. The South Atlantic gag grouper gear with the highest catchability was spearfishing. Overall, the Gulf of Mexico longline fleet had the largest vessels and fished the longest trips, while the South Atlantic Snapper Grouper 225lb Trip Limit (SA2) fleet was characterized by the smallest vessels and shortest fishing trips.

Jointly-caught species were accumulated at different rates for each fleet, primarily due to interactions between gears and habitats. The revenue from ψ jointly-caught species was estimated individually for each g fleet, resulting in 30 regressions fitted for all combinations of the 4 yellowtail fleets and 6 gag grouper fleets with 3 defined categories of jointly-caught species, ψ (Tables 3.8 & 3.9). Simulated jointly-caught

Table 4.5: Fleet characteristics required for estimation of costs and jointly-caught revenue included average vessel length (feet), average days per trip, average trips per month, and estimated catchability.

Stock	**Fleet**	**Length**	**Days/Trip**	T_g	F_g	f_g	q_g
Yellowtail	SA2	25.4	1.06	0.01	0.000125	52.9	2.36E-06
Yellowtail	SA1	27.7	1.44	0.22	0.004679	268.3	1.74E-05
Yellowtail	GM	30.6	3.05	0.04	0.000844	37.8	2.24E-05
Yellowtail	GMSA1	32.2	1.82	0.09	0.001844	52.6	3.51E-05
Gag, GM	HL	36.8	3.91	0.16	0.003646	295.8	1.23E-05
Gag, GM	LL	45.8	8.98	0.14	0.003036	63.0	4.82E-05
Gag, GM	SP	30.9	1.83	0.02	0.000511	18.1	2.83E-05
Gag, SA	HL	29.8	1.60	0.24	0.002320	274.7	8.45E-06
Gag, SA	LL	42.2	1.90	0.01	0.000061	15.3	3.98E-06
Gag, SA	SP	26.9	1.71	0.18	0.001675	56.7	2.96E-05

reef fish revenue captured the trends observed in the actual data throughout the timeframe, as shown in Gulf of Mexico gag grouper spearfishing fleets (Figure 4.9). Following this example, the simulated number of trips was plugged into the remaining 29 jointly-caught revenue functions and summed up across fleets, with appropriate transformations applied. The fleet-wide jointly-caught revenue was added to the ‘analysis’ species revenue by fleet and summed across jurisdictions, resulting in the simulated aggregate revenue for ‘analysis’ species in the South Atlantic and Gulf of Mexico (Figure 4.10).

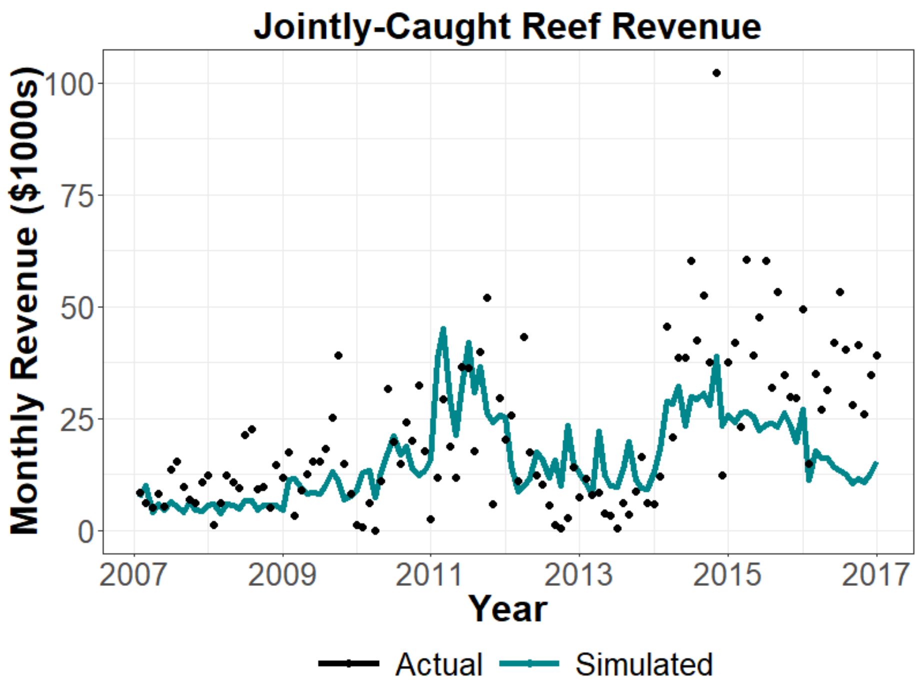
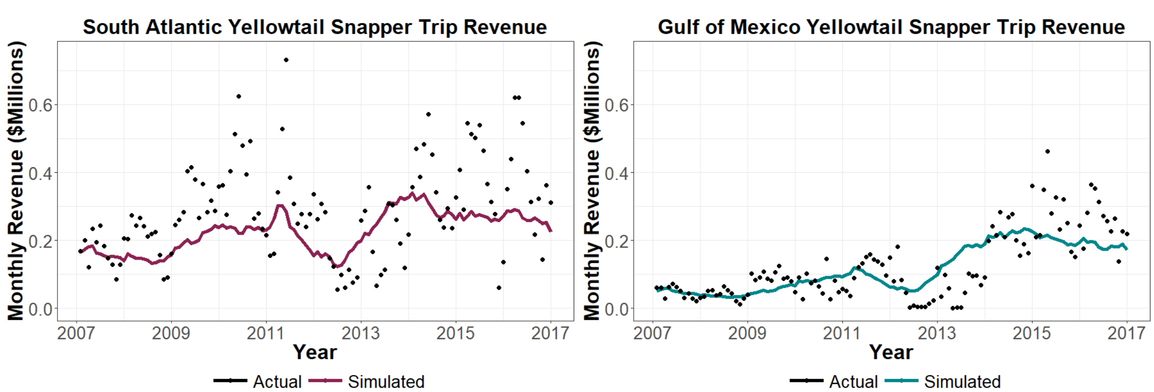
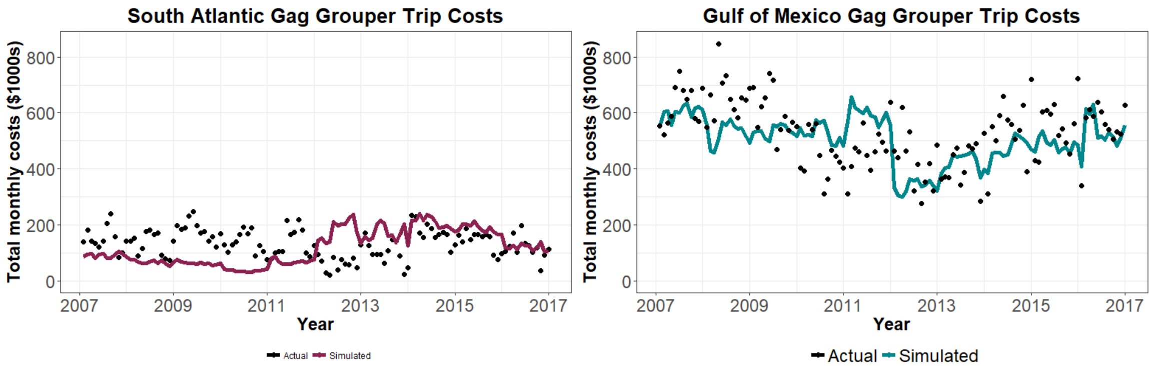


Figure 4.9: Simulated jointly-caught reef fish revenue for the Gulf of Mexico gag grouper spearfishing fleet (blue line) tracked the actual revenue from this source (black dots) and was counted towards the Gulf of Mexico gag grouper aggregate revenue.



(a) South Atlantic (b) Gulf of Mexico

Figure 4.10: Aggregate revenue for the yellowtail snapper fleets was estimated by summing up 'analysis' and jointly caught revenue monthly. These were then compiled by management jurisdiction and compared here.



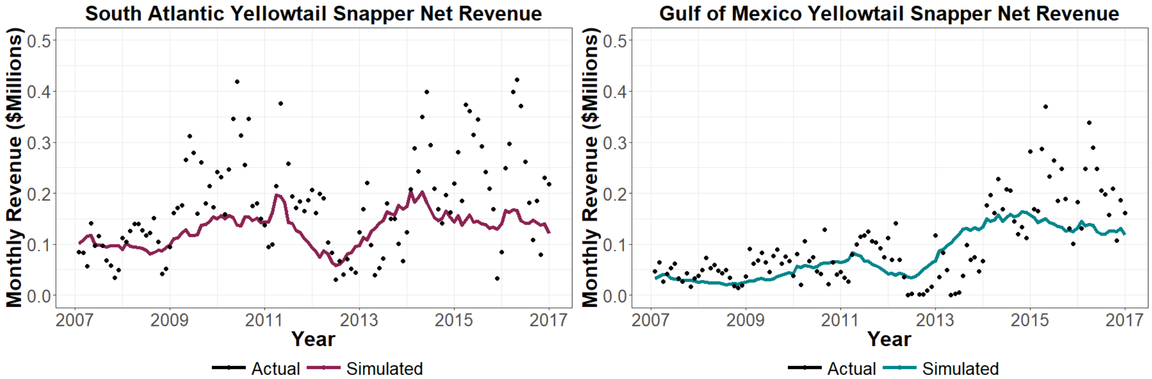
(a) South Atlantic

(b) Gulf of Mexico

Figure 4.11: Simulated gag grouper costs (lines) were calculated individually for each permit type then aggregated by federal jurisdiction: South Atlantic and Gulf of Mexico. The simulated costs followed the patterns observed in the actual cost data (black dots).

The simulated number of trips per fleet was translated to fishing costs through Equation 4.23, incorporating fleet-specific parameters including monthly averages of vessel lengths and days per trip. Costs were allocated to the jurisdiction where the permits were distributed and validated with observed costs (Figure 4.11). Finally, costs were subtracted from the aggregate revenues to obtain estimates of net revenue for each fleet and management jurisdiction (Figure 4.12).

In conclusion, these functions fit the observed data well, especially considering the model was built around an annual mortality estimator. Florida's primary source of yellowtail snapper revenue came from the South Atlantic (Figure 4.8), while the primary source of revenue from gag grouper came from the Gulf of Mexico (Figure 4.13). From 2012–2016, the Gulf of Mexico gag grouper model appeared to have conflicting signals in revenue sources. The simulated GM gag grouper revenue was greater than what was observed in the actual data (Figure 4.13), while the simulated jointly caught revenue was much lower than what was observed for this time frame (Figure 4.14). In 2012, the Gulf of Mexico reduced the minimum size limit, which



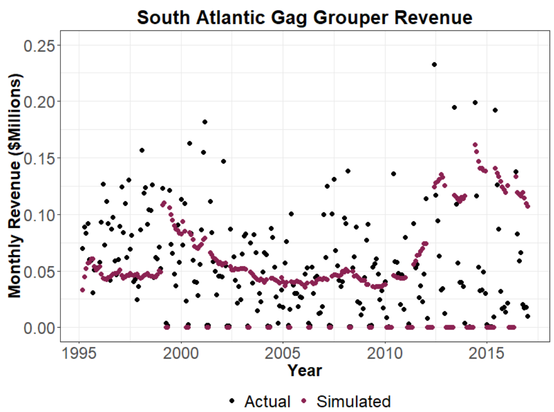
(a) South Atlantic

(b) Gulf of Mexico

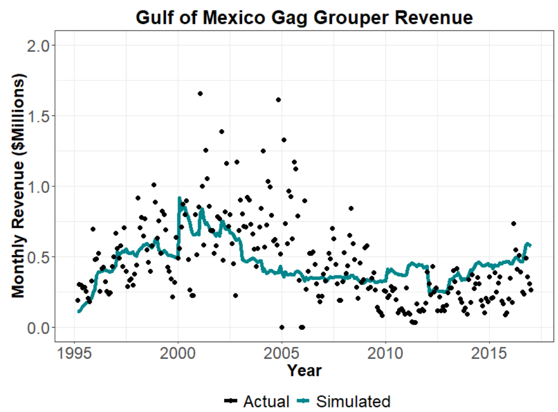
Figure 4.12: Simulated yellowtail snapper net revenue was estimated by subtracting the fleet costs from the sum of target and jointly caught revenue for each month. Fleets were allocated to their respective management jurisdiction accordingly.

could have influenced the ability of the average length estimator to predict mortality rates, resulting in observations conflicting predictions. On the other hand, if these trends were accurate, it could be indicative of a reduction in gag grouper recruits, a shift from gag grouper towards another target species, or both.

The bioeconomic model built in this Chapter sufficiently mapped the reality of actual biological and economic aspects of the Florida commercial reef fisheries. The actual management regimes were inputs to this model, and the results matched the conditions observed under these regulations. Therefore, regulations can be adjusted to simulate different management strategies to quantify bioeconomic impacts.

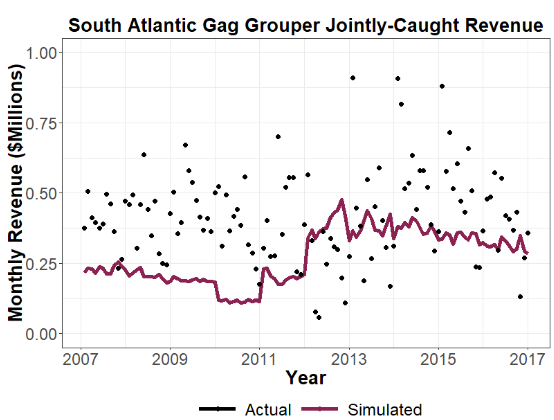


(a) South Atlantic

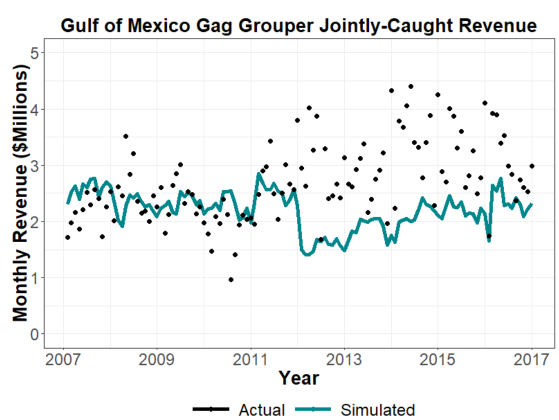


(b) Gulf of Mexico

Figure 4.13: Revenue was simulated by multiplying landings from the age-structured production model with the estimated ex-vessel price monthly. There have been closed seasons in the South Atlantic since 1999. The Gulf of Mexico implemented closed seasons for partial months 2000–2009, but never had closed the fishery for an entire month.



