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ESTIMATION OF MORTALITY RATES FOR THE GULF MENHADEN STOCK AND REFERENCE POINT IMPLEMENTATION FOR THE FISHERY

Catherine Wilhelm

ESTIMATION OF MORTALITY RATES FOR THE GULF MENHADEN STOCK AND REFERENCE POINT IMPLEMENTATION FOR THE FISHERY

by

Catherine Wilhelm

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Approved by:

Dr. Robert Leaf, Committee Chair
Dr. Amy M. Schueller
Dr. Kim De Mutsert

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ABSTRACT

The Gulf menhaden stock is the target of a large commercial fishery in the Gulf of Mexico. To address the needs of both the stock and fishery, I performed two studies. I first addressed the natural mortality rates currently used in the Gulf menhaden stock assessment. To update these rates, I used data from a tagging study conducted from 1970 to 1988. Adult and juvenile menhaden were tagged, released, and recovered in fish processing plants. To evaluate the data, I built a Bayesian model using the negative binomial distribution to estimate natural mortality, catchability, and the overdispersion factor parameters. I established a Base model and n = 17 sensitivity models for robustness. I estimated a constant instantaneous natural mortality of 1.08 y^{-1} with 95% confidence intervals of 1.04 y^{-1} to 1.13 y^{-1}. This estimate falls within the stock assessment confidence intervals, and validates the estimation of Ahrenholz (1981). These updated natural mortality rates can be directly used in the stock assessment. I then addressed the need for an implemented fishery reference point. For the fishery to provide verification of sustainability on an annual basis, the Gulf State Marine Fisheries Commission developed an index-based reference point through an algorithm adopted by managers and stakeholders. I developed a web-based application and 3 time series for the use and implementation of the fishery reference point. This dashboard uses processed fishery independent survey data and creates interactive elements for stakeholders to identify the risks of different threshold index values to be used for management.

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LIST OF ABBREVIATIONS

AL	Alabama
BAM	Beaufort Assessment Model
CI	Confidence Interval
CJS	Cormack-Jolly-Seber
FEC	Fecundity
FL	Fork Length
GSMFC	Gulf State Marine Fisheries Commission
LA	Louisiana
MLE	Maximum Likelihood Estimate
MS	Mississippi
MSC	Marine Stewardship Council
NOAA	National Oceanic and Atmospheric Administration
PBIAS	Percent Bias
RMSE	Root Mean Square Error
SD	Standard Deviation
SEDAR	Southeast Data, Assessment, and Review
SSB	Spawning Stock Biomass
TAC	Total Allowable Catch
TL	Total Length
USM	The University of Southern Mississippi
WWII	World War II

INTRODUCTION

Gulf menhaden *Brevoortia patronus*, is a member of the family Alosidae in the order Clupeiformes (Q. Wang et al., 2022). The species is the target for an economically valuable commercial purse seine reduction fishery (Ahrenholz, 1991; Smith, 1991) in the Gulf of Mexico. Some of the Clupeiformes (e.g. sardines, herring, shad, menhaden) are ‘forage fishes’, which are a multispecies group of small, schooling, and generally pelagic species preyed upon by a wide variety of marine predators (Cury et al., 2000; Olsen et al., 2014; Pikitch et al., 2012). Forage fishes are primarily planktivorous and provide a trophic link between larger consumers and primary producers (Cury et al., 2000; Geers et al., 2014; Pikitch et al., 2012). Their abundance can have a profound impact on the dynamics of higher trophic levels and the ecosystem (Geers et al., 2014; Pikitch et al., 2012). Gulf menhaden abundance directly effects the mortality and biomass of commercially valuable predators such as sharks, mackerel, tarpon (*Megalops atlanticus*), bonito (*Sarda sarda*), red drum (*Sciaenops ocellatus*), and other species (Berenshtein et al., 2023; Geers et al., 2014).

1.1 Movement and Distribution

Gulf menhaden are found throughout the Gulf of Mexico from Florida to the Yucatan Peninsula. They aggregate in the coastal zone in the summer and move offshore during the fall and winter to spawn (Ahrenholz, 1981; Govoni, 1997; Nicholson & Schaaf, 1978). Consistent east-to-west movement of Gulf menhaden in the Gulf of Mexico has not been detected (Ahrenholz, 1981, 1991), although older fish have been documented to migrate towards the Mississippi River Delta (Ahrenholz, 1981; Govoni, 1997). Larvae are transported to estuarine systems during late winter and early spring (Brown-Peterson et al., 2017; Deegan, 1990; Govoni, 1997). Juvenile young-of-the-year (age-0) live in the estuarine systems until the following fall and winter when they begin to move offshore and into the adult populations (Brown-Peterson et al., 2017; Deegan, 1990; Govoni, 1997). At age-1, they begin to reach sexual maturity (Ahrenholz, 1991; Brown-Peterson et al., 2017). The length of sexually mature fish ranges from approximately 140 mm fork length (FL) to a little over 300 mm FL (Brown-Peterson et al., 2017; Nicholson & Schaaf, 1978). The spawning season is protracted and lasts from October to late March (Brown-Peterson et al., 2017; Nicholson & Schaaf, 1978). Females have high fecundity, and are indeterminate batch spawners, capable of producing over 1 million eggs each season per individual (Ahrenholz, 1991; Brown-Peterson et al., 2017; Nelson & Ahrenholz, 1986). *B. patronus* can live 5 to 7 years (Schaaf et al., 1975).

The Gulf menhaden fishery has been active since the 1800s, although very little documentation exists of fishing effort or catch (Nicholson, 1978) in the early years of the fishery (Figure 1.1). After WWII, the fishery grew in size into what is considered the modern Gulf menhaden fishery, accompanied by greater monitoring and reporting efforts (Chapoton, 1972; Nicholson, 1978; SEDAR63, 2018; Smith, 1991; Vaughan et al., 2006). The history of recorded landings indicates that harvests of less than 30,000 metric tons (mt) before 1948 increased to 480,700 mt in 1962 (Nicholson, 1978; SEDAR63, 2018) . The catch then declined until 1969, where it increased to 523,700 mt and continued to increase overall until 1984, where the catch peaked at 985,120 mt (Chapoton, 1972; Nicholson, 1978; SEDAR63, 2018; Smith, 1991). The catch remained above 800,000 mt until 1988 then generally decreased (SEDAR63, 2018; Vaughan et al., 2006). From 1988 to 2020, catch ranged from 771,770 mt in 1994 to 400,720 mt in 2014 (SEDAR63, 2018). Catch in 2020 was approximately 413,855 mt (Mroch, 2021). The fishery primarily targets age-1 and age-2 fish (Ahrenholz, 1981; Chapoton, 1972; GDAR03, 2021; Nicholson & Schaaf, 1978; SEDAR63, 2018; Smith, 1991).

1.2 Fishing Operations

In 1949 spotter planes were introduced to guide the larger carrier vessels (120-200 feet in length) to schools (Nicholson, 1978; Smith, 1991). When carrier vessels observe a school of fish, purse boats (up to 40 feet in length) are unloaded to encircle schools or parts of schools. Each purse boat carries half the purse seine net each (SEDAR63, 2018; Smith, 1991). A purse line running at the bottom of the seine net is tightened to effectively trap the school once encircled, forming one ‘set’ (Smith, 1991). Fishing trips are generally several days, and average about 4 to 5 sets per day with a median of 17 to 22 mt per set (SEDAR63, 2018; Smith et al., 2002). The fish in the purse seine are then pumped into the larger carrier vessel’s hold. Refrigerated holds were first used in vessels in 1957, keeping the fish in good condition (Smith, 1991) and increasing the time and distance of fishing trips (Nicholson, 1978).

The fishery’s main product consists of fish oil and fish meal, which are used for animal feed, fish oil supplements, and fertilizer (Nicholson, 1978; Nicholson & Schaaf, 1978; SEDAR63, 2018; Smith, 1991). The commercial fishery is a reduction fishery because the harvest is ‘reduced’ into fish meal, oil, and solubles through a manufacturing process. Fish are cooked and pressed to separate oil and liquid, each of which are refined for use in specific products, and the leftover fish scrap is also refined into fish meal (Smith, 1991). This process takes place in a reduction plant facility located on shore. Historically, reduction plants were located from Florida to Texas. After the first reduction plant opened in 1946, the number of plants increased to 10 by 1950 and ranged from 10 to 14 from the 1950s to 1975, after which the number stabilized to 11 functioning plants until 1984 (Nicholson, 1978; SEDAR63, 2018; Smith, 1991; Smith et al., 2002; Vaughan et al., 2006). After 1984, the number of plants steadily decreased until only 3 remained open (SEDAR63, 2018; Vaughan et al., 2006). In 1995 and 2003 respectively, Florida and Alabama banned the use of purse-seining for menhaden in state waters (Vaughan et al., 2006). Today, the only reduction plants currently operating are in Louisiana (n = 2) and Mississippi (n = 1).

The fishery operates from mid-April through the end of October, with the greatest landings occurring during June to August (Brown-Peterson et al., 2017; Nicholson & Schaaf, 1978; Schaaf et al., 1975; Smith, 1991). The season was officially limited to the current extent in 1977 from a previously undefined temporal season (Smith, 1991). The purse-seine fishery operates nearshore (under 10 miles) during daytime hours (Kemmerer, 1980; Smith, 1991; Smith et al., 2002).

The assessment of the Gulf menhaden stock is coordinated by the Gulf States Marine Fisheries Commission and conducted by scientists from NOAA Fisheries. A statistical catch-at-age model, the Beaufort Assessment Model (BAM), is used to estimate abundance and develop metrics for stock and fishery status determinations (SEDAR63, 2018; Williams & Shertzer, 2015). The model incorporates Lorenzen age-specific natural mortality scaled to the instantaneous, age-specific natural mortality rates

(M y^{-1}) estimated from a tagging study done by Ahrenholz in the early 1970s (Ahrenholz, 1981; Nelson & Ahrenholz, 1986; SEDAR63, 2018). Data included in the BAM are a time series of landings, age compositions of the landings, two fishery independent indices (an adult index and a recruitment index), and length compositions for the adult abundance index. Recruitment is estimated using a Beverton-Holt stock recruitment curve. Selectivity in the model is specified as dome-shaped for the fishery and flat-topped for the adult abundance index (SEDAR63, 2018). Dome-shaped selectivity occurs in the fishery: age-0 fish are not generally caught and older fish have a lower probability of being caught. The adult abundance index has flat-topped selectivity because older fish are fully selected by the gear. The Menhaden Advisory Council uses the output of this model to determine stock and fishery status. Currently, the Gulf menhaden stock is considered not overfished nor is it experiencing overfishing (SEDAR63, 2018).

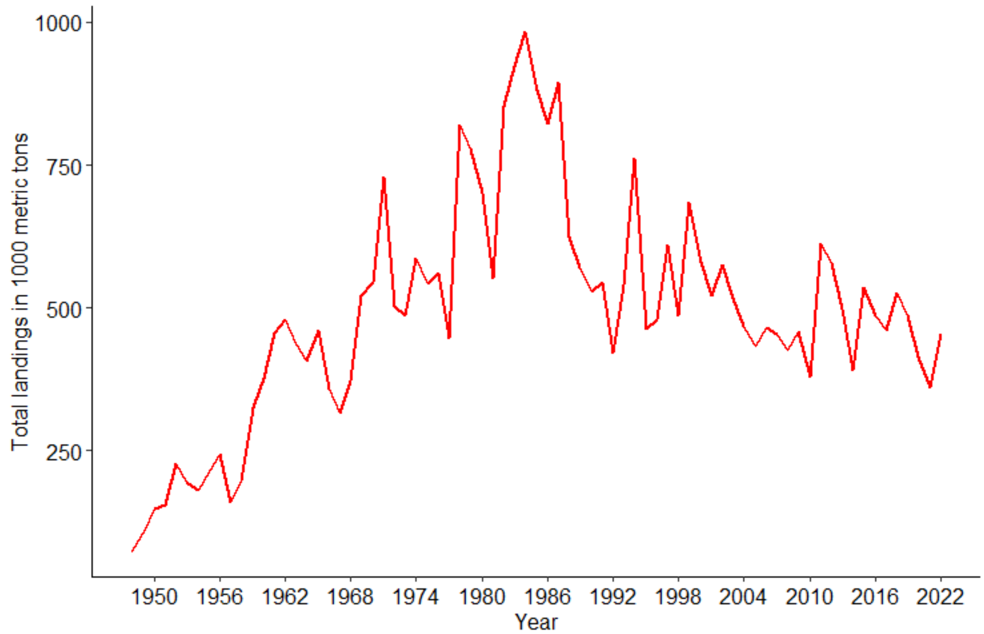
1.3 Goals and Objectives

In the first chapter, I employed a mark-recovery model using ADMB to evaluate archived and recently digitized mark and recapture records. This model estimated mortality rates based on the mark recovery data while using contemporary statistical methods to provide validation of current parameters, indicate the annual variation in mortality, and confirm our understanding of the range of observed mortality. My model makes use of recovery data in which the fish were captured, tagged, and then recovered dead.

In the second chapter, I developed a web-based application for the use and implementation of the reference point developed by D. Butterworth. This reference point was investigated in a series of public meetings conducted by the Gulf States Marine Fisheries Commission's Menhaden Advisory Committee. However, it has yet to be implemented. There are incentives for the adoption of this reference point by the fishery, primarily to ensure that the Marine Stewardship Council sustainability criteria for the fishery are met. To move the algorithm developed and tested by D. Butterworth to implementation, I operationalized the reference point in an online application to create an interactive method for users to identify periods of change in stock status. The online application 1) processes sampling data from state agencies, 2) performs some limited quality assurance and control to fix data errors, and 3) creates three time series from the fishery-independent data, the Louisiana seine and gillnet indices and a combined standardized index. The combined index represents the relative abundance of the stock, and can be compared to the threshold index chosen as the reference point of abundance. This representation of relative abundance allows direct comparison between the chosen reference point and the current year's index to ascertain if any regulations need to be enacted.

Figure 1.1 *Total Fishery Landings*

Total landings of the gulf menhaden Fishery in 1,000 metric tons per year since the 1940s.