(a) South Atlantic



(b) Gulf of Mexico

Figure 4.14: Gag grouper jointly caught revenue was estimated for each fleet by species categories $\psi =$ Reef, Cobia & King/Spanish Mackerel, and Other. All of these estimates were summed up then plotted with observed data here.

CHAPTER 5

Retrospective Bioeconomic Assessment of Florida's Commercial Reef Fisheries

The dynamic age-structured bioeconomic model developed and validated in the previous Chapters was utilized to evaluate management strategies relative to sustainability and profitability goals (Figure 5.1).

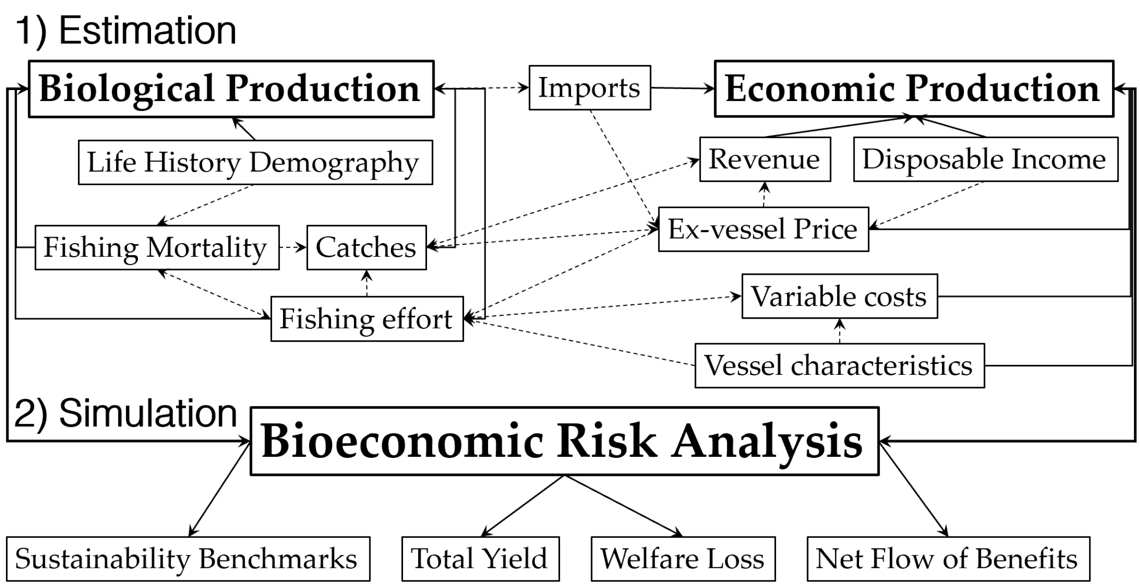


Figure 5.1: Biological and economic production of commercial fisheries are intertwined, and following (1) the estimation of parameters, (2) a bioeconomic risk analysis was simulated to identify optimal management strategies and characterize transitions to these optima.

5.1 Methods

5.1.1 Defining Optimal Management Strategies

The age-structured bioeconomic model validated in Chapter 4 was utilized to investigate biological and economic outcomes under any combination of fishing mortality rate, F, and minimum age at first capture, a_c. The time vector t was expanded to a matrix with dimensions $[i, j]$ where i was the vector of fishing mortality rates and j was the vector of ages of first capture. This translated the matrix $[a, t]$ with a set mortality rate and age at first capture to an array of dimension $[a, i, j]$ under all possible $F \in i$ and $a_c \in j$. Fishing mortality rate was allocated to the recreational and commercial fleets using the proportionality constant, τ_c, estimated in Equation 4.11, averaged 2014-2016.

$$F(a, i, j) = \tau_c \cdot F(a, i, j) + (1 - \tau_c) \cdot F(a, i, j) \tag{5.1}$$

Total mortality rate, $Z(a, i, j)$, was estimated as the sum of natural and fishing mortality rates. The fishing mortality rate i was only applied to ages a above the simulated age at first capture j.

$$Z(a, i, j) = \begin{cases} M(a, i, j) & \text{if } j < a_c \\ M(a, i, j) + F(a, i, j) & \text{if } j \geqslant a_c \end{cases} \tag{5.2}$$

These conditions resulted in a matrix of dimension $[i, j]$ of management strategies pairing all combinations of fishing mortality rates and ages at first capture. This matrix of management strategies was extended into a third ‘population dimension’ of height a_λ, the maximum age in months. Population biomass was estimated for each age class a in the population under every combination of $F \in i$ and $a_c \in j$,

$$\bar{B}(a,i,j) = \frac{N(a,i,j)}{Z(a,i,j)} \cdot (1 - e^{-Z(a,i,j)}) \cdot W(a) \tag{5.3}$$

where $W(a)$ parameters were defined in Table 2.8 and applied in Equation 2.2. Sustainability status was estimated by calculating the Spawning Potential Ratio (SPR) for all management simulations.

$$SPR(i,j) = \frac{\sum_{a=a_m}^{a_\lambda} \bar{B}(a,i,j)}{\sum_{a=a_m}^{a_\lambda} \bar{B}(a,0,j)} \tag{5.4}$$

The numerator represents the spawning stock biomass (SSB) under every management strategy, and the denominator defines SSB with no fishing mortality, $i = 0$. Total yield was estimated for each $[i,j]$ management strategy.

$$\hat{Y}(i,j) = \sum_{a=1}^{a_\lambda} F(a,i,j) \cdot \bar{B}(a,i,j) \tag{5.5}$$

Catch biomass was allocated to the commercial sector by multiplying the fishing mortality rate by the estimated proportion of fishing mortality due to the commercial fleets, τ_c.

$$\hat{Y}_c(i,j) = \sum_{a=1}^{a_\lambda} \tau_c \cdot F(a,i,j) \cdot \bar{B}(a,i,j) \tag{5.6}$$

Estimated commercial yield, $\hat{Y}_c(i,j)$, was used as an input into the inverse demand functions defined in Equation 3.6 to estimate ex-vessel price, which was multiplied by commercial yield to estimate 'analysis' species revenue, $R_s(i,j)$,

$$\hat{R}_s(i,j) = \hat{Y}_c(i,j) \cdot \left[\beta_0 + \beta_1 \cdot [\hat{Y}_c(i,j)] + \beta_2 \cdot x_2 + \beta_3 \cdot x_3 + \beta_4 \cdot x_4\right] \tag{5.7}$$

where coefficients β were reported in Tables 3.2–3.3, and the explanatory x variables that included import prices of substitute goods, implementation of the IFQ program, and disposable income were averages of variables 2014–2016. Number of trips by each fleet were estimated via Equation 4.21 for every combination of management strategies,

$$\hat{f}_g(i) = \frac{F(i) \cdot \nu_g}{q_g} \tag{5.8}$$

where ν_g was the proportion of fishing mortality rate attributed to each fleet estimated in Equation 4.17 and q_g was the catchability of each fleet estimated in Equation 4.19. Coefficients for the trip-level cost functions in Equation 3.25 were reported in Table 3.12. Explanatory variables for these functions included days per trip and vessel length, and 2014–2016 monthly averages for each fleet, Φ_{gr}, were used as inputs to the trip-level cost functions, transformed into real space (Table 3.11), then multiplied by the estimated number of trips per fleet for each management simulation, $\hat{f}_g(i)$.

$$\hat{C}_g(i) = [\beta_{g0} + \beta_{gr} \cdot \Phi_{gr}]^2 \cdot \hat{f}_g(i) \tag{5.9a}$$

$$\hat{C}_g(i) = exp[\beta_{g0} + \beta_{gr} \cdot \Phi_{gr}] \cdot \hat{f}_g(i) \tag{5.9b}$$

'Analysis' species Net revenue, $\hat{\Pi}_s$, was defined under all management simulations, resulting in a matrix of $[i, j]$ estimates of net revenue from the 'analysis' species only (i.e. not accounting for revenue from jointly-caught species).

$$\hat{\Pi}_s(i, j) = \hat{R}_s(i, j) - \sum\nolimits_g \hat{C}_g(i) \tag{5.10}$$

Finally, this matrix of management strategies was investigated relative to sustainability and profitability goals. The simulated surfaces were used to locate potentially optimal management strategies to be implemented in 1998 and compare their effectiveness relative to the actual management regime. Four management scenarios were identified: (1) a baseline management regime in 1998, $[i = F_{1998}\ ,\ j = a_{c,1998}]$; (2)

regulations currently in place, $[i = F_{2016}\ , j = a_{c,2016}]$; (3) maximizing revenue under current fishing mortality rates by adjusting age at first capture, $\max(\hat{R}_s[i = F_{1998}\ , j])$; (4) optimizing target species net revenue by adjusting mortality rate and age at first capture, $\max(\hat{\Pi}_s[i, j])$.

5.1.2 Simulating Transitions to Optimal Management Conditions

Four management strategies identified in Section 5.1.1 were simulated in the bioeconomic model defined in Section 4.1.3 beginning implementation in 1998: (1) 1998 baseline [1998], (2) current regulations [2016], (3) maximizing revenue [MR], and (4) maximizing net revenue [MNR]. Management strategies were implemented in 1998 to investigate where the resource could have been today if immediate action was taken 20 years ago when the resource was estimated to be overcapitalized (Ault *et al.* , 1998). Effectiveness of the actual management regime was examined relative to a baseline scenario of no change since 1998 and the two strategies that sought to optimize economic goals.

Time series of biological and economic metrics were simulated including population size, total catch, commercial catch, ex-vessel price, 'analysis' species revenue, jointly-caught revenue, cost, and net revenue through models outlined in Chapters 3 & 4. A discount rate of 3% was applied to all cost and revenue projections beginning in 1998 to estimate the Net Present Value (NPV) of each scenario.

$$\hat{R}_g^d(t) = \frac{\hat{R}_g(t)}{(1+0.03)^t} \qquad \hat{C}_g^d(t) = \frac{\hat{C}_g(t)}{(1+0.03)^t} \tag{5.11}$$

5.1.3 Evaluating Biological and Economic Benchmarks

Biological and economic benchmarks were calculated for all management scenarios to evaluate which strategies could have resulted in the most sustainable population sizes, the greatest economic benefits for the industry, and potential trade-offs between simulations. The fishing mortality rate that attains maximum sustainable yield, F_{MSY}, and associated population biomass, B_{MSY}, were used as benchmarks to determine if a population was undergoing overfishing or was currently overfished, respectively. Overfishing was defined as $F(t)/F_{MSY} > 1$, and overfished populations were defined as $\bar{B}(t)/\bar{B}_{MSY} < 1$.

Economic viability was assessed through estimation of NPV and welfare loss or gain relative to the baseline simulation. Net present value was used to assess how valuable the resource would be today under each management strategy considering other potential investments. NPV was estimated for the total costs and the revenue generated by the 'analysis' species s alone because it incorporated biological realism relevant to population growth.

$$NPV_{ms} = \sum_t \sum_g \left(\hat{R}^d_{gs}(t) - \hat{C}^d_g(t) \right) \tag{5.12}$$

NPV was also estimated for the sum of aggregate revenue minus cost because it represented a more realistic range of NPV of the commercial fisheries' net revenue.

$$NPV_m = \sum_t \sum_g \left(\hat{R}^d_{gs}(t) + \hat{R}^d_{g\psi}(t) - \hat{C}^d_g(t) \right) \tag{5.13}$$

The percent deviation from the baseline 1998 strategy was calculated and yielded welfare loss or gain compared to other management strategies considering the 'analysis' species only, WL_{ms}, and considering total net revenue WL_m.

$$WL_{ms} = \frac{[NPV_{ms} - NPV_{1998,s}]}{NPV_{1998,s}} \qquad WL_m = \frac{[NPV_m - NPV_{1998}]}{NPV_{1998}} \tag{5.14}$$

In these definitions, $NPV_{1998,s}$ referred to the baseline strategy of NPV considering the 'analysis' species only, and NPV_{1998} referred to the overall NPV of the baseline strategy.

Table 5.1: Variables required for the retrospective bioeconomic simulation model were compiled and defined here.

Variable	**Definition**	**Units**	**Equation**
a	Population age vector	months	
t	Time vector	months	
i	Fishing mortality rate vector		
j	Age at first capture vector	months	
$[i, j]$	Management strategy matrix		
$F(a, i, j)$	Fishing mortality rate		5.1
$Z(a, i, j)$	Total mortality rate		5.2
$\bar{B}(i, j)$	Average population biomass	pounds	5.3
$SPR(i, j)$	Spawning Potential Ratio		5.4
$\hat{Y}(i, j)$	Yield	pounds	5.5
$\hat{Y}_c(i, j)$	Commercial yield	pounds	5.6
$\hat{R}(i, j)$	Target species revenue	2016$	5.7
$\hat{f}_g(i)$	Fleet fishing effort	trips	5.8
$\hat{C}_g(i)$	Fleet costs	2016$	5.9
$\hat{\Pi}_s(i, j)$	Target net revenue	2016$	5.10
$\hat{R}_g^d(t)$	Discounted revenue of fleet g at time t	2016$	5.11
$\hat{C}_g^d(t)$	Discounted cost of fleet g at time t	2016$	5.11
F_{MSY}	Fishing mortality rate at MSY		
B_{MSY}	Population biomass at MSY	pounds	
NPV_{ms}	NPV of species s only for strategy m	2016$	5.12
NPV_m	NPV of strategy m	2016$	5.13
$NPV_{1998,s}$	NPV of species s only for baseline strategy	2016$	
NPV_{1998}	NPV for baseline strategy	2016$	

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Variable	Definition	Units	Equation
WL_{ms}	Welfare loss or gain of strategy m (species s only)		5.14
WL_m	Welfare loss or gain of strategy m		5.14

5.2 Results

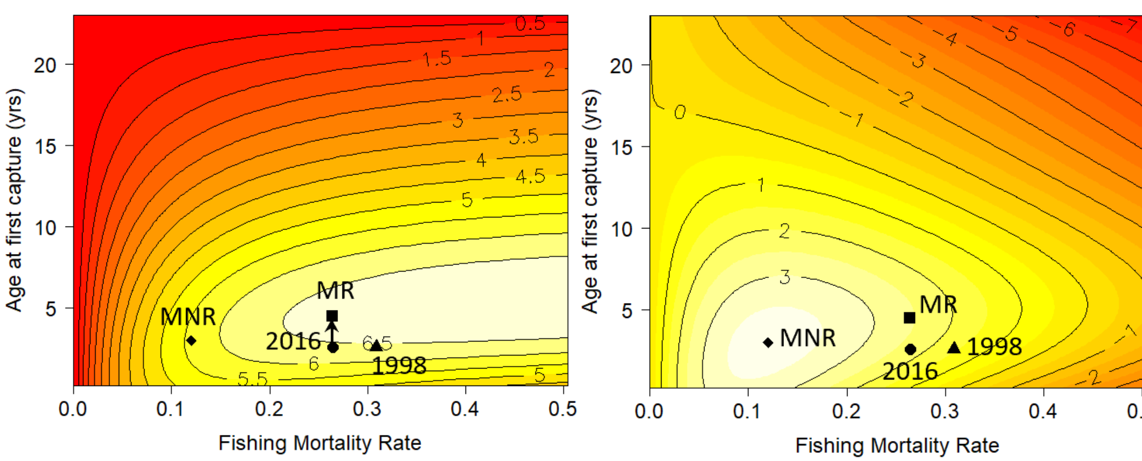
5.2.1 Optimal Management Simulations

Optimal management strategies were identified within a matrix of $[i, j]$ possible combinations of $F \in i$ ranging from no fishing mortality up to five times natural mortality rates and $a_c \in j$ from age at first capture of 1 month up to maximum age, a_λ. Four management strategies defined in Section 5.1.1 were simulated in the bioeconomic model defined in Section 4.1.3 for yellowtail snapper and Gulf of Mexico gag grouper: (1) 1998 baseline [1998], (2) current regulations [2016], (3) maximum revenue [MR], and (4) maximum net revenue [MNR].

Yellowtail snapper 1998 baseline conditions were $F = 0.312, a_c = 2.50$, which resulted in the smallest population size, smallest SPR, and lowest net revenue out of all simulations (Table 5.2). Current 2016 conditions were $F = 0.264, a_c = 2.50$ and improved all metrics, but raising the minimum size limit to $a_c = 4.50$ improved these metrics further, and resulted in an increase from \$6.40 million to \$6.70 million annual revenue. The management strategy that maximized net revenue (MNR) for yellowtail snapper was associated with the lowest costs, leading to the highest net target revenue (Figure 5.2). The strategies designed to optimize economic benefits also resulted in the best population metrics; SPR increased from the baseline 22% up to 36.9% under MR and up to 47.5% under MNR (Table 5.2).

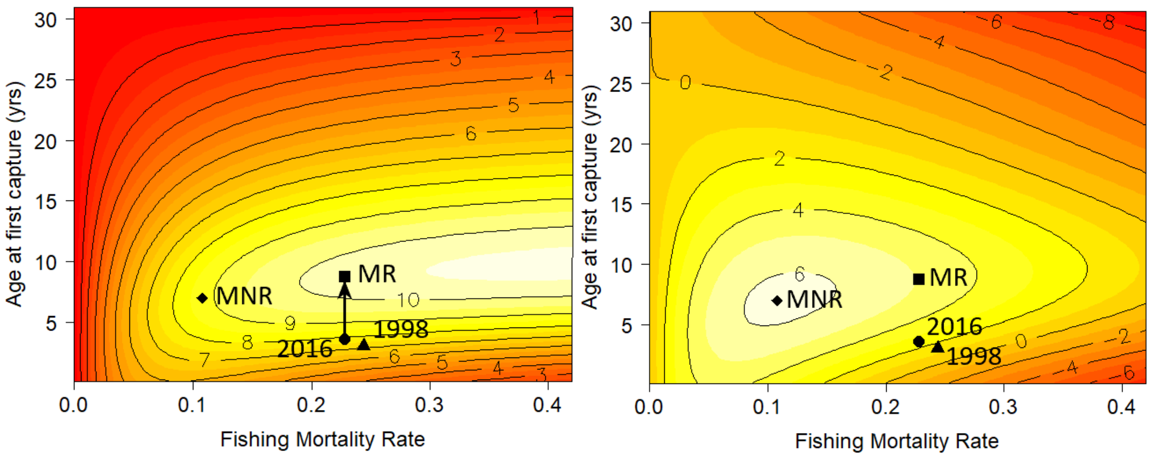
Table 5.2: Annual fishing mortality rate F and age at first capture a_c were the primary constraints for the four management strategy simulations. Revenue, cost, and net revenue were reported in millions of dollars per year, and standing population biomass, $\bar{B}$, was reported in millions of pounds. The best management strategy was bolded for every metric: highest revenue, lowest cost, highest net revenue, largest population, and highest spawning potential ratio.

	Yellowtail Snapper, M=0.130				**Gulf Gag Grouper, M=0.097**			
	1998	2016	MR	MNR	1998	2016	MR	MNR
a_c (years)	2.58	2.58	4.50	3.00	3.04	3.63	8.75	7.00
L_c	12”	12”	15”	13”	20”	22”	60”	32”
F	0.312	0.264	0.264	0.12	0.240	0.228	0.228	0.108
Revenue	\$6.41	\$6.40	**\$6.70**	\$5.62	\$7.14	\$7.89	**\$10.25**	\$8.78
Cost	\$4.89	\$4.14	\$4.14	**\$1.88**	\$5.59	\$5.31	\$5.31	**\$2.51**
Net Revenue	\$1.52	\$2.27	\$2.57	**\$3.74**	\$1.55	\$2.59	\$4.94	**\$6.26**
$\bar{B}$ (pounds)	9.6	10.7	14.9	**18.6**	15.7	19.1	42.6	**51.8**
SPR	22.0%	25.3%	36.9%	**47.5%**	13.2%	16.4%	39.2%	**48.2%**



(a) **Revenue (\$Millions)** (b) **Net Revenue (\$Millions)**

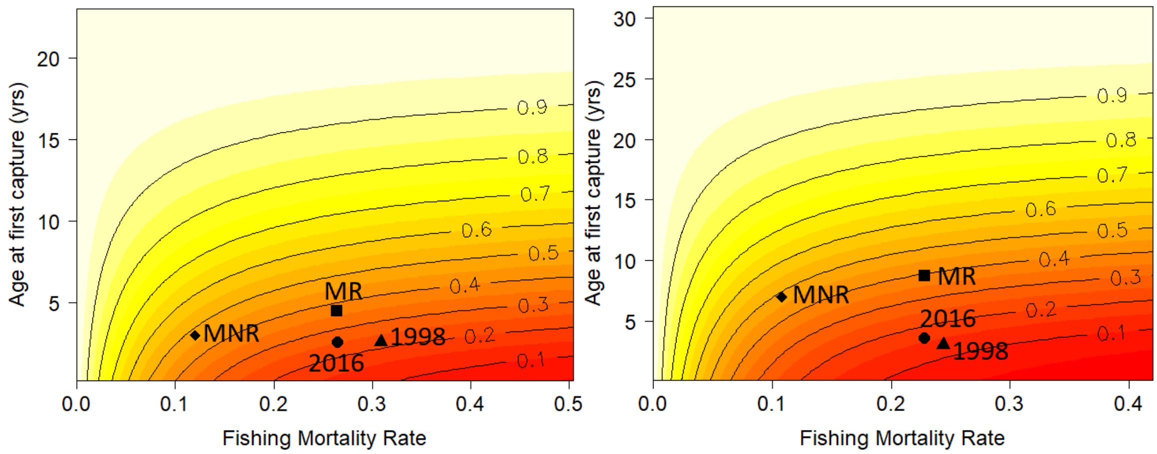
Figure 5.2: (a) Yellowtail snapper age at first capture was increased under F_{2016} to maximize revenue (MR). (b) The strategy that maximized net revenue (MNR) required a reduction in effort which reduced costs. Contour lines represent millions of dollars annually.



(a) **Revenue ($Millions)** (b) **Net Revenue ($Millions)**

Figure 5.3: (a) Gag grouper age at first capture was increased under F_{2016} to maximize revenue (MR). (b) The strategy that maximized net revenue (MNR) required a reduction in effort which reduced costs. Contour lines represented millions of dollars annually.

Gag grouper 1998 baseline conditions were $F = 0.240, a_c = 3.04$ and represented the lowest revenue, highest cost, lowest net revenue, and smallest population metrics of any simulated management strategy (Table 5.2). Under 2016 conditions, the minimum size limit was raised and effort was reduced, improving all of these metrics. To maximize revenue under F_{2016}, age at first capture had to be increased by nearly 6 years and resulted in $3.11 million more than the baseline strategy per year (Figure 5.3). Maximizing net revenue of the commercial fleet would require cutting fishing effort in half to reduce costs and increasing the age at first capture by 4 years over the baseline strategy. The MNR strategy resulted in $6.26 million per year in net revenue, a more than 300% increase over the baseline strategy. Gulf of Mexico gag grouper management simulations followed the same trends in sustainability displayed in yellowtail snapper. The two strategies designed to maximize profitability were also characterized by the highest SPR (Figure 5.4).



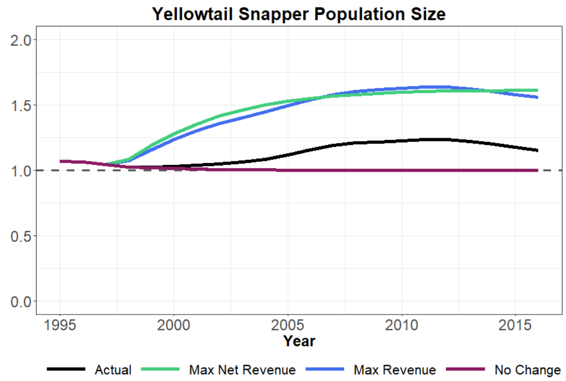
(a) **Yellowtail Snapper** (b) **Gag Grouper**

Figure 5.4: Spawning potential ratio was calculated for every combination of simulated age at first capture and fishing mortality rate. Management strategies designed to optimize economic benefits, MNR and MR, yielded the highest spawning potential for both (a) yellowtail snapper and (b) gag grouper.

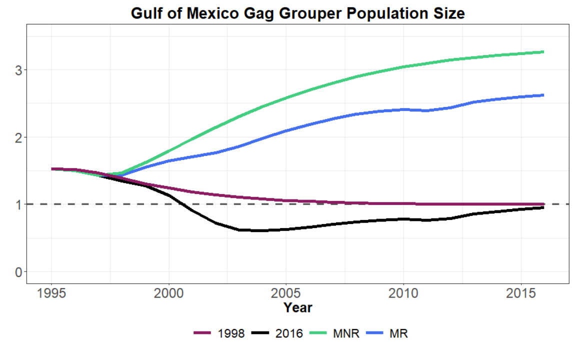
5.2.2 Retrospective Bioeconomic Evaluation of Management

Optimal management strategies identified in Table 5.2 were implemented beginning in 1998 within the bioeconomic simulation model (Chapter 4). The values in Table 5.2 were calculated under equilibrium conditions, while metrics presented in this section represented how the 1998 population would have responded, given its exploitation history, and where these metrics would have been by 2016 for each management strategy.

All simulated management strategies resulted in larger populations compared to the baseline for yellowtail snapper, while gag grouper's actual management regulations 1998-2016 were the only conditions that performed worse than the baseline (Figure 5.5). This showed that in some cases, preventing an effort increase was a management strategy in itself, making the baseline strategy somewhat misleading.



(a) Yellowtail Snapper

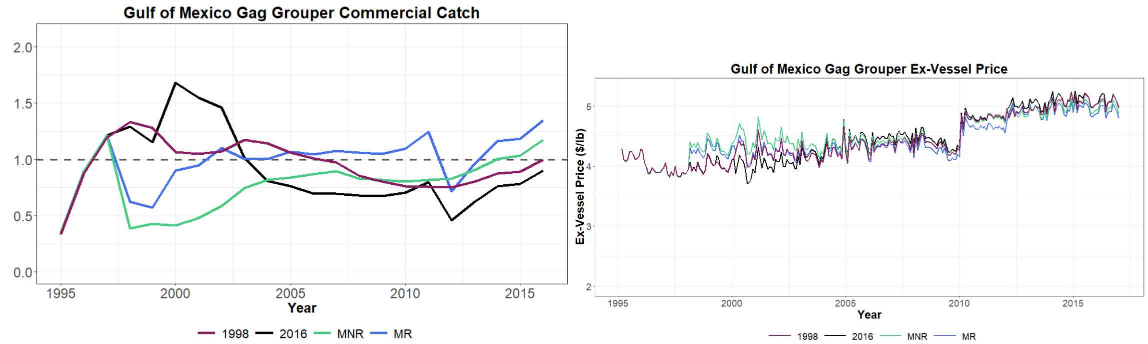


(b) Gag Grouper

Figure 5.5: Population size for each management simulation was standardized to the baseline 1998 strategy in the most recent year, 2016. Maximum Net Revenue (MNR) and Max Revenue (MR) resulted in the largest population sizes for yellowtail snapper and gag grouper.