ESTIMATION OF MORTALITY RATES FOR THE GULF MENHADEN STOCK

2.1 Abstract

The Gulf menhaden stock assessment currently uses natural mortality rates estimated from the 3-year tagging study done by Ahrenholz in the 1970s combined with a Lorenzen age-specific mortality. I used data from a 19-year tagging study conducted from 1970 to 1988 on Gulf menhaden with assistance from the fishery. Adult and juvenile menhaden were tagged, released, and then recovered in the fish processing plants. I built a Bayesian model in ADMB using the negative binomial distribution to estimate a constant natural mortality, catchability, and an overdispersion factor. I ran a Base model and 17 sensitivity models to test the precision of the parameter estimates. The Base model estimated a constant instantaneous natural mortality of 1.082 y^{-1} with 95% confidence intervals 1.04 y^{-1} to 1.13 y^{-1}. This estimate more closely represents age-1 and age-2 fish due to the selectivity of the fishery towards those age classes. My mean estimate is 1% lower than the projected estimate by Ahrenholz of 1.0935 y^{-1}, which is also assumed by the assessment to represent age-2 fish (SEDAR63, 2018). These updated natural mortality rates will directly improve the assessment model used for the Gulf menhaden stock.

2.2 Mark-Recapture Models

Mark-recapture models are widely used to model population dynamics and estimate rates of immigration, death, and emigration (Lindberg & Rexstad, 2002). These models have grown in use and complexity, aided by computer processing speed and improved methodologies (Brownie et al., 1993; Liljestrand et al., 2019a, 2019b;

Lindberg, 2012). Mark-recapture (or capture-recapture) is a tagging method that is based on the deployment of tags on individuals and one or more sightings or recapture events (known as sampling occasions) after the marked individuals have been ‘at large’ for some period of time (an interval). Interaction with sampled organisms is called an ‘encounter’ (Lindberg & Rexstad, 2002). In some cases and depending on the design of the study, newly encountered organisms (those not previously marked) are tagged and previously tagged organisms that are encountered are recorded and released back into the population (Lebreton et al., 1992; Pollock et al., 1990). The record of re-sighting a tagged individual or tagging a new individual allows the building of an individual’s capture history and are the primary data used in statistical modeling. An individual’s capture history is the record of when or if it was encountered during different sampling occasions throughout the study.

The structure and design of mark-recapture studies are determined by the parameters of interest and if assumptions can be met (Lindberg, 2012; Lindberg & Rexstad, 2002). Mark-recapture studies generally need to meet at least some of the following assumptions (Kendall et al., 1995; Lindberg, 2012; Lindberg & Rexstad, 2002; Pollock et al., 1990):

1.) tagged organisms represent their population and the population parameters can be inferred from the estimation of sample parameters,

2.) every organism tagged and alive at an arbitrary time in the population has the same chance of capture,

3.) all tagged organisms’ behavior and survival is independent of each other,

4.) tags do not affect the organism’s survival or behavior, 5.) tags are recordable and unique,

6.) there is no movement in or out of the population both physically or by birth or death,

7.) sampling is instantaneous, and

8.) all tagged organisms have the same survival probability within sampling occasions.

There are two primary classes of mark-recapture studies. The first are those that are used to model the dynamics of closed populations and the second are for open populations. Mark-recapture studies of closed populations are generally used for abundance estimation (Kendall, 1999; Lindberg, 2012). These models require that assumption #6 has been met. However, this assumption has been relaxed in recent years to allow for movement to be described quantitatively, by describing movement dynamics with parameters that can be estimated (Brownie et al., 1993; Lebreton et al., 1992; Liljestrand et al., 2019b; Lindberg & Rexstad, 2002). Open mark-recapture studies are primarily used to understand the dynamics of open populations and generally violate assumption #6 and sometimes assumption #8. The Cormack-Jolly-Seber (CJS) family of models, or live recapture models, are commonly used to estimate survival and capture probability (commonly represented as φ and p, respectively) parameters during multiple encounter mark-recapture studies. In these studies, the tagged organism's fate is unknown after the study, and therefore it is not possible to conclude if the organism is dead or has emigrated from the study area. Thus, parameters to define these processes are usually inseparable, that is, you are not able to differentiate between the probability of survival and the probability of being encountered (Lindberg, 2012). Inseparable parameters have several separate parameter variables that cannot be solved for individually, and consequently have one combined value. Multistate models have been developed to address this confounding process. States may be geographic regions or physical stages like age or organism health (Barnett et al., 2018; Kemmerer, 1980; Lindberg & Rexstad, 2002).

Known fate studies are used when the fate of the tagged organism is known or monitored after the study or sampling has finished, and typically only estimate true survival probability (Lindberg, 2012; Lindberg & Rexstad, 2002). Mark-recovery models have a similar design to both mark-recapture and known fate studies (Lindberg, 2012). In mark-recovery models, organisms are captured and tagged during sampling occasions, but all encounters of previously tagged organisms are of dead recoveries, a known fate, and occur usually after a period of time with little to no observation of tagged organisms has passed (Brownie & Pollock, 1985; Lindberg, 2012; Lindberg & Rexstad, 2002). Survival, encounter, and specific types of mortality rates are estimable (Lindberg, 2012). This type of model is most applicable for scenarios where the tag is reported by fishers or hunters.

Parameters are estimated in mark-recapture models using maximum likelihood and Bayesian approaches. Maximum likelihood is an approach to estimate model parameters as random variables and finds those values that are most likely to have resulted in the observed data (Myung, 2003). MLE provides asymptotically unbiased and consistent estimators that are normally distributed with minimum variance (Lebreton et al., 1992; Myung, 2003). Model fitting applying MLE uses an approach where the probability of observing the data is determined given the candidate values of the parameters. By examination of the spread of the parameters, it is possible to understand which parameter values are most likely to have resulted in the observed data. An alternative, the Bayesian approach, estimates model parameters by deriving the posterior probability, the probability of a given parameter value, given the observed data. The posterior probability is determined by the product of the prior probability and the likelihood divided by combination of the probabilities of data given the possible parameter values (Gotelli & Ellison, 2004; Kass & Raftery, 1995).

Mortality is an important parameter in stock assessment models (Vetter, 1988) as a key part of understanding the life history and dynamics of a species. It is essential when estimating population productivity, and can depend on factors such as age, sex, time and location (Punt et al., 2021). Mortality is usually separated and quantified into three different types for population assessments. The first is fishing mortality, which is the rate of anthropogenic harvesting (Quinn & Deriso, 1999). Accurate fishing mortality is influenced by catchability, which will naturally vary by time and space for many fisheries. Fishing mortality can be calculated using direct fishery landings and effort. The second is mortality from natural causes, or natural mortality (Quinn & Deriso, 1999) and can be a lot harder to quantify, as even gut content analysis of predators is contingent on spatial and temporal match-up with prey. Together fishing and natural mortality make up the third type, total mortality which, as the inverse of the survival rate, is used in the estimation of expected recoveries in a tagging model. In stock assessments, mortality usually varies by age class or time (Punt et al., 2021).

In the first chapter, I employ a mark-recovery model written in ADMB to evaluate archived and recently digitized mark and recapture records. I use a Bayesian modeling approach to estimate mortality with contemporary methods to provide validation of current parameters, indicate the annual variation in mortality, and confirm our understanding of the range of observed mortality.