Management strategies designed to optimize economic benefits resulted in the largest population sizes for both yellowtail snapper and gag grouper, where MNR population had more than tripled the baseline case by 2016.

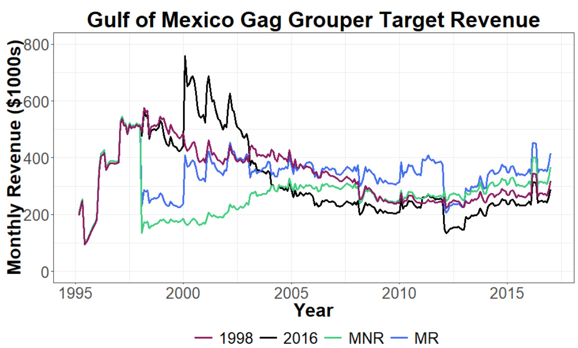
During rebuilding phases of the MR and MNR strategies that allowed the populations to more than double the baseline, catches were initially reduced, then surpassed the baseline 1998 catch by the end of the simulation time frame (Figure 5.6). Prices for these strategies spiked with the decrease in landings (increase in demand), then decreased once landings increased. 'Analysis' species' revenues were calculated by multiplying these two metrics, which followed a pattern similar to that observed in commercial yield (Figure 5.7). All economic data were discounted from 1998 to estimate net present value of each management strategy. Estimated variable costs were identical for 2016 and MR strategies because, by definition, the fishing mortality rates were equal for these simulations. Variable costs of the MNR strategy plummeted and remained low throughout the simulation time frame. Because of these



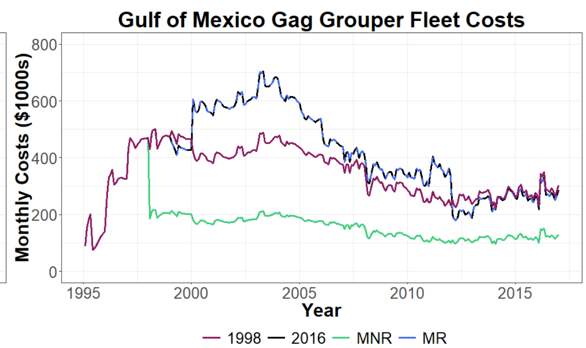
(a) Commercial Yield (b) Ex-Vessel Price

Figure 5.6: Gag grouper commercial yields were standardized to the baseline 1998 strategy in the most recent year of the simulation. When catches for MR and MNR strategies were reduced, prices increased. By the end of the simulation, populations were rebuilt, catches were back to the baseline level, and ex-vessel prices decreased accordingly.

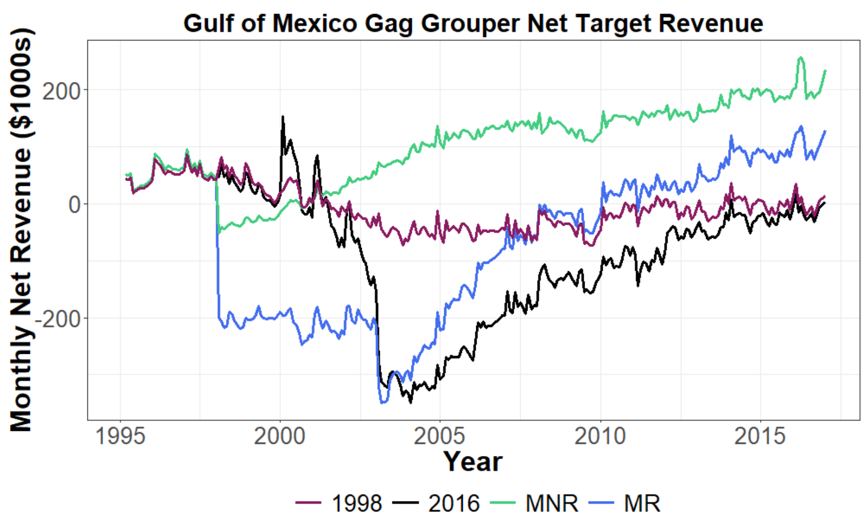
extremely low costs, discounted net revenue from the 'analysis' species immediately began to climb when this strategy was implemented, particularly when the population was sufficiently rebuilt allowing catch to increase. The gag grouper fleet relies heavily upon jointly-caught species as a source of revenue, therefore most simulated strategies resulted in a negative discounted net revenue because the revenue from gag grouper was not sufficient to cover the deficit of variable costs (Figure 5.7). Yellowtail snapper strategies followed similar patterns as gag grouper, except due to the yellowtail snapper fleets' high dependence on the 'analysis' species itself, the yellowtail-only discounted net revenues never dropped below zero (Figure 5.8).



(a) Gag Grouper Revenue

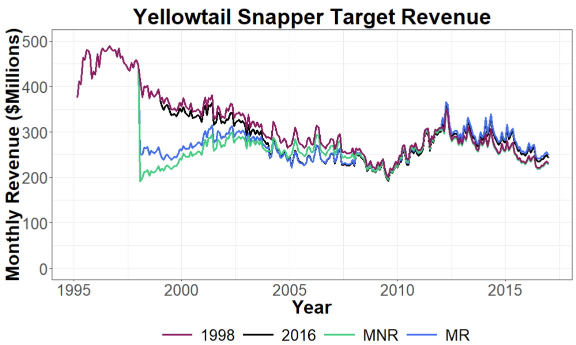


(b) Gag Grouper Fleet Variable Costs

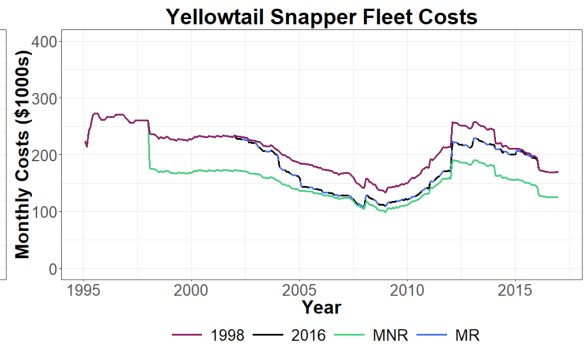


(c) Net Revenue from Gag Grouper Only

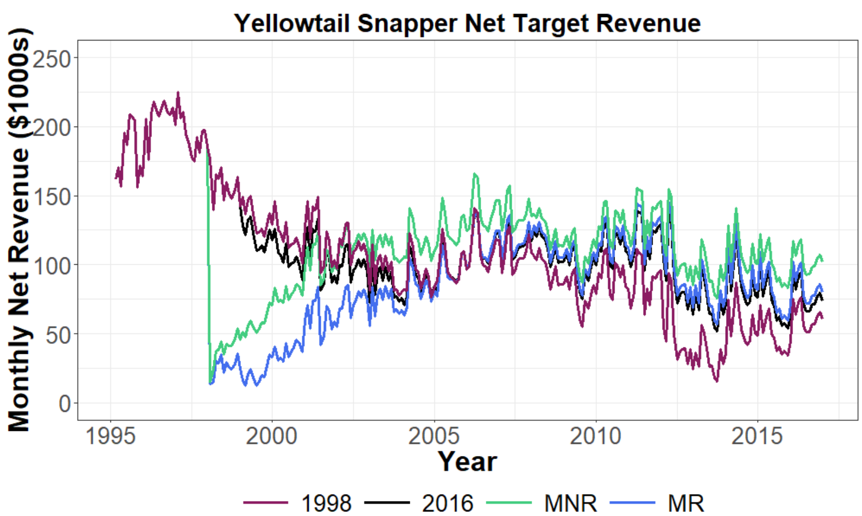
Figure 5.7: Gag grouper revenue, variable cost, and gag-only net revenue were discounted 3% from 1998, when the management simulations began, to estimate the net present value of each management strategy. The MNR strategy included an immediate drop in costs and temporary decrease in revenue, which resulted in the highest discounted net revenue for nearly every point in these simulations.



(a) Yellowtail Snapper Revenue



(b) Yellowtail Snapper Fleet Variable Costs



(c) Net Revenue from Yellowtail Snapper Only

Figure 5.8: Yellowtail snapper revenue, variable cost, and yellowtail-only net revenue were discounted 3% from 1998, when the management simulations began, to estimate the net present value of each management strategy. The MNR strategy included an immediate drop in costs and temporary decrease in revenue, which resulted in the highest discounted net revenue by approximately 2003.

5.2.3 Biological and Economic Benchmarks

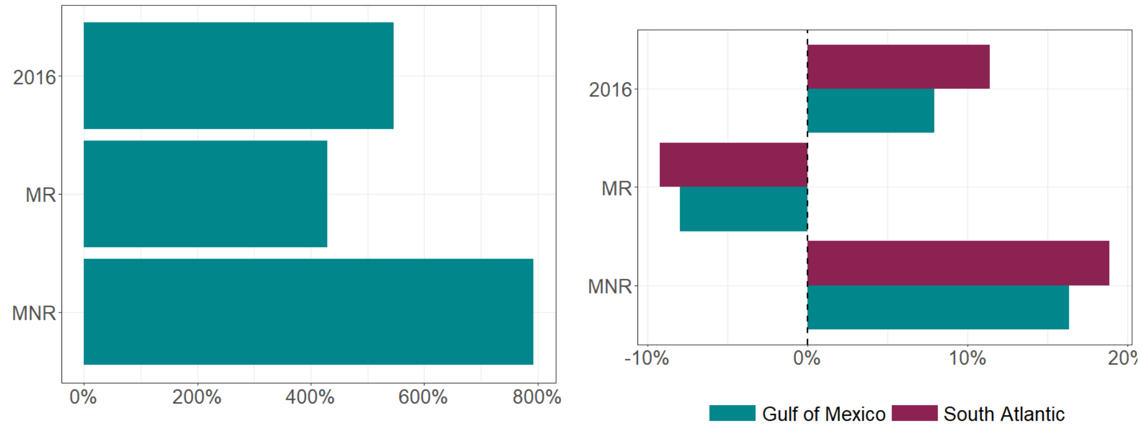
Biological and economic benchmarks were calculated for all management strategies to assess their bioeconomic performance of each stock assessed here. Net present value for each management strategy was calculated for 'analysis' species only revenue (NPV_{ms}), which represented a more realistic indicator of the best strategy related to management of the 'analysis' species, and for aggregate revenue (NPV_m), which represented a more realistic estimate of the magnitude of total net present value of each strategy (Table 5.3). Total net present value was not utilized to determine the best management strategy because it does not incorporate demographics and nearly always favors increasing effort to attain more revenue from jointly-caught species. Conversely, it was shown in the simulated strategies, if effort was relaxed, the stock grew and allowed for a higher catch-per-unit-effort (i.e. larger revenue for lower costs). This is impossible to model for jointly caught species without incorporating demographics information.

Gag grouper maximum net revenue strategy attained the largest 'analysis' species only NPV at \$24.51 million, and total NPV_m ranged \$198.68–\$390.03 million. Yellowtail snapper also attained the highest NPV_{ms} for the maximum net revenue simulation at \$24.54 million, and NPV_m was valued between \$82.68–\$140.27 million NPV. Both the gag grouper and yellowtail snapper recommended strategies were associated with the lowest aggregate NPV_m due to the reduction of fishing effort. Therefore, these may be on the lower end of NPV_m strategies, but, depending on the demographics of the jointly-caught species, a similar increase may be observed.

Table 5.3: Simulated net present value for the 'analysis' species only (NPV_{ms}) and for the aggregate net revenue (NPV_m) for each simulated management strategy. These values have been discounted by 3% monthly since 1998, and summed 1998–2016. The NPV_m trends (considering aggregate revenue) may not be biologically accurate because there was no penalty for infinitely increasing effort and were reported to represent the ranges of possible net present value of the fisheries.

	Gag Grouper		**Yellowtail Snapper**	
Strategy	NPV_{ms}	NPV_m	NPV_{ms}	NPV_m
1998	-$3.54	$348.33	$23.22	$121.10
2016	-$22.86	$385.89	$19.18	$117.06
MR	-$18.72	$390.03	$21.04	$140.27
MNR	**$24.51**	$198.68	**$24.54**	$82.68

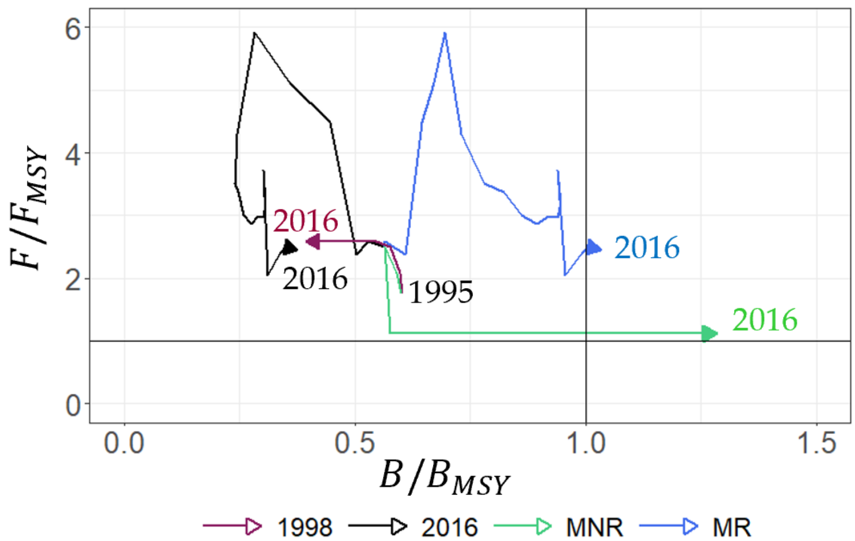
Welfare loss or gain was estimated using NPV_{ms} relative to the baseline simulation summed over 1998-2016. For Gulf of Mexico gag grouper, all strategies were an improvement over the baseline strategy (Figure 5.9). MNR strategy resulted in a welfare gain of 793% over the baseline simulation. For yellowtail snapper, the welfare gain was highest for the MNR strategy as well, with an increase of 19% in the South Atlantic and 16% in the Gulf of Mexico (Figure 5.9). For the MR strategy where size limit was increased but fishing mortality remained the same as actual levels, increases in catches weren't enough to make up for short-term losses during the rebuilding phase compared to the 2016 strategy, despite this strategy yielding the second highest net revenue in 2016.



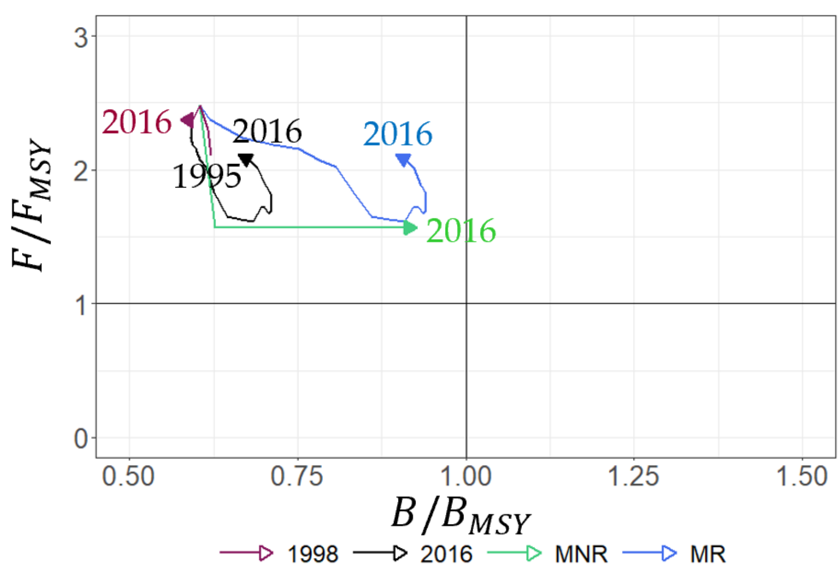
(a) Gag Grouper (b) Yellowtail Snapper

Figure 5.9: Gag grouper realized a huge welfare gain over the baseline across all simulations, with the largest gain realized under MNR. Yellowtail snapper also attained the largest welfare gain under MNR, but the MR strategy resulted in a welfare loss due to the short-term losses in the first 10 years following implementation.

Yellowtail snapper remained in an overfished state undergoing overfishing for all management strategies by the end of the simulation time frame (Figure 5.10). The strategies designed to optimize economic performance were the most sustainable strategies as defined by biological benchmarks. Gag grouper management strategies designed to optimize economic performance were the only two not in an overfished state by the end of the simulation. Across all target species simulated here, the management strategies designed to optimize revenue or net target revenue also performed the best biologically.



(a) **Gag Grouper**



(b) **Yellowtail Snapper**

Figure 5.10: 1995 marked the start point for the population simulations. Management strategies were implemented in 1998, where the simulations diverged and ended at 2016. Management strategies designed to optimize commercial economic benefits resulted in the most sustainable populations, as defined here, across every simulation.

CHAPTER 6

Conclusions

Fisheries management often has seemingly conflicting goals, but the disparity between biological sustainability and economic profitability is not one of them. In this study, strategies designed to optimize economic benefits were also the most sustainable across all species and regions, as defined by Spawning Potential Ratio (SPR). This finding aligns with the theoretical definition of Maximum Economic Yield (MEY), which is found at a lower rate of fishing effort–and larger population biomass–compared to that of Maximum Sustainable Yield (MSY), the peak yield of the surplus production model. MEY is theoretically achieved where the distance between the linear cost function and parabolic revenue function is maximized, but does not consider age structure of the population or other users of the resource (Clark, 1975; Hoshino *et al.* , 2018). Fisheries scientists have been modeling age-structured populations that incorporate minimum size limits for decades, a common management strategy used to protect spawning stock biomass and maximize yield (Pope, 1972; Methot, 1986; Methot & Wetzel, 2013). The bioeconomic analyses conducted here incorporated age-structured populations and partitioned the fishing mortality rates between recreational and commercial sectors, as well as among commercial fleets, which were characterized by varying catchabilities and cost functions. Recreational effort contributed to approximately 25% of the yellowtail snapper catch and 70% of the gag grouper catch on average throughout the simulation time frame. Despite these recreational constraints varying by nearly a factor of 3, all simulations of optimal commercial economic output resulted in the largest populations, achieved through various combinations of increased minimum size limits and effort reductions. Optimal economic productivity is dependent on biological sustainability, but the emerging picture of Florida reef fish sustainability has become increasingly dire as species' life histories are better studied and maximum age estimates continue to increase.

Commercial fishery data collected since the 1980s consistently sampled larger–and likely older–fish than were represented in the scientific literature, suggesting that parameter estimates derived from life history studies may not accurately characterize population dynamics. Maximum age uncertainty can only extend in one direction, as the species' true maximum age can only be greater than or equal to the currently observed maximum age (Ault *et al.* , 2018). Translating this to population dynamics implications, current estimates of natural mortality rates, M, are maximum estimates, and fishing mortality rates recommended with the intention of achieving maximum sustainable yield, $F \approx M$, could be creating a situation where a higher F is recommended than would be sustainable under the true M (Hoenig, 1983; Alagaraja, 1984; Ault *et al.* , 1998, 2018). Assessment models are always constrained by the best available data at the time, and in 2001, gag grouper were estimated to not be overfished or undergoing overfishing (Turner *et al.* , 2001). At that time, gag grouper maximum age was believed to be approximately 20 years old, which resulted in a higher estimated natural mortality rate and associated sustainable fishing mortality rate when compared to the estimated maximum age of 31 today. Furthermore, size-at-age data from 1992-2000 (SEFSC Panama City Lab) was used in that assessment, but by 1993 the gag grouper size distribution was already truncated, and the sex ratio was skewed down to 1% male from 17% male documented in the 1970s (Collins *et al.* , 1987; Hood & Schlieder, 1992; Coleman *et al.* , 1996). The confounding effects of mischaracterized demographics data on population dynamics models require conservative management strategies to allow reef fish populations to rebuild. If the species assessed here are at risk of overfishing, most species in the reef fish complex are also at risk due to historically high exploitation and low natural mortality rates. Often, when catch rates fall below an economically sustainable level, fisheries move from one species to another within the same ecosystem. In the reef ecosystem, the overall effect is continued pressure on an overexploited species as well as increased targeting on a less valuable species which may have otherwise been considered bycatch.

Management strategies have been to maintain historical catches, not rebuild resources, and it is likely we have been maintaining depleted resources rather than allowing populations to rebound. Based on the case studies investigated here, Florida reef fishes were even more severely depleted in 2016 than in 1998. A primary cause of this was increasing global population, which was exacerbated in Florida due its desirable climate and waters, with 85.1% of the state's population increase between 2010-2017 due to net migration rather than natural increase (BEBR, 2017). Increasing residential and tourist populations ultimately resulted in heavier exploitation rates on already over-exploited resources, despite attempts to manage these stocks primarily through Annual Catch Limits, size limits, seasonal closures, and spatial closures (US-DOC, 2007). Effort controls are in place in the form of bag limits and closed seasons but have not sufficiently reduced the mortality imposed on these stocks. Catch rates are so low, recreational bag limits are often never hit, despite groupers and snappers being a primary target, rendering this effort control obsolete. Under current minimum size limits, a large portion of the population is subjected to unsustainably high fishing mortality rates, and economic benefits to the commercial fleet have dissipated. Simulations conducted in this dissertation showed that when the minimum size limits were increased, sufficient portions of stock biomass were protected, even without reducing effort, resulting in increased long-term economic benefits for the commercial fleets while rebuilding the populations to more sustainable levels. Under the threat of a rapidly growing population and seemingly infinitely increasing demands for seafood and recreational fishing effort, this was a surprising and positive finding. Unfortunately, without effort reduction across the commercial and recreational sectors, initial declines in commercial economic productivity could be detrimental to this industry, as this simulated strategy resulted in a welfare loss for yellowtail snapper compared to the baseline simulation, despite its long-term gains.

One consistent result across all simulations was that strategies with higher net revenue was always associated with the larger population sizes, despite original population status or magnitude of recreational component. Under equilibrium conditions, all economically optimal (as defined here) management strategies were associated with an SPR close to 50%. Typically, SPR targets are set at 30%, but Ault *et al.* (2018) found that SPR at MSY for snappers and groupers in this region was approximately 40%, and the work here suggests an economically optimal benchmark for SPR may be closer to 50%. This aligns with the theory surrounding MEY, where a fishing mortality rate less than MSY optimizes profits. During transitional years towards optima, there were initial declines in economic productivity of the commercial fleets while populations rebuilt. Yellowtail snapper and gag grouper initial SPR and rates of recovery varied, but there was a 19–793% increase in welfare from the economically optimal strategies throughout the simulation time frame (1998–2016). Because of the Gulf of Mexico gag grouper's initially overexploited status, its population increased by more than a factor of 3 by the end of the simulations under the strategy designed to maximize net revenue. Despite differing regions, initial population status, recreational components, and life history characteristics, the conclusion that economically favorable management strategies resulted in healthier populations was robust across all simulations.

Fishery Management Councils were formed in the early 1980s to prevent the open-access nature of the fisheries that led to reduced catches and overexploited natural resources (GMFMC, 1981; SAFMC, 1983; CFMC, 1985). While the FMCs effectively excluded foreign fleets from accessing U.S. waters, the primary regulatory strategy for U.S. fleets was through quotas or annual catch limits that maintained suboptimal catches. The commercial reef fishery has been closed-access since 1992 in the Gulf of Mexico and since 1998 in the South Atlantic, but by this time, resources were already depleted. Open-access resources have resulted in overexploitation and overcapitalization of fisheries throughout history where commercial fleets essentially break even (Hoshino *et al.* , 2018). Rights- and privilege-based fishing is an effective way to more strictly define access to natural resources, a strategy that encourages investment and provides incentives to fish sustainably (Ostrom *et al.* , 1999; Sanchirico & Springborn, 2011; Solis *et al.* , 2014). Implementing the Individual Fishing Quota system in the Gulf of Mexico increased ex-vessel price by allowing fishermen to take advantage of favorable market conditions, rather than race to fish, and resulted in a $0.50 increase in ex-vessel price for gag grouper (Chapter 3). The Gulf of Mexico IFQ system also increased the technical efficiency and reduced costs for the reef fish fleets operating two primary gears modeled here, hook-and-line and longline (Solis *et al.* , 2014). Defining fishing privileges within the commercial sector promoted sustainable fishing practices, but the recreational sector in Florida remains essentially open-access. A recreational saltwater fishing license is required to fish in Florida, with tags required to land certain species, but there is currently no limit to the number of licenses sold annually.

In tropical marine ecosystems, diversity of commercial catches can prove overwhelming to properly manage every species. Eighty-four commercially important reef fishes were identified in U.S. waters surrounding Florida and the U.S. Caribbean Islands. Of these, just over half had demographics data to allow for parameterization of stock assessment models. Statistical methodologies have been developed to estimate life history parameters where no species-specific data are available. Nadon & Ault (2016) built regressions between parameters within familial groups which allowed for stochastic estimation of a full set of life history parameters from an initial estimate of maximum length. Thorson *et al.* (2017) developed another Bayesian estimation methodology, which requires knowledge of three life history variables to estimate the remaining set within any family. The literature synthesis conducted here promotes both methodologies and highlights potential pitfalls, such as weak parameter estimates or insufficient estimates of maximum growth potential. This synthesis also helps direct biological efforts to define demographics information for reef fishes where parameters are missing or unreliable. Federal reef fish assessments have been hindered by a lack of data and resources, but there has been substantial expansion of demographics information and progress within the data-limited assessment framework (Ault *et al.* , 2005a; Carruthers *et al.* , 2015; Nadon *et al.* , 2015; Sagarese *et al.* , 2018). While the Florida commercial reef fish fishery lands a diverse array of species, and the state imports the majority of the its available snappers and groupers, a few local species dominate the catch and drive the Florida seafood market. Within the commercial reef fish landings, 19 groupers and 17 snappers were identified to the species level, but 2 groupers (gag, red) and 3 snappers (yellowtail, red, vermilion) accounted for over approximately 85% of their respective Family's (Epinephelinae, Lutjanidae) total landings. Despite the strong multi-species component within this fishery, a few species drive the profitability of the industry. This finding supports a management strategy of controlling effort through constraints defined by these primary species, and setting minimum size limits for the rest of the complex above their respective length at maturity, allowing for reproduction and contribution to future generations.

In the valuable reef ecosystem, the health of all component parts are inherently intertwined due to the vast diversity of the system. If all species in the complex are determined to require management intervention, then a system-wide strategy would be beneficial to allow concerted population rebuilding. Simplistically, adaptive management supports implementing marine protected areas, which have the economic benefit of just not fishing in a certain area to reduce costs of fishing and potential catch and release mortality (Bohnsack, 1998). Another management strategy could be to adapt fishing gears and areas to effectively target larger fishes that allow for the growth, maturation, and reproduction of a species before allowing the fish to be susceptible to fishing mortality. Managing reef fish stocks by promoting maximum commercial economic productivity and sustainability could strengthen the Florida reef ecosystem to outside stressors including climate change and reef degradation while also strengthening an industry. The recreational and charter fisheries contribute substantially to the economic productivity of the reef ecosystem, and their utilities could be applied within this bioeconomic framework as well. Incorporating the utility of all stakeholders and balancing regulations to optimize net benefits allows for a more holistic evaluation of management strategies. From this work, management regulations producing bioeconomic optima can be identified for any region and fishery where data are available. Quantification of transitional years provides guidance for population rebuilding years and input to support long-term economic gains, while protecting against harming an industry. In conclusion, sustainable regulation of exploited fishes creates a stronger fishing industry, increases net benefits to the region, and maintains a healthier ecosystem.