2.3 Methods

The goal of this work is to estimate annual and monthly mortality rates using data from a long-term mark-recovery study conducted by Dean Ahrenholz and others in the 1970s to 1980s. The available data consist of three different components. The first component is the tagging study. In the tagging study, Gulf menhaden from two different life stages were tagged. Around n = 90,210 adult fish (fish of age-1 or older) were tagged with stainless steel plate tags from 1976 to 1985 during October (Table A2.1) and recovered from 1977 to 1988 (Ahrenholz et al., 1991). Approximately n = 142,313 juvenile fish (fish younger than age-0) were similarly tagged from 1970 to 1985 from July to October and recovered from 1971 to 1988. There were n = 28 combined release events across life stages and time (Table A1.1). Tagging methods followed the procedure outlined by Pristas and Willis (1973): fish were caught and injected with uniquely numbered ferromagnetic tags. Adults were injected with a large tag (14.0 x 3.0 x 0.5mm) and juveniles with a small tag (7.0 x 2.5 x 0.4mm) and were then released (Ahrenholz et al., 1991). Tagging mortalities were estimated to be 30% for juvenile Gulf menhaden (Byars, 1981) and 17% for adults, the median of a tagging study with a 10 to 24% adult loss estimate (Ahrenholz et al., 1991; Kroger & Dryfoos, 1972). The Gulf menhaden were then harvested by the purse-seine fishery during the fishing season. Tags were recovered from fishery reduction plant processing machinery by magnets fitted to the equipment. During the time of the study, there were 8 to 13 reduction plants operating in the northern Gulf of Mexico (Ahrenholz, 1981; Vaughan et al., 2006), but only 11 plants had recoveries used in this study. Two plants were removed from the analysis due to their closing only a year after the study began. The second data component of the study was a plant magnet efficiency study. This study quantified the observation probability of the magnets recovering the tags given that the tags are in the fishery plant. In this approach a known number of tagged fish (41,756 fish from 1971 to 1988) were entered in batches of 100 fish as 'trials' into the operating reduction facilities and monitoring the proportion of those that were recovered. The third component of the data was a small study in a single year, 1969, in which n = 3,446 adults were recovered. In this study, fork length was recorded during tagging. This allowed me to determine what length fish the taggers categorized as 'juvenile' and 'adult' age classes in the 19-year tag study.

For all data sources, I performed extensive quality assurance and quality control. Data were scanned for noticeable errors. These errors were then fixed to the correct value or removed due to uncertainty. The n = 283 recoveries with months labeled as >12 were removed due to uncertainty. The n = 3 fish with time at large outliers of 7+ years were removed from the data set. The n = 64 tags that were recovered in the month of November were reassigned to October, on the assumption that these tags were from fish recovered in the last sets of the season and had not be cleaned and recorded yet until November. There were n = 45 fish recovered at plants that either didn't exist or were not in operation during the study, so these records were removed. In total, n = 232,223 fish were tagged, and the n = 231,891 fish that were tagged in the fishing season were used in the analysis. A total of n = 3,848 tagged adult fish were recovered and n = 4,589 fish tagged as juveniles were recovered, ranging from n = 32 to 1,063 fish caught per year

(Figure 2.1). Only n = 3,732 tagged adults and n = 4,471 tagged as juveniles were used due to missing recovery months in the data. The majority were recovered after two years from tagging with recoveries decreasing rapidly afterwards. I assumed the Gulf menhaden population was a closed population and did not experience movement.

2.3.1 Magnet Efficiency

I estimated the magnet efficiency from the second data set. The estimate of magnet efficiency quantifies how well the magnets installed in the plant recovered the ferromagnetic tags. Each trial, *i*, fed batches *n*, of 100 fish, into each plant, *j*, for each year, *y*, during the length the tagging study. I estimated the magnet efficiency, $\varepsilon_{j,y}$, using a binomial distribution of recoveries, *H*, per fishery plant and year by minimizing the sum of the negative log likelihood, NLL_{ε}, (Liljestrand et al., 2019a).

$$NLL_{\varepsilon} = \sum_{i} -\ln\left(\frac{n_{i}!}{H_{i,j,y}!\left(n_{i}! - H_{i,j,y}!\right)}\varepsilon_{j,y}^{H_{i,j,y}}\left[1 - \varepsilon_{j,y}\right]^{(n_{i} - H_{i,j,y})}\right) \quad (1)$$

I then calculated each annual magnet efficiency, ε_{y}, by taking the sum of all $\varepsilon_{j,y}$ for each year divided by total number of estimate magnet efficiency rates, $v_{j,y}$, to account for variation of efficiencies in each plant.

$$\varepsilon_{y} = \sum_{y} \frac{\varepsilon_{j,y}}{v_{j,y}} \quad (2)$$

For years without magnet testing, fishing months were assigned the average of all $\varepsilon_{j,y}$.

$$\varepsilon_{\bar{y}} = \sum_{j} \sum_{y} \frac{\varepsilon_{j,y}}{v_{j,y}} \quad (3)$$

The efficiencies were the same for all months except April because there was little monthly variation in magnet efficiencies when I performed a negative log likelihood estimate. April only had 4 trials run, 2 in 1987 and 2 in 1985, which were the only tests done for that year. I used April's estimated magnet efficiency for the months of April in all years. The year 1985 was given the average magnet efficiency, $\varepsilon_{\bar{y}}$, and 1987 was calculated without April tests. The year 1970 was given a ε_y of 0 due to no recoveries and no tests done on the magnets. Likewise, for months with no landings, ε_y was set to 0, which pertained to November through March of 1971 to 1988.

2.3.2 ADMB Model

AD Model Builder (ADMB) is a program that converts the user's source code to C++ with an AD library to minimize the function of the built model (Fournier et al., 2012). It is commonly used in fish stock assessments. Using ADMB, I built a Brownie dead-recovery model (Brownie et al., 1993) using Bayesian statistics to estimate natural mortality and fishing mortality. I constructed a base model with a constant natural mortality, M_c, a static catchability, q, and an overdispersion factor k to allow the estimated recoveries to be fit to the observed recoveries while accounting for zero inflated data. In the sensitivity runs, I allowed monthly and annual variations in natural mortality and catchability. The fitting of the model to the data and the parameter estimation can be separated by the user in a series of sequential phases. The constant catchability parameter q was estimated in phase 1, M_c and overdispersion k in phase 2, catchability variation in phase 3, and natural mortality deviations in phase 4. The constant catchability is a scalar of fishing mortality, so I estimated it first before natural mortality or k. The parameters M_c, k, q, and c were bounded and estimated in the natural log scale. Natural mortality deviations were bounded in the exponential scale based on the model developed by Liljestrand et al (2019). I assumed that tagged fish did not have higher rate of mortality and experienced the same conditions at each time step regardless of release cohort. I also assumed that all tagged fish were independent and were an accurate representation of the Gulf menhaden population.

2.3.2.1 Catchability

The model was used to estimate a constant q value for months with fishing effort, E, in the natural log scale. In months without fishing effort q was set to 0. Thus, catchability for a month and year, $Q_{m,y}$, was set as follows:

$$Q_{\mathrm{m,y}} = \left\{ \begin{array}{ll} e^{q} & when\ E > 0 \\ 0 & when\ E = 0 \end{array} \right\} \tag{4}$$

The estimated catchability parameter q is a scalar on fishing effort for fishing mortality. I assumed all tagged fish had an equal chance of being caught annually once they recruited to the adult population.

The total calculated catchability, $Q_{m,y}$, was evaluated with 4 different values for allowing deviations in catchability over time. All sensitivity runs including deviations in catchability included a constant q parameter and then varied based on a specified time step: 1) Constant deviation, or c, 2) annual varying deviation, c_y, 3) full time deviation of months and years, c_{my}, and 4) a monthly varying deviation, c_m.

$$Q_{\mathrm{m,y}} = e^{q} \ln(c) \tag{5}$$

$$Q_{\mathrm{m,y}} = e^{q} \ln(c_{\mathrm{y}}) \tag{6}$$

$$Q_{\mathrm{m,y}} = e^{q} \ln(c_{\mathrm{m,y}}) \tag{7}$$

$$Q_{\mathrm{m,y}} = e^{q} \ln(c_{\mathrm{m}}) \tag{8}$$

A constant deviation was estimated such that those months and years with recovery data were estimated to have a catchability different than the months and years with no recovery data. Months with magnet efficiencies and fishery landings but no recoveries were assigned the constant q estimate.