Bibliography

Aiken, K.A. 1975. Chapter 15: The biology, ecology and bionomics of the triggerfishes, Balistidae. *In:* Munro, J.L. (ed), *Caribbean Coral Reef Fishery Resources*. Manila, Philippines: International Center for Living Aquatic Resources Management.

Alagaraja, K. 1984. Simple methods for estimation of parameters for assessing exploited fish stocks. *Indian Journal of Fisheries*, **31**(2), 177–208.

Anderson, B., Lombardi-Carlson, L., & Hamilton, A. 2009. Age and growth of wenchmand (*Pristipomoides aquilonaris*) from the northern Gulf of Mexico. *Proceedings of the Gulf and Caribbean Fisheries Institute*, **61**, 210–217.

Andrews, K., Nall, L., Jeffrey, C., & Pittman, S. 2005. *The State of Coral Reef Ecosystems of Florida*. Tech. rept. National Centers for Coastal Ocean Science.

Arnold, L.M., & Heppell, S.S. 2015. Testing the robustness of data-poor assessment methods to uncertainty in catch and biology: a retrospective approach. *ICES Journal of Marine Science*, **72**(1), 243–250.

Ault, J.S., Bohnsack, J.A., & Meester, G.A. 1998. A retrospective (1979-1996) multispecies assessment of coral reef fish stocks in the Florida Keys. *Fishery Bulletin*, **96**(3), 395–414.

Ault, J.S., Luo, J., Smith, S.G., Serafy, J.E., Wang, J.D., Humston, R., & Diaz, G.A. 1999. A spatial dynamic multistock production model. *Canadian Journal of Fisheries and Aquatic Sciences*, **56**(1), 4–25.

Ault, J.S., Smith, S.G., & Bohnsack, J.A. 2005a. Evaluation of average length as an estimator of exploitation status for the Florida coral-reef fish community. *ICES Journal of Marine Science*, **62**, 417–423.

Ault, J.S., Bohnsack, J.A., Smith, S.G., & Luo, J. 2005b. Towards sustainable multispecies fisheries in the Florida, USA, coral reef ecosystem. *Bulletin of Marine Science*, **76**(2), 595–622.

Ault, J.S., Smith, S.G., Luo, J., Monaco, M.E., & Appeldoorn, R.S. 2008. Length-based assessment of sustainability benchmarks for coral reef fishes in Puerto Rico. *Environmental Conservation*, **35**, 221–231.

Ault, J.S., Smith, S.G., & Tilmant, J.T. 2009. Are the coral reef finfish fisheries of south Florida sustainable? *Proceedings of the 11th International Coral Reef Symposium*, 989–993.

Ault, J.S., Smith, S.G., Browder, J.A., Nuttle, W., Franklin, E.C., Luo, J., DiNardo, G.T., & Bohnsack, J.A. 2014. Indicators for assessing the ecological dynamics and sustainability of souther Florida's coral reef and coastal fisheries. *Ecological Indicators*, **44**, 164–172.

Ault, J.S., Smith, S.G., Bohnsack, J.A., Luo, J., Stevens, M.H., DiNardo, G.T., Johnson, M.W., & Bryan, D.R. 2018. Length-based risk analysis for assessing sustainability of data-limited tropical reef fisheries. *ICES Journal of Marine Science*, **doi:10.1093/icesjms/fsy123**.

Baisre, J.A., & Paez, J. 1981. *Los recursos pesqueros del archipielago cubano*. Tech. rept. Programa Interregional de Ordinacion y Desarrollo Pesqueros.

Barten, A.P., & Bettendorf, L.J. 1989. Price formation of fish: An application of and inverse demand system. *European Economic Review*, **33**, 1509–1525.

BEBR. 2017 (April). *Florida Estimates of Population 2017*. Tech. rept. Bureau of Economic and Business Research, University of Florida, Gainesville, FL.

Beebe, W. 1928. *Beneath Tropic Seas: A Record of Diving Among the Coral Reef of Haiti*. New York and London: Knickerbocker Press and G.P. Putnam's Sons.

Beverton, R.J.H., & Holt, S.J. 1957. *On the Dynamics of Exploited Fish Populations*. Fisheries Investigations.

Billings, V.C., & Munro, J.L. 1974. The biology, ecology, and bionomics of Caribbean reef fishes: Pomadasydae (Grunts). *Res. Rep. Zool. Dep., Univ. West Indies*, **3**, 1–128.

Bohnsack, J.A. 1998. Application of marine reserves to reef fisheries management. *Australian Journal of Ecology*, **23**, 298–304.

Bohnsack, J.A., & Ault, J.S. 1996. Management strategies to conserve marine biodiversity. *Oceanography*, **9**(1), 73–82.

Bohnsack, J.A., & Harper, D.E. 1988 (December). *Length-weight relationships of selected marine reef fishes from the southeastern United States and Caribbean*. Tech. rept. NMFS-SEFC-215. National Oceanic and Atmospheric Administration, Miami, Florida.

Brander, L.M., Beukering, P.V., & Cesar, H.S.J. 2007. The recreational value of coral reefs: A meta-analysis. *Ecological Economics*, **61**(1), 209–218.

Bullock, L.H., & Murphy, M.D. 1994. Aspects of the life history of the yellowmouth grouper, *Mycteroperca interstitialis*, in the eastern Gulf of Mexico. *Bulletin of Marine Science*, **55**(1), 30–45.

Burton, M.L. 2001. Age, growth, and mortality of gray snapper, *Lutjanus griseus*, from the east coast of Florida. *Fishery Bulletin*, **99**, 254–265.

Burton, M.L. 2002. Age, growth, and mortality of mutton snapper, *Lutjanus analis*, from the east coast of Florida, with a brief discussion of management implications. *Fisheries Research*, **59**, 31–41.

Burton, M.L., Potts, J.C., & Carr, D.R. 2012. Age, growth, and natural mortality of rock hind, *Epinephelus adscensionis*, from the Gulf of Mexico. *Bulletin of Marine Science*, **88**(4), 903–917.

Burton, M.L., Potts, J.C., & Carr, D.R. 2014. Age, growth, and mortality of yellowmouth grouper from the southeastern United States. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science*, **6**, 33–42.

Burton, M.L., Potts, J.C., & Carr, D.R. 2015. Age, growth, and natural mortality of coney (*Cephalopholis fulva*) from the southeastern United States. *PeerJ 3:e825; DOI 10.7717/peerj.825.*

Burton, M.L., Potts, J.C., & Carr, D.R. 2016. Age, growth, and natural mortality of blackfin snapper (*Lutjanus buccanella*) from the southeastern United States and U.S. Caribbean. *Gulf and Caribbean Research*, **27**(1), 66–73.

Caballero-Arango, D, Brule, T, Noh-Quinones, V, Colas-Marrufo, T., & Perez-Diaz, E. 2013. Reproductive Biology of the Tiger Grouper in the Southern Gulf of Mexico. *Transactions of the American Fisheries Society*, **142**, 282–299.

Carruthers, T., Kell, L., Butterworth, D., Maunder, M., Geromont, H., Walters, C., McAllister, M., Hillary, R., Levontin, P., Kitakado, T., & Davies, C. 2015. Performance review of simple management procedures. *ICES Journal of Marine Science*, **73**, 464–482.

CFMC. 1985 (February). *Fishery Management Plan, Final Environmental Impact Statement, and Draft Regulatory Impact Review for the Shallow-water Reef Fish Fishery of Puerto Rico and the U.S. Virgin Islands.* Tech. rept. Caribbean Fishery Management Council, San Juan, Puerto Rico.

Choat, J.H., & Robertson, D.R. 2002. Age-based studies on coral reef fishes. *Pages 57–80 of:* Sale, P.F. (ed), *Coral Reef Fishes: Dynamics and Diversity in a Complex System.* Massachusetts, USA: Academic Press.

Choat, J.H., Robertson, D.R., Ackerman, J.L., & Posada, J.M. 2003. An age-based demographic analysis of the Caribbean stoplight parrotfish *Sparisoma viride*. *Marine Ecology Progress Series*, **246**, 265–277.

Clark, C.W. 1975. *Mathematical Bioeconomics, The Optimal Management of Renewable Resources.* New York: John-Wiley & Sons, Inc.

Claro, R. 1981. Ecologia y ciclo de vida del pargo criollo, *Lutjanus analis* (Cuvier), en la plataforma cubana. *Inf. Cient. Tec., Acad. Cienc. Cuba*, **186**, 1–83.

Claro, R., & Garcia, J.P. 2001. Growth patterns of fishes of the Cuban shelf. *In:* Claro, R. (ed), *Ecology of the Marine Fishes of Cuba.* Quintana Roo, Meixco: Inst. Oceanol. Cent. Invest. pp. 149-178.

Claro, R., & Garcia-Arteaga, J.P. 1994. Crecimiento. *In:* Claro, R. (ed), *Ecologia de los Peces Marinos de Cuba.* Quintana Roo, Mexico: Inst. Oceanol. Cent. Invest. pp. 321-402.

Claro, R., Garcia-Cagide, A., & de Alaiza, R. Fernandez. 1989. Caracteristicas biologicas del pez perro, *Lachnolaimus maximus* (Walbaum), en el golfo de Batabano, Cuba. *Revista Investigaciones Marinas*, **10**, 239–252.

Claro, R., Sierra, L.M., & Garcia-Arteaga, J.P. 1999. Biologia del jocu, *Lutjanus jocu* (Pisces: Lutjanidae) en las NE y SW de la plataforma Cubana. II. Alimentacion, edad y crecimiento. *Rev. Invest. Mar.*, **20**(1–2), 45–53.

Claro, R., Lindeman, K.C., & Parenti, L.R. 2001. *Ecology of the Marine Fishes of Cuba.* Smithsonian Institution.

Coleman, F.C., Koenig, C.C., & Collins, L.A. 1996. Reproductive styles of shallow-water groupers (Pisces: Serranidae) in the eastern Gulf of Mexico and the consequences of fishing spawning aggregations. *Environmental Biology of Fishes*, 129–141.

Collins, M.R., Waltz, C.W., Roumillat, W.A., & Stubbs, D.L. 1987. Contribution to the life history and reproductive biology of gag, *Mycteroperca microlepis* (Serranidae), in the South Atlantic Bight. *Fishery Bulletin*, **85**(3), 648–653.

Cooper, W., Collins, A., O'hop, J., & Addis, D. 2013. *The 2013 Stock Assessment Report for Hogfish in the South Atlantic and Gulf of Mexico.* Tech. rept. Florida Fish and Wildlife Conservation Commission, St. Petersburg, Florida.

Costello, C., Ovando, D., Clavelle, T., Strauss, C.K., Hilborn, R., Melnychuk, M.C., Branch, T.A., Gaines, S.D., Szuwalski, C.S., Cabral, R.B., Rader, D.N., & Leland, A. 2016. Global fishery prospects under contrasting management regimes. *Proceedings of the National Academy of Sciences of the United States of America*, **113**(18), 5125–5129.

Crabtree, R.E., & Bullock, L.H. 1998. Age, growth, and reproduction of black grouper, *Mycteroperca bonaci*, in Florida waters. *Fishery Bulletin*, **96**, 735–753.

Cunningham, S., Bennear, L.S., & Smith, M.D. 2016. Spillovers in regional fisheries management: Do catch shares cause leakage? *Land Economics*, **92**(2), 344–362.

Cushion, N.M. 2010. *Growth, reproductive life-history traits and energy allocation in Epinephelus guttatus (red hind), E. Striatus (Nassau grouper), and Mycteroperca venenosa (yellowfin grouper) (Family Serranidae, Subfamily Epinephelinae*. Ph.D. thesis, University of Miami.

de Albuquerque, C.Q., Martins, A.S., de Oliveira Leite Junior, N., de Araújo, J.N., & Ribeiro, A.M. 2011. Age and growth of the queen triggerfish *Balistes vetula* (Tetraodontiformes, Balistidae) of the central coast of Brazil. *Brazilian Journal of Oceanography*, **59**(3), 231–239.

de Araujo, J.N., & Martins, A.S. 2006. Age and growth of coney (*Cephalopholis fulva*), from the central coast of Brazil. *J. Mar. Biol. Ass. U.K.*, **86**, 187–191.

Dennis, G.D. 1988. Commercial catch length-frequency data as a tool for fisheries management with an application to the Puerto Rico trap fishery. *Memoria de la Sociedad de Ciencias Naturales La Salle*, 289–310.

Domeier, M.L., & Colin, P.L. 1997. Tropical reef fish spawning aggregations: Defined and reviewed. *Bulletin of Marine Science*, **60**(3), 698–726.

Dutka, J., & Murie, D.J. 2001. Age and growth of sheepshead, *Archosargus probatocephalus* (Pisces: Sparidae), from the northwest coast of Florida. *Bulletin of Marine Science*, **68**(1), 69–83.

Ehrhardt, N.M., & Ault, J.S. 1992. Analysis of two length-based mortality models applied to bounded length frequencies. *Transactions of the American Fisheries Society*, **121**(1), 115–122.

FAO. 2016. *The State of World Fisheries and Aquaculture 2016. Contributing to food security and nutrition for all.* Tech. rept. Rome.

Farmer, N.A., Heyman, W.D., Karnauskas, M., Kobara, S., Smart, T.I, Ballenger, J.C., Reichert, M.J.M., Wyanski, D.M., Tishler, M.S., Lindeman, K.C., Lowerre-Barbieri, S.K., Switzer, T.S., Soloman, J.J., McCain, K., Marhefka, M., & Sedberry, G.R. 2017. Timing and locations of reef fish spawning off the southeastern United States. *PLoS One*, **12**(3), e0172968.