2.3.2.2 Mortality

Fishing effort, $E_{m,y}$, is defined as the landings in thousand metric tons per month and year from the fishery. I assumed a positive correlation between catch in landings and human effort made. Only 1971 to 1973 had landings in March and November, so the fishing effort for these months was fixed at 0 due to lack of recoveries and historically low fishing. Since these months were not commonly fished for all years, I removed them. Fishing morality, $F_{m,y}$, for each year and month was calculated by multiplying effort by year and month by the estimated catchability by year and month:

$$F_{\mathrm{m,y}} = E_{\mathrm{m,y}}\, Q_{\mathrm{m,y}} \tag{9}$$

Natural morality M_c y^{-1} was estimated as constant across all months and years for the base model. Total morality $Z_{m,y}$ was calculated from the natural morality and fishing mortality.

$$Z_{\mathrm{m,y}} = F_{\mathrm{m,y}} + M_c \tag{10}$$

2.3.2.3 Estimated Recoveries

The observed recovery data, $r_{T,A,t,a}$, were organized into a 4-d array following each release stage, *A*, and year's release cohort, *T*, to the time, *t*, that it was recovered in the fishery plant (Liljestrand et al., 2019a) and its recovery stage *a* (all recoveries were in adult stage). Because juveniles were tagged in the latter part of the year, they transitioned to the adult stage the year after they were tagged. Stage-based tagging morality, G_A, was applied to initial releases, $R_{T,A}$, to calculate the abundance of a release cohort $N_{T,A}$. Juveniles were assigned a tagging mortality of 30% (Byars, 1981) and adults were assigned the median tagging mortality of 17% from the study of tagging mortality on Atlantic menhaden (Kroger & Dryfoos, 1972).

$$N_{\mathrm{T,A}} = R_{\mathrm{T,A}}\,(1 - G_{\mathrm{A}}) \tag{11}$$

I estimated the expected recoveries, $\check{r}^{1}_{T,t+1,a}$, for the month of release as the product of the abundance after tagging mortality to the number that survived for that month,

$$\check{r}^{1}{}_{T,t+1,a} = N_{T,A}\, e^{-Z_{m,y}} \tag{12}$$

The following expected recoveries, $\check{r}^{*}{}_{T,A,t,a}$, were estimated from the abundance of fish and survival from the month prior.

$$\check{r}^{*}{}_{T,A,t,a} = \check{r}_{T,A,t-1,a}\, e^{-Z_{m-1,y}} \tag{13}$$

Months of expected recoveries that coincided with a 0% probability of observation were set to 0. The probability of observation is the probability of observing the tag from the fishery plant, given a certain fishing mortality rate, total mortality rate, and magnet efficiency. To determine the probability of observation I used the Baranov catch equation (Liljestrand et al., 2019a; Quinn & Deriso, 1999) to calculate the total expected recoveries, $\check{r}_{T,t,A,a}$,

$$\check{r}_{T,A,t,a} = \check{r}^{*}_{T,A,t,a}\, \frac{F_{m,y}}{Z_{m,y}} \left(1 - e^{-F_{m,y} - M}\right) \varepsilon_{y} \tag{14}$$

I performed a percent bias (PBIAS) and root mean square error (RMSE) calculation between the observed recoveries and the estimated expected recoveries for the Base model. These indices were used to determine how well the model fits the data.

2.3.2.4 Likelihood Function

The prior, M_p, for instantaneous M_c was 0.09 m^{-1}, the approximate monthly value estimated by Ahrenholz and currently used in the stock assessment (Ahrenholz, 1981; SEDAR63, 2018). The stock assessment uses the annual natural mortality value 1.1 y^{-1} (SEDAR63, 2018), which is a monthly rate of 0.09 m^{-1}. The prior was assumed to follow a normal distribution N (0.09, 1). Deviation of the estimated M_c from the prior was summed and minimized for all months to obtain the likelihood component P_{M}. For the sensitivity runs, equation 15 was calculated for all months without M_{c} deviations (the months with no tag recoveries). For the base model this included all months in the study.

$$P_{\mathrm{M}} = \sum_{m} \sum_{y} \frac{\left[M_{c_{\mathrm{m,y}}} - M_{\mathrm{p}}\right]^2}{2\,[s]^2} \tag{15}$$

The mark-recovery study had two observation events, that of tagging and recovery at death, so I used a negative binomial distribution when fitting the expected and observed recoveries. Fish tagging often has a large number of nonreturns, or zeros, in the final data and must be accounted for. To do this I used an overdispersion parameter. My estimated overdispersion parameter, *k*, is assumed constant for all years and months.

$$NLL_{\mathrm{r}} = \sum_{T}\sum_{A}\sum_{t>T}\sum_{a} - \ln\left\{\frac{\Gamma(k + r_{\mathrm{T,A,t,a}})}{\Gamma\left((k)(r_{\mathrm{T,A,t,a}})\right)}\left(\frac{k}{k + \check{\mathrm{r}}_{\mathrm{T,A,t,a}}}\right)^{k}\left(\frac{\check{\mathrm{r}}_{\mathrm{T,A,t,a}}}{k + \check{\mathrm{r}}_{\mathrm{T,A,t,a}}}\right)^{r_{\mathrm{T,A,t,a}}}\right\} \tag{16}$$

The total negative log likelihood *NLL* is the sum of the negative log likelihood of the observed recovery and expected recovery data, NLL_{r}, the natural mortality prior, P_{M}, the penalty on catchability, P_{q}, and time varying deviations, $D_{\mathrm{m,y}}$, from natural mortality M_{c}. The penalty for catchability and M_{c} deviations were 0 for the base model, as there was no variation from the estimated catchability *q* and M_{c}.

$$NLL = NLL_{\mathrm{r}} + P_{\mathrm{M}} + P_{\mathrm{q}} + D_{\mathrm{m,y}} \tag{17}$$

2.3.2.5 Sensitivity Runs

I performed n = 17 sensitivity trials for comparison with the Base model (Table 2.2). These included 4 monthly and yearly variations of catchability, 3 deviations of natural morality, 2 runs evaluating the bounds of adult tagging mortality, and 4 runs evaluating the influence of the prior. I then tested the sensitivity of the estimates when the month of April recoveries and effort were censored. For one test, all April data and effort were taken out of the model entirely. For another two tests, April was not allowed to be estimated separately from the constant, and was fixed at the constant. For the final sensitivity test I removed all fishing effort from 1970, as there were no tags recovered from this first tagging year.

I confined the deviations of the 4 sensitivity runs of monthly and yearly catchability variations using a sum of squares on the log scale to penalize departure from the estimated constant *q* value.

$$P_{\mathrm{q}} = \sum_{m} \sum_{y} \left[\ln\left(Q_{\mathrm{m,y}}\right) - \ln(q)\right]^2 \tag{18}$$

All penalties disregarded months with no effort, indicated by values of 0. Penalty values were used in the calculation of the negative log likelihood of the objective function. Deviations from *q*, which only occurred for months with recoveries, were estimated in the log scale and initialized at the natural log of, 2.7182, or at 1. Sensitivity models AE.c and AE.c_m are deviations of two of the catchability time varying models. For these sensitivity models all months of April were assigned the constant *q* value and therefore April was not allowed a varied catchability estimate. These catchability sensitivity models explore the potential time variation in the scaling of fishing mortality and the possibility of these time variation models better fitting the data.

The natural mortality *M* variations explored in the sensitivity runs included deviations in the natural mortality from the constant natural mortality estimate for each month (variation in months but not years, M_m), each year (M_y), and each month and year combination (M_{my}). Variations in natural mortality were constrained by using a penalty, which was added to the total likelihood estimate. These deviations were estimated in the exponential scale.

$$D_{\mathrm{m}} = \sum_{m} [\ln(M_{\mathrm{m}}) - \ln(1.0)]^2 \tag{19}$$

$$D_{\mathrm{y}} = \sum_{y} \left[\ln\left(M_{\mathrm{y}}\right) - \ln(1.0)\right]^2 \tag{20}$$

$$D_{\mathrm{m,y}} = \sum_{m,y} \left[\ln\left(M_{\mathrm{m,y}}\right) - \ln(1.0)\right]^2 \tag{21}$$

Months with no recoveries were assigned M_c. These parameters are considered partial parameters due to their dependence on the main constant natural mortality estimate M_c. These time varying natural mortality sensitivity runs provide time series of natural mortality, which are currently unavailable, and provide a better fit of the data by way of accounting for time in estimation of natural mortality.

I tested the impact of the informative prior M_p by changing the value to 0.07, 0.08, 0.1 and 0.11 m^{-1} for the 4 sensitivity runs. These runs equate to annual priors of 0.84, 0.96, 1.2, and 1.32 y^{-1}. The alteration of the priors serves to change the mean value of the P_M normal distribution and increase the P_M likelihood component if the M_c value is consistent to the Base model estimate. These variations in prior values will test the precision of the parameter estimates and the influence of the prior value. The range of adult tagging mortality found in a previous study was addressed in the sensitivity runs. I ran the models G.1 and G.3 with an adult tagging mortality of 10% and 30% respectively (Ahrenholz, 1981; Ahrenholz et al., 1991; Kroger & Dryfoos, 1972) to determine if a low or high tagging mortality would affect the parameter estimates. The sensitivity model testing the exclusion of April from the model, censoring both effort and recoveries, (model NE) was run to test the effect April exclusion would have on model fit and if it would remove some of the mathematical constraints it imposed. April had such low recoveries and effort that I wanted to evaluate if removing it entirely would provide more flexibility in the recovery estimates. Finally, I ran a sensitivity model with no effort, and therefore no estimation of any parameters, for the year 1970 (model B.1970). I did this to see if removing this year, which had releases that began in the later part of the year and no tag recoveries, would change the parameter estimates after removing superfluous estimated months that had no data to inform them.

I performed a percent bias (PBIAS) calculation to quantitatively evaluate model fit between the Base and sensitivity models. The PBIAS is the percentage of how overestimated or underestimated the sensitivity recoveries are when compared to the Base model estimated recoveries. Positive PBIAS indicates overestimation and a negative PBIAS indicates underestimation.

2.4 Results

The magnet efficiency tests were conducted in 14 of the 19 years, from 1 plant to all 11 plants in any given year. Tests were not conducted in any plant in the years 1972, 1974, 1982, and 1988, and only 2 tests, both in April, were conducted in 1985. Prior to averaging all plants together for each year, the magnets had an efficiency rate from 5% to 80% depending on year and plant, and a mean of 42%. Of these efficiency rates, 65.3% of the magnets had a 30% to 70% rate of tag recovery. Since the distribution per year of all the plants was roughly uniform, I calculated an unweighted average efficiency rate for each year. Years had a wide range of magnet efficiency rates, the smallest rate of 25% in 1971 and the largest rate of 69% in 1976 and approximately 67% of efficiency rates greater than 40% (Table 2.3). All April months were given the monthly estimate for April, 17%, to account for the significant difference in that month's testing.

The catchability estimate, q, was considered uniform across all months and years in the Base model. Parameter q was only applied to months with fishery landings. I had an estimated constant catchability of 0.00032 thousand metric tons (mt). I also estimated a constant instantaneous natural mortality, M_c, of 0.09 m^{-1}, with an annual rate of 1.08 y^{-1} and 95% confidence intervals (CI) of 1.04 y^{-1} to 1.13 y^{-1} for all 19 years. My estimated overdispersion factor k had a value of 0.72. Estimated standard deviations and 95% CI for all estimates are in Table 2.4.

Fishing mortality, F y^{-1}, was calculated using a different definition of fishing effort than the stock assessment. The model's instantaneous annual F y^{-1} generally follows the trend of the BAM estimation. The maximum estimated F is 0.32 y^{-1}, and occurs during 1984, the year of largest recorded catch (Figure 2.2). The values for F

range from 0.14 to 0.32 y^{-1}. *F* was lower during the early years of the study and slowly increased, following the trend of increasing annual catch at the time. I slightly underestimated F y^{-1} during the first three years of the study due to my censor of catch occurring in March or November of those years.

I consistently overestimated recoveries for the months of April and May for all years except 1986 and 1987 (Figure 2.3) with a PBIAS of 110.5%. The observed recoveries for June and July oscillated in numbers between years after 1976. The Base model compromised by fitting between the oscillating peaks and troughs. For the month of August, the Base model followed the trend set by the observed recoveries, but alternated between underestimating and overestimating after 1976 until 1984, where the model underestimated August for every year afterwards. September and October do not have as great of fluctuations in observed recoveries. The estimated recoveries came close to matching September's observed trend and some of October's, but were overestimated in both September and October during the years 1979 to 1983, with a PBIAS of 27.7%.

I overestimated Base model recoveries for all months during 1971 and 1982, and were underestimated during June, July, and August in 1973 and 1981 (Figure 2.4). Observed recoveries were fewer for the early years of the tagging study (1971 to 1976); however, the Base model was able to fit to the early recoveries with a PBIAS of 9.2%. I underestimated recoveries in June through October for 1977, 1986, and 1987 and underestimated recoveries from April through July in 1978, 1984 and 1985. Overall, I calculated a PBIAS of 7.2% for Base model when compared to the observed recoveries and a RMSE of 51.48.