Figuerola, M., Matos, D., & Torres, W. 1998. Maturation and reproductive seasonality of four reef fish species in Puerto Rico. *Proceedings of the Gulf and Caribbean Fisheries Institute*, **50**, 938–945.

Fischer, A.J., M.S. Baker, Jr., Wilson, C.A., & Nieland, D.L. 2005. Age, growth, mortality, and radiometric age validation of gray snapper (*Lutjanus griseus*) from Louisiana. *Fishery Bulletin*, **103**, 307–319.

Garcia, C.B., & Duarte, L.O. 2006. Length-based estmiates of growth parameters and mortality rates of fish populations of the Caribbean Sea. *Journal of Applied Ichthyology*, **22**, 193–200.

Garcia-Arteaga, J.P. 1983. Edad y crecimiento del jallao, *Haemulon album*, en la plataforma SW de Cuba. *Rep. Invest. Inst. Oceanol. Acad. Cienc. Cuba*, **14**, 1–25.

Garcia-Arteaga, J.P., Sierra, L.M., & Claro, R. 1999. Biologia del bonaci gato, *Mycteroperca tigris* (Pisces: Serranidae) en la plataforma SE de Cuba. II. Alimentacion, eded y crecimiento. *Rev. Invest. Mar.*, **20**((1-3)), 15–21.

Garcia-Cagide, A. 1986. Reproductive characteristics of the bluestriped grunt, *Haemulon sciurus*, in the eastern zone of the Batabano Gulf, Cuba. *In: Biological Sciences and Living Resources*.

Gaut, V.C., & Munro, J.L. 1974. Chapter 10: The biology, ecology, and bionomics of the grunts, Pomadasyidae. *In:* Munro, J.L. (ed), *International Center for Living and Aquatic Resources Management (ICLARM) studies and reviews*.

Gedamke, T., & Hoenig, J.M. 2006. Estimating mortality from mean length data in nonequilibrium situations, with application to the assessment of goosefish. *Transactions of the American Fisheries Society*, **135**, 476–487.

GMFMC. 1981. *Environmental Impact Statement and Fishery Management Plan for the Reef Fish Resources of the Gulf of Mexico*. Tech. rept. Gulf of Mexico Fishery Management Council, Tampa, FL.

GMFMC. 1989 (August). Tech. rept. Amendment Number 1 to the Reef Fish Fishery Management Plan, Tampa, FL.

GMFMC. 1991 (November). *Framework Adjustment to the Reef Fish Fishery Management Plan for the Reef Fish Resources of the Gulf of Mexico*. Tech. rept. Gulf of Mexico Fishery Management Council, Tampa, FL.

GMFMC. 1993 (February). *Amendment 5 to the Reef Fish Fishery Management Plan for the Reef Fish Resources of the Gulf of Mexico*. Tech. rept. Gulf of Mexico Fishery Management Council, Tampa, FL.

GMFMC. 1999 (August). *Regulatory Amendment to the Reef Fish Fishery Management Plan to Set 1999 Gag/Black Grouper Management Measures (Revised)*. Tech. rept. Gulf of Mexico Fishery Management Council, Tampa, FL.

Gobert, B., Guillou, A., Murray, P., Berthou, P., Turcios, M.D.O., Lopez, E., Huet, P.L.J., Diaz, N., & Gervain, P. 2005. Biology of queen snapper (*Etelis oculatus*: Lutjanidae) in the Caribbean. *Fishery Bulletin*, **103**, 417–425.

Haas, H. 1952. *Diving to Adventure*. London: Jarrolds, Ltd.

Hanna, S., Blough, H., Allen, R., Iudicello, S., Matlock, G., & McCay, B. 2000. *Fishing Grounds: Defining a New Era for American Fisheries Management*. Washington, D.C. and Covelo, California: Island Press.

Harris, P.J., Wyanski, D.M., White, D.B., & Moore, J.L. 2002. Age, growth, and reproduction of scamp, *Mycteroperca phenax*, in the southwestern north Atlantic, 1979–1997. *Bulletin of Marine Science*, **70**(1), 113–132.

Hewitt, D.A., & Hoenig, J.M. 2005. Comparison of two approaches for estimating natural mortality based on longevity. *Fishery Bulletin*, **103**, 433–437.

Hildebrand, S.F., & Cable, L.E. 1930. Development and life history of fourteen teleostean fishes at Beaufort, N.C. *Bull. U.S. Bur. Fish.*, **46**, 383–488.

Hoenig, J.M. 1983. Empirical use of longevity data to estimate mortality-rates. *Fishery Bulletin*, **81**(4), 898–903.

Hood, P.B., & Schlieder, R.A. 1992. Age, growth, and reproduction of gag, *Mycteroperca microlepis* (Pisces:Serranidae), in the eastern Gulf of Mexico. *Bulletin of Marine Science*, **51**(3), 337–352.

Horvath, M.L., Grimes, C.B., & Huntsman, G.R. 1990. Growth, mortality, reproduction and feeding of knobbed porgy, *Calamus nodosus*, along the southeastern United States coast. *Bulletin of Marine Science*, **46**(3), 677–687.

Hoshino, E., Pasco, S., Hutton, T., Kompas, T., & Yamazaki, S. 2018. Estimating maximum economic yield in multispecies fisheries: A review. *Rev Fish Biol Fisheries*, **28**, 261–276.

Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourgue, B.J., Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J.W., Hughes, T.P., Kidwell, S., Lange, C.B., Lenihan, H.S., Pandolfi, J.M., Peterson, C.H., Steneck, R.S., Tegner, M.J., & Warner, R.R. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science*, **293**(5530).

Johns, G.M., Leeworthy, V.R., Bell, F.W., & Bonn, M.A. 2001. *Socioeconomic Study of Reefs in Southeast Florida: Final Report*. Tech. rept. Hazen and Sawyer Environmental Engineers and Scientists.

Johns, G.M., Lee, D.J., Leeworthy, V.B., Boyer, J., & Nuttle, W. 2014. Developing economic indices to assess the human dimensions of the South Florida coastal marine ecosystem services. *Ecological Indicators*, **44**, 69–80.

Kadison, E., D'Alessandro, E.K., Davis, G.O., & Hood, P.B. 2010. Age, growth, and reproductive patterns of the great barracuda, *Sphyraena barracuda*, from the Florida Keys. *Bulletin of Marine Science*, **86**(4), 773–784.

Kellner, J.B., Sanchirico, J.N., Hastings, A., & Mumby, P.J. 2011. Optimizing for multiple species and multiple values: tradeoffs inherent in ecosystem-based fisheries management. *Conservation Letters*, **4**(1), 21–30.

Kutner, M.H., Nachtsheim, C.J., Neter, J., & Li, W. 2004. *Applied Linear Statistical Models*. McGraw-Hill/Irwin.

Leopold, A. 1949. *A Sand County Almanac*. Oxford University Press.

Liubimova, T.G., & Capote, A.M. 1971. Biological characteristics of some commercial fish species of the Campeche Bank [In Russian]. vol. 3.

Lombardi, L.A., Cook, M., Lyon, H., Barnett, B., & Bullock, L. 2012. A Description of Age, Growth, and Reproductive Life History Traits of Scamps from the Northern Gulf of Mexico. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science*, **4**(1), 129–144.

Luckhurst, B.E., & Dean, J.M. 2009. Age estimates of two large misty grouper, *Epinephelus mystacinus* (Serranidae) from Bermuda with a comparison of the age of tropical groupers in the Western Atlantic. *Gulf and Caribbean Research*, **21**(1), 73–77.

Manooch, C.S. 1984. *Fisherman's guide, fishes of the southeastern United States*. Raleigh, NC: Museum of Natural History.

Manooch, C.S., & Barans, C.A. 1982. Distribution, abundance, and age and growth of the tomtate, *Haemulon aurolineatum*, along the southeastern United States coast. *Fishery Bulletin*, **80**(1), 1–19.

Manooch, C.S., & Mason, D.L. 1987. Age and growth of the Warsaw grouper and black grouper from the Southeast region of the United States. *Northeast Gulf Science*, **9**(2), 65–75.

Marques, S., & Ferreira, B.P. 2011. Sexual development and reproductive pattern of the Mutton hamlet, *Alphestes afer* (Teleostei: Epinephelidae): a dyandric, hermaphroditic reef fish. *Neotropical Ichtyology*, **9**(3), 547–558.

Mason, D.L., & Manooch, C.S. 1985. Age and growth of mutton snapper along the east coast of Florida. *Fisheries Research*, **3**, 93–104.

Matheson, R.H., Huntsman, G.R., & Manooch, C.S. 1986. Age, growth, mortality, food, and reproduction of the scamp, *Mycteroperca phenax*, collected off North Carolina and South Carolina. *Bulletin of Marine Science*, **38**(2).

Maunder, M.N., & Piner, K.R. 2015. Contemporary fisheries stock assessment: many issues still remain. *ICES Journal of Marine Science*, **71**(1), 7–18.

McBride, R. 2001. *Age, growth, and reproduction of hogfish, Lachnolaimus maximus.* Tech. rept. Florida Fish and Wildlife Conseration Commission, St. Petersburg, FL. NOAA Award Number: NA87FF0422.

McBride, R.S., & Richardson, A.K. 2007. Evidence of size-selective fishing mortality from an age and growth study of hogfish (Labridae: *Lachnolaimus maximus*), a hermaphroditic reef fish. *Bulletin of Marine Science*, **80**(2), 401–417.

McCauley, D.J., Pinsky, M.L., Palumbi, S.R., Estes, J.A., Joyce, F.H., & Warner, R.R. 2015. Marine defaunation: Animal loss in the global ocean. *Science*, **347**(6219), 247–254.

McClenacan, L. 2009. Documenting loss of large trophy fish from the Florida Keys with historical photographs. *Conservation Biology*, **23**(3), 636–643.

McRae, G. 2010. A history of Florida's Fish and Wildlife Research Institute. *Gulf of Mexico Science*, **28**(1–2), 173–179.

Mendes, R.A., Hazin, F.H.V., de Oliveira, P.G., & de Carvalho, F.C. 2007. Reproductive biology of the squirrelfish, *Holocentrus adscensionis* (Osbeck, 1975), caught off the coast of Pernambuco, Brazil. *Scientia Marina*, **71**(4), 715–722.

Methot, R. D., & Wetzel, C. R. 2013. Stock synthesis: a biological and statistical framework for fish stock assessment and fishery management. *Fisheries Research*, **142**, 86–99.

Methot, R.D. 1986. *Synthetic estimates of historical abundance and mortality for northern anchovy, Engraulis mordax.* Tech. rept. NMFS Southwest Fisheries Science Center Admin.

Molina-Urena, H., & Ault, J.S. 2007. Towards an ecosystem-based approach to assess non-target tropical reef fishes. *Gulf and Caribbean Fisheries Institute*, **59**, 185–190.

Montes, M. 1975. *Personal communication in Mason & Manooch (1985).*

MSA. 1976. *Magnuson-Stevens Fishery Conservation and Management Act.* Tech. rept. 94th Congress.

Munro, J.L. 1976. Aspects of the biology and ecology of Caribbean reef fishes: Mullidae (goat-fishes). *Journal of Fish Biology*, **9**, 79–97.

Munro, J.L. 1999. *Marine protected areas and the management of coral reef fisheries*. Tech. rept. International Center for Living Aquatic Resources Management (ICLARM), Caribbean and Eastern Pacific Office.

Murie, D.J., & Parkyn, D.C. 2005. Age and growth of white grunt (*Haemulon plumieri*): A comparison of two populations along the west coast of Florida. *Bulletin of Marine Science*, **76**(1), 73–93.

Murphy, M.D., Murie, D.J., & Muller, R.G. 1999. *Stock assessment of white grunt from the west coast of Florida*. Tech. rept. Florida Fish and Wildlife Conservation commission.

Murray, P.A., & Moore, E.A. 1993. Recruitment and exploitation rate of *Etelis oculatus* Val. in the St. Lucian fishery. *Proc. Gulf. Carib. Fish. Inst.*, **42**, 262.

Mutz, S.J. 2006. *Comparative growth dynamics of Acanthurid fishes*. Ph.D. thesis, Marine Biology and Aquaculture, James Cook University, Townsville, QLD, AUS.

Nadon, M.O., & Ault, J.S. 2016. A stepwise stochastic approach to estimate unobserved life history parameters for data-poor fisheries. *Canadian Journal of Fisheries and Aquatic Sciences*, **73**, 1874–1884.

Nadon, M.O., Ault, J.S., Williams, I.D., Smith, S.G., & DiNardo, G.T. 2015. Length-based assessment of coral reef fish populations in the main and northwestern Hawaiian Islands. *PLoS ONE*, **10**(8), e0133960.

Nagelkerken, W.P. 1979. Biology of the graysby, *Epinephelus cruentatus*, of the coral reef of Curacao. *Studies on the Fauna of Curacao and Other Caribbean Islands*, **60**, 1–118.

Nelson, R.S., Manooch, C.S., & Mason, D.L. 1985. *Ecological effects of energy development of reef fish of the Flower Garden Banks: Reef fish bioprofiles*. Tech. rept. Southeast Fisheries Science Center, National Marine Fisheries Service, NOAA, Beaufort, NC.

Newman, D., Berkson, J., & Suatoni, L. 2015. Current methods for setting catch limits for data-limited fish stocks in the United States. *Fisheries Research*, **164**, 86–93.

O'Hop, J., Murphy, M., & Chagaris, D. 2012. *The 2012 Stock Assessment Report for Yellowtail Snapper in the South Atlantic and Gulf of Mexico, SouthEast Data, Assessment, and Review (SEDAR) 27A*. Tech. rept. Florida Fish and Wildlife Conservation Commission, St. Petersburg, FL. 341 pp.

O'Hop, J., Muller, R.G., & Addis, D.T. 2015. *Stock Assessment of Mutton Snapper (Lutjanus analis) of the U.S. South Atlantic and Gulf of Mexico through 2013, SEDAR Update Assessment*. Tech. rept. 15. Florida Fish and Wildlife Conservation Commission, St. Petersburg, FL.

Olaechea, A., Hernandez, C., & de Leon, M. 1975. *Evaluacion de peces dermsales en el Banco de Campeche*. Tech. rept. Cent. Invest. Pesq., La Habana, Cuba.

Olsen, S. 2008. *Norwegian Spring-spawning Herring and Northeast Arctic Cod: 100 Years of Research and Management*. Chap. Technological development in Norwegian fisheries, pages 33–40.

Ostrom, E., Burger, J., Field, C.B., Norgaard, R.B., & Policansky, D. 1999. Revisiting the commons: Local lessons, global challenges. *Science*, **284**, 278–282.

Palazon, J.L., & Gonzalez, L.W. 1986. Edad y crecimiento del pargo cebal, *Lutjanus analis* (Cuvier, 1828), (Teleostei: Lutjanidae) en la Isla Margarita y alrededores de Venezuela. *Invest. Pesq. (Barc.)*, **50**(2), 151–165.

Pitt, J.A., Trott, T.M., & Luckhurst, B.E. 2010. Bluestriped grunt (*Haemulon sciurus*) in Bermuda: Age, growth, and reproduction studies. *Pages 241–245 of: Proceedings of the 62nd Gulf and Caribbean Fisheries Institute*.

Pope, J.G. 1972. An investigation of the accuracy of virtual population analysis using cohort analysis. *Int. Comm. Northwest Atl. Fish. Res. Bull.*, **9**, 65–74.

Posada, J.M., & Appeldoorn, R.S. 1996. The validity of length-based methods for estimating growth and mortality of groupers, as illustrated by comparative assessment of the creole fish (*Paranthias furcifer*)(Pisces: Serranidae). *Pages 163–173 of: Biology, fisheries, and culture of tropical groupers and snappers*. ICLARM.

Poso, E., & Espinosa, L. 1982. Estudio de la edad y crecimiento del pargo del alto (*Lutjanus vivanus*) en la plataforma suroriental de Cuba. *Rev. Cub. Invest. Pesq.*, **7**(2), 1–23.

Poso, E., & Espinosa, L. 1983. *Resultados de las investigaciones biologicopesqueras del pargo del alto en la plataforma cubana*. Tech. rept. Cent. Invest. Pesq., La Habana.

Potts, J.C., & Manooch, C.S. 1995. Age and growth of red hind and rock hind collected from North Carolina through the Dry Tortugas, Florida. *Bulletin of Marine Science*, **56**(3), 784–794.

Potts, J.C., & Manooch, C.S. 1999. Observations on the age and growth of graysby and coney from the southeastern United States. *Transactions of the American Fisheries Society*, **128**, 751–757.

Potts, J.C., Burton, M.L., & Myers, A.R. 2016. Age, growth, and natural mortality of schoolmaster (*Lutjanus apodus*) from the southeastern United States. *PeerJ*, **4:e2543**. DOI 10.7717/peerj.2543.

Pozo, E. 1979. Edad y crecimiento del pargo criollo (*Lutjanus analis* [Cuvier, 1828]) en la plataforma noroccidental de Cuba. *Rev. Cub. Invest. Pesq.*, **4**(2), 1–24.

Previero, M., Minte-Vera, C.V., Freitas, M.O., de Moura, R. Leao, & Tos, C.D. 2011. Age and growth of th edog snapper *Lutjanus jocu* (Bloch & Schneider, 1801) in Abrolhos Bank, Northeastern Brazil. *Neotropical Ichthyology*, **9**(2), 303–401.

Quinn, T.J., & Deriso, R.B. 1999. *Quantitative Fish Dynamics*. Oxford University Press.

Radcliffe, L. 1919. *Fishery Industries of the United States*. Tech. rept. 892. Department of Commerce, Bureau of Fisheries.

Randall, J.E. 1962. Tagging reef fishes in the Virgin Islands. *Proceedings of the Gulf and Caribbean Fisheries Institute*, **14**, 201–241.

Reeson, P.H. 1975a. Chapter 13: The biology, ecology and bionomics of the parrotfishes, Scaridae. *In:* Munro, J.L. (ed), *Caribbean Coral Reef Fishery Resources*. Manila, Philippines: International Center for Living Aquatic Resources Management.

Reeson, P.H. 1975b. Chapter 14: The biology, ecology and bionomics of the surgeonfishes, Acanthuridae. *In:* Munro, J.L. (ed), *Caribbean Coral Reef Fishery Resources*. Manila, Philippines: International Center for Living Aquatic Resources Management.

Render, J.H., & Wilson, C.A. 1992. Reproductive biology of sheepshead in the northern Gulf of Mexico. *Transactions of the American Fisheries Society*, **121**, 757–764.

Richardson, A.K., Thurman, P.E., & Bullock, L.H. 2008. Regional variations of hogfish (*Lachnolaimus maximus*) life history: consequences for spawning biomass and egg production models. *Journal of Northwest Atlantic Fishery Science*, **41**, 1–12.

Roa, R., Ernst, B., & Tapia, F. 1999. Estimation of size at sexual maturity: an evaluation of analytical and resampling procedures. *Fishery Bulletin*, **97**, 570–580.

Roberts, C. 2007. Plunder of the Caribbean. *Chap. 5, pages 57–69 of: The Unnatural History of the Sea*. Island Press.

Robertson, D.R., & Warner, R.R. 1978. Sexual patterns in the labroid fishes of the western Caribbean II: The parrot fishes (Scaridae). *Smithson. Contrib. Zool.*, **255**, 1–26.

Ruiz, L.J, Figueroa, R.M., & Prieto, A.A. 1999. Ciclo reproductivo de *Lactophrys quadricornis* (Pisces: Osctaciidae) de la costa nororiental de Venezuela. *Rev. Biol. Trop.*, **47**(3), 561–570.

Sadovy, Y., & Eklund, A.M. 1999. *Synopsis of biological data on the Nassau grouper, Ephinephelus striatus (Bloch 1792), and the Jewfish, E. itajara (Lichtenstein, 1822).* Tech. rept. NOAA Technical Report NMFS 146, Seattle, Washington, USA.

Sadovy, Y., Figuerola, M., & Román, A. 1992. Age and growth of red hind *Epinephelus guttatus* in Puerto Rico and St. Thomas. *Fishery Bulletin*, **90**, 516–528.

Sadovy, Y., Rosario, A., & Román, A. 1994. Reproduction in an aggregating grouper, the red hind, *Epinephelus guttatus*. *Environmental Biology of Fishes*, **41**, 269–286.

SAFMC. 1983. *Source Document for the Snapper-Grouper Fishery of the South Atlantic Region*. Tech. rept. South Atlantic Fishery Management Council, Charleston, SC.

Sagarese, S.R., Rios, A.B., Cass-Calay, S.L., Cummings, N.J., Bryan, M.D., Stevens, M.H., Harford, W.J., McCarthy, K.J., & Matter, V.M. 2018. Working towards a framework for stock evaluations in data-limited fisheries. *North American Journal of Fisheries Management*, **38**, 507–537.

Sanchirico, J.N., & Springborn, M. 2011. How to Get There From Here: Ecological and Economic Dynamics of Ecosystem Service Provision. *Environmental Resource Economics*, **48**, 243–267.

Santana, F.M., Morize, E., & Lessa, R. 2006. Age and growth of the spotted goatfish, *Pseudupeneus maculatus* (Bloch, 1793) in Brazil, validated through marginal increment and oxytetracycline dyes in the sagittae. *Journal of Applied Ichthyology*, **22**, 132–137.

SEDAR. 2002. *Stock assessment of South Atlantic red porgy*. Tech. rept. Charleston, SC.

SEDAR. 2004. *Complete Stock Assessment Report of SEDAR 6, Hogfish Snapper*. Tech. rept. 6. Southeast Data, Assesment, and Review, Charleston, SC. 127pp.

SEDAR. 2006. *Stock assessment of red porgy off the southeastern United States, SEDAR update assessment*. Tech. rept. Beaufort, NC.

SEDAR. 2008 (February). *SEDAR15A, Stock Assessment Report 3 (SAR3), South Atlantic and Gulf of Mexico Mutton Snapper*. Tech. rept. 15. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2010. *Gulf of Mexico and South Atlantic Black Grouper*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2011. *Gulf of Mexico Yellowedge Grouper*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2013a. *SEDAR 31: Gulf of Mexico Red Snapper Stock Assessment Report*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2013b. *SEDAR 36: South Atlantic Snowy Grouper Stock Assessment Report*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2014. *SEDAR 33: Gulf of Mexico Gag Stock Assessment Report*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2015a (August). *SouthEast Data, Assessment, and Review (SEDAR) 42, Gulf of Mexico Gray Triggerfish*. Tech. rept. North Charleston, SC. 286 p.

SEDAR. 2015b. *SouthEast Data, Assessment, and Review (SEDAR) 42, Gulf of Mexico Red Grouper. Section II: Data Workshop Report*. Tech. rept. North Charleston, SC. 286 p.

SEDAR. 2016a. *Gulf of Mexico data-limited species: Red drum, lane snapper, wenchman, yellowmouth grouper, speckled hind, snowy grouper, almaco jack, lesser amberjack*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2016b. *Gulf of Mexico Vermilion Snapper*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

SEDAR. 2016c. *Southeastern U.S. Goliath Grouper*. Tech. rept. SouthEast Data, Assessment, and Review, North Charleston, SC.

Seitz, R.D., Wennhage, H., Bergstrom, U., Lipcius, R.N., & Ysebaert, T. 2014. Ecological vlaue of coastal habitats for commercially and ecologically important species. *ICES Journal of Marine Science*, **71**(3), 648–665.

Shertzer, K.W., Fieberg, J., Potts, J.C., & Burton, M.L. 2017. *Identifying growth morphs from mixtures of size-at-age data.*

Solis, D., del Corral, J., Perruso, L., & Agar, J.J. 2014. Evaluating the impact of individual fishing quotas (IFQs) on the technical efficiency and composition of the U.S. Gulf of Mexico red snapper commercial fishing fleet. *Food Policy*, **46**, 74–83.

Starck, W.A. 1970. Investigation on the gray snapper, *Lutjanus griseus*. *Stud. Trap. Oceanogr.*, **10**, 224.

Tapia, M., Yanez-Arancibia, A., Sanchez-Gil, P., & Garcia-Abad, M.C. 1995. Distribution, abundance, and reproduction of *Priacanthus arenatus* Cuvier (Pisces: Priacanthidae) in the continental shelf of the Gulf of Mexico. *Biotropica*, **27**(2), 232–237.

Taylor, H.F. 1916. The structure and growth of the scales of the squeteague and the pigfish as indicative of life history. *Bull. U.S. Bur. Fish*, **34**, 285–330.

Thompson, R., & Munro, J.L. 1978. Aspects of the biology and ecology of Caribbean reef fishes: Serranidae (hinds and groupers). *Journal of Fish Biology*, **12**, 115–146.

Thompson, R., & Munro, J.L. 1983. The biology, ecology, and bionomics of the snappers, Lutjanidae. Caribbean coral reef fishery resources. *ICLARM Studies and Reviews*, **7**, 94–109.

Thorson, J.T., Munch, S.B., Cope, J.M., & Gao, J. 2017. Predicting life history parameters for all fishes worldwide. *Ecological Applications*, **27**(8), 2262–2276.

Trott, T.M. 2006. An evaluation of the status of the coney, *Cephalopholis fulva*, population in Bermuda. *Proceedings of the Gulf and Caribbean Fisheries Institute*, **60**, 257–262.

Turner, S.C., Porch, C.E., Heinemann, D., Scott, G.P., & Oritz, M. 2001 (August). *Status of Gag in the Gulf of Mexico, Assessment 3.0.* Tech. rept. Southeast Fisheries Science Center, Miami, FL.

Tyler-Jedlund, A.J., & Torres, J.J. 2015. Age, growth, and reproduction of the littlehead porgy, *Calamus proridens*, from the eastern Gulf of Mexico. *Bulletin of Marine Science*, **91**(2), 101–123.

USDOC, United States Department of Commerce. 2007. Magnuson-Stevens Fishery Conservation and Management Act. *USA Public Law*, 94–265.

Walker, D.J.R. 1992. *Columbus and the Golden World of the Arawaks.* The Book Guild Ltd.

Waltz, C.W., Roumillat, W.A., & Wenner, C.A. 1982. Biology of the whitebone porgy, *Calamus leucosteus*, in the South Atlantic Bight. *Fishery Bulletin*, **80**(4), 863–874.

Wessels, C.R., & Anderson, J.L. 1992. Innovations and progress in seafood demand and market analysis. *Marine Resource Economics*, **7**, 209–228.

Wyatt, J.R. 1983. The biology, ecology, and bionomics of the squirrelfishes, Holocentridae. *Chap. 125, pages 50–58 of:* Munro, J.L. (ed), *Caribbean Coral Reef Fishery Resources ICLARM Stud. Rev.*

Ximenes, M.O., Fonteles, A.A., & Paiva, M.P. 2009. Parametros de crescimento e mortalidade do olho-de-cao, *Priacanthus arenatus* (Teleostei: Priacanthidae), no sedeste do Brasil. *Arq. Cien. Mar, Fortaleza*, **42**(1), 5–11.

Ziskin, G.L. 2008. *Age, growth, and reproduction of speckled hind, Epinephelus drummondhayi, off the Atlantic coast of the southeast United States.* M.Phil. thesis, College of Charleston, Charleston, SC.

Ziskin, G.L., Harris, P.J., Wyanski, D.M, & Reichert, M.J.M. 2011. Indications of continued overexploitation of speckled hind along the Atlantic coast of the southeastern United States. *Transactions of the American Fisheries Society*, **140**, 384–398.