2.4.1 Sensitivity Analysis

Some of the sensitivity runs failed to converge with a criterion of 0.0001 (C_{my}, M_m, M_y, M_{my}, and $AE.c_m$, gradient > 0). Most of the sensitivity models had negligible differences in expected recovery estimates from the Base model I ran. However, a couple of sensitivity models did have much higher differences in the estimated recoveries. Approximately 65% of estimated recoveries under model C_y were lower than the Base model recoveries, differing from -78.48 to 48.13 tag recoveries for a given month and a PBIAS underestimation tendency of -8.5% with a RMSE of 19.77 when compared to the Base model. Compared to the observed data model C_y had a RMSE of 42.66. The estimate recoveries under the C_y model were lower from 1978 to 1985 in all months but April (Figure 2.5). The estimate recoveries under C_y were also lower for almost all months in 1971 and 1978 to 1985 but were greater than the Base model for all months in 1973, 1977, and 1986, and for all months but April in 1987 (Figure 2.6). Over 85% of the estimated recoveries under model NE were greater than the Base model with an overall range of differences from -39.01 to 43.18 estimated recovered tags. Model NE had an average tendency to overestimate, with a deviation (PBIAS) of 10% and had a RMSE of 14.57. The estimated recoveries under NE were equal to or greater than the Base model for every year (Figure 2.7), with the exception for the month of April, since model NE did not have April recoveries (Figure 2.8). The C_m model was the other sensitivity model with considerable difference in estimated recoveries from the Base model. Model C_m did not on average have greater or lower recoveries than the Base model, with an even 50 to 55% of recoveries greater or lower than the Base and had a PBIAS close to 0 at -1% with a RMSE of 10.69. The estimated recoveries under C_m were lower for all May months and for most of the June months except for the years 1978 and 1979 (Figure 2.5). During the month of July, the expected recoveries under C_m were at or above the Base model estimations. During August and September, the C_m model alternated between slightly lower expected recoveries and slightly higher recoveries. Model C_m had higher recoveries than the Base model for the whole month of October. All sensitivity models estimated roughly equal recoveries to the Base model for the years 1972, 1974, 1975, 1976 and 1988. The rest of the sensitivity models (AE.c, C, B.1970, G.1, G.3, P0.07, P0.08, P0.1, P0.11) had approximately the same expected recoveries as the Base model (Figure 2.8 and 2.9. The difference in estimated tag recoveries of model C from the Base model ranged from -1.46 to 0.36 expected recovered tags with a PBIAS of 0 and a RMSE of 0.28. Model AE.c ranged from -4.57 to 2.25 expected tags with a PBIAS of 0.4% and a RSME of 1.02. Model B.1970 had a range of different expected tag recoveries from -0.64 to 0.29 with a PBIAS of -0.2% and a 0.2 RMSE. G.1 had different estimated recoveries from -1.71 to 8.22 with a PBIAS of 1.5% and a 2.12 RMSE. Model G.3 had a difference of -14.09 to 4.07 in expected recoveries from the Base model with a PBIAS of -2.2% and a RMSE of 3.59. All of the prior sensitivity models estimated recoveries with less than 1 tag difference from the Base (Figure 2.9 and 2.10), with PBIAS values of 0 for all runs and RMSE less than 1. Model G.1 had 8 more expected recoveries in 1983 and model G.3 had 14 less expected recoveries the Base model in 1983, with other lower recoveries in 1984 and 1986, but the afore mentioned sensitivity models had very little differences otherwise (Figure 2.8 and 2.9).

I found that most of the sensitivity models also had little difference in the q estimations, except for two. I estimated different q estimates from the Base model for the models NE and C_y (Figure 2.11b) (Table A2.3). I estimated a q of 0.00028 thousand mt for C_y, a -15% difference from the Base model with 68.2% CI (or 1 standard deviation) of 0.00031 to 0.00025 thousand mt. I estimated a q of 0.00038 thousand mt for NE, a 15% difference from the Base model and a CI of 0.00041 to 0.00034 thousand mt. Both model's CI fall within the 68.2% CI of the Base model estimate, but their CI do not overlap with the Base model q estimate. C_y had a range of $c_y * q$ deviation estimates from 0.00058 to 0.00016 thousand mt. I estimated a q of 0.00029 thousand mt for model C_m, a -10% difference from the Base model and model AE.c was estimated at 0.00028 thousand mt; a -12.5% difference from the Base model. The estimated parameter q for all other sensitivity models fell within the first standard deviation of the Base model.

My constant M_c estimations again had very similar estimated values for all sensitivity models (Table A2.2). I estimated a constant natural mortality of 0.091 m^{-1} for both NE and C_y models, a 1% higher monthly instantaneous M_c than the Base model (Figure 2.11a), but their estimates fell in the CI of the Base model. C_y had a 95% CI of 1.047 y^{-1} to 1.14 y^{-1}, a 1% increase from the Base. All other sensitivity models had the same percent difference or smaller.

I estimated the overdispersion factor k for models NE and C_y at 0.84, a 16% greater difference from the Base model. Both models' estimate and CI fell outside of the Base model's 68.2% CI (Figure 2.11c) (Table A2.4). All other sensitivity model estimates overlapped with the CI of the Base model, with models C_m and AE.c possessing greater estimates, 4% and 1.5% respectively, than the Base.

2.5 Discussion

In this work, my primary results consisted of 1) estimated magnet efficiency rates for each year of the study, 2) a new natural mortality estimate for Gulf menhaden, 3) a fishing mortality time series estimated using a different effort approach, 4) a catchability estimate for the model, and 5) an overdispersion estimate to better fit a zero-inflated distribution. The range of plant test recoveries I reported is comparable to past Gulf menhaden mark-recovery studies (Pristas et al., 1976). I chose not to vary magnet efficiency by month due to the Base model failing to reach the convergence criteria when I ran it with a month varying magnet efficiency, and from the results of a negative binomial estimate of trials per month (varying by month and not year). These results indicated a less than 8.6% difference in the efficiencies between all months except April, whose 17% efficiency rate was about 40% less than the other months. All months of April were therefore given the month specific magnet efficiency rate, though for some years this could be an underestimation since only 4 tests were ever performed on magnet efficiency in April. April had relatively low fish landings across years, and had a very low number of observed recoveries, so I decided to use April's 17% magnet efficiency rate to match what the data showed. I kept all other recovery months static annually due to the wider range of differences found in the annual negative binomial estimates and the plant negative binomial estimates. I decided to account for this greater variation of year and plant trials through a combined negative binomial. All 11 plants were tested in 1971, the year with the lowest magnet efficiency and the first year of tag recoveries, so it is possible that more magnets were installed to increase the efficiency rate in the years following. Only 3 plants were tested in 1976, the year of the highest magnet efficiency, and all had high efficiency rates. These high rates could be from an increase in tag recovery checks in some plants for the first year of adult tagging being included in the study. The reduction plants had some magnets installed previous to the study to remove metal scraps that came in with the fish (Ahrenholz et al., 1991) and magnets that were added to the plants for the study remained in the plant after the end of the tagging.

I consistently estimated natural mortality, M_c, with a monthly rate around 0.09 m^{-1}, and the annual rate around 1.08 y^{-1}, under the Base model and all sensitivity models. This demonstrates the precision and accuracy in the estimate and validates the range of the estimates set down by Ahrenholz (Ahrenholz, 1981) and used by the current stock assessment (GDAR03, 2021; SEDAR63, 2018). My annual natural mortality 95% CI of 1.04 y^{-1} to 1.13 y^{-1} is higher than Ahrenholz's reported 95% CI from the 3 year tagging study (Ahrenholz, 1981; SEDAR63, 2018), and my annual instantaneous natural mortality rate of 1.08 y^{-1} is slightly lower than Ahrenholz's recommended rate of 1.094 y^{-1}. The stock assessment rounds this estimate to 1.1 y^{-1} for their value of natural mortality. My upper bound of 1.13 y^{-1} encompasses this current rate. My mean estimate falls within the confidence intervals set by Ahrenholz and within the scope of the BAM model sensitivity runs (SEDAR63, 2018), and therefore provides validation that the current mortality rates used in the stock assessment are within reason for the population's natural mortality as is estimated by current science.

The estimated mortality I reported is for all adult Gulf menhaden selected by the fishing gear of the reduction fishery, which includes age-1 to age-6, with a tendency towards selection of age-1 and age-2 fish. I assumed mortality was the same for all adult ages. Fish were not measured at time of tagging, so I cannot estimate their age at tagging.

The data only indicates which life stage the fish were in, adult or juvenile. I was unable to provide a time series of natural mortality since the varying *M* models were unable to converge due to the maximum gradient. I believe this is due to the low number of recoveries, low number of releases, and lack of information available in the data. A greater number of tagged fish would have provided a more accurate depiction, since fish tag studies do not recover many tags and Gulf menhaden are such a numerous forage fish that 232,223 tagged fish do not represent a large portion of the population. However it is worth noting that this study is one of the largest fish tagging studies ever performed for an individual species. Many fish tag studies to not reach the 100,000+ number of tags released, as either the fish species is not a numerous species or there is a lack of time or materials to carry out such a large study. The National Oceanic and Atmospheric Administration (NOAA) has a tagging program for more than 70 highly migratory species that has been ongoing since 1954 and in that time have tagged 270,000 fish (*Atlantic Highly Migratory Species: Cooperative Tagging Program*, 2023), whereas in this study over 230,000 of one species were tagged within 15 years. Species such as striped bass (Corbett, 2008) have long term tagging studies that have enabled them to tag over 100,000 fish, but many studies fluctuate between a couple thousand (Craig et al., 2015; Johnson et al., 1993; Patterson et al., 2001) and less than 100 fish tagged (Farmer & Ault, 2011; Hammerschlag et al., 2011).

Annual fishing mortality for 1971, 1972, and 1973 were underestimated from the censorship of March and November landings for those years. Landings for these years were nominal compared to general monthly landings during the peak of the season and had minimal fishing catch in those months before 1973 and no fishing in those months after 1973. These landings were therefore censored to develop a more robust model. An official fishing season of April to October was implemented in 1977 (Smith, 1991), but there was little to no catch in March and November in years prior to the official fishing season. I defined effort as thousands landings per month while the stock assessment used vessel-ton-weeks (SEDAR63, 2018). The scale of the calculated F y^{-1} is therefore different than the F y^{-1} estimated by the BAM model. I chose to use monthly landings as the effort unit from lack of data to calculate the monthly vessel-ton-week.

Model C_y had the lowest estimation of catchability, q, due to its annual variation estimates. C_y allows the years to have their own individual c deviations around q, and its range of $c_y * q$ deviation estimations stretched from 0.00058 to 0.00016 thousand mt around its estimation of 0.00028 thousand mt. Even then, C_y model's parameter q estimate only diverged from the Base model estimation of 0.00032 thousand mt by -15%, the largest divergence of any of the sensitivity models. Model NE also diverged 15%, but had no deviations of q. This estimation therefore was a result of the likelihood fitting after the month of April had been removed from the observed recoveries. Low q values decrease the fishing mortality, which decreases the total mortality. A smaller total and fishing mortality will decrease the probability of observation, reducing the number of expected recoveries. This is the reason why the C_y model has lower estimated recoveries in general than the Base model, and a reason why the NE model has such higher estimated recoveries.

Model C_y had a lower RMSE than the Base model, and therefore fit the observed data better than the Base model did. The Base model had a RMSE of 51.48 and C_y a RMSE of 42.66 when compared to the observed data. The sensitivity runs, when compare to the Base model, had lower RMSE. All runs but C_y, NE, and C_m had RMSE under 4. The afore mentioned 3 runs had RMSE values over 2.5 times higher than the other sensitivity runs.

The overdispersion parameter, *k*, estimates were within the 1 standard deviation bounds for all sensitivity models except two. Models NE and C_y had approximately equal *k* estimates that were 16% greater than the Base model estimated *k* of 0.72. The larger the *k*, the less the right skew of the negative binomial distribution, so NE and C_y have a slightly less right skew than the base model. A right skew is found when a curve on a graph has a longer 'tail' on the right side of the curve that gradually tapers off. This indicates the observations or values with a gradual decrease are found on the right side of the x axis.

In the comparison of observed and expected calculated recoveries, I overestimated April and May for almost all years. The PBIAS of 110.5% for April and May indicates that these two months had a strong average tendency to overestimate. These were the first months in the following year after tagging in which tags could be recovered, so the model overestimated the number that were recovered due to the abundance of live tagged fish in the population. Adult and juvenile fish were tagged at the end of the fishing season and any juveniles that were tagged earlier in the season had not recruited to the adult population yet, so no tags were ever recovered in the year the fish were tagged. These overestimations likely caused a mathematical constraint on the likelihood model, as well as increased the general inflexibility the model showcased when fitting to the observed recoveries. The Base model and the sensitivity models were unable to closely follow the oscillating recoveries in the same month between years, which resulted in common overestimation and underestimation. The Base model also could not fit the distribution of each year's observed recoveries closely after 1976 when the adult tagged fish were introduced into the study. This is likely due to the large increase in recoveries that followed the first 5 years of lower recoveries, which the model had a hard time fitting with the parameters I gave it. Overall, the Base model had a weak average tendency across the study to overestimate recoveries (7.2%).

Of the catchability deviation sensitivity models, only C_y and C_m had a large difference in expected recoveries from the Base model. C_y on average estimated lower recoveries from the Base model -8.5% of the time. The annual variation in catchability allowed it to estimate the low observed recoveries during certain years better than the Base model, but also increased the underestimation of the high observed recovery months. C_m differed uniformly from the Base model in the amount of greater or lower differences in expected recoveries, the monthly variation in catchability enabling it to estimate low for the whole month of May in its effort to decrease overestimation.

I chose the adult tagging mortality from a range of 10 to 24% (Kroger & Dryfoos, 1972), using the median value of 17%. Changing the adult tagging mortality from 0.17 to 0.1 or 0.3, the range tested by Ahrenholz (Ahrenholz, 1981), had little to no effect on expected recovery estimation nor *M* estimation. Both the G.1 and G.3 models only differed from the Base model at most by 8 and 14 recoveries in 1983. This is an indicator that the range of adult menhaden tagging mortality explored does not affect the model fit of recovered tags. I used the Gulf menhaden juvenile tagging mortality estimated by Byars (1981) for tagged juveniles. No tagging mortality for adult Gulf menhaden has ever been estimated, so I used the Atlantic menhaden range of adult tagging mortality, assuming Gulf menhaden adults would have a similar tagging mortality to adult Atlantic menhaden.

My sensitivity analysis with the prior values show that my natural mortality prior has low influence on the model estimation of the parameters. For sensitivity models P0.07, P0.08, P.1, and P.11 in which I fixed the prior monthly natural mortality values at 0.07, 0.08, 0.1, and 0.11 respectively, all parameter estimates had statistically no difference from the Base model estimated parameters. This lack of divergence from the Base model and the similarity of my monthly natural mortality estimate of 0.09 m^{-1} from the 0.09 m^{-1} estimate suggested by Ahrenholz (1981) and used by the assessment (SEDAR63, 2018) indicates that the current natural mortality value used by the assessment is a valid and accurately estimated parameter.

Magnet efficiency estimates have a high chance of being affected by human error. Magnet efficiency trials were run to test the magnets, but did not differentiate the magnet capabilities from the human component during the season. I do not know how well magnets were cleaned during the fishing season, though it was recorded that they were cleaned on a daily or daily interval schedule (Ahrenholz et al., 1991). Error could have been introduced by lack of daily cleaning during the height of the season or at the end and beginning of the season. It is likely however, that the magnets and plants were thoroughly cleaned at the end of the season and swept through again before the start. The cleaning at the end of the season could cause a higher amount of recoveries present at the end of the season from more than just recapture. Any November recoveries could be from the last sets caught in October at the end of the season and not fully processed until the next month. I accounted for this by placing recoveries from the first 3 days of November back into the month of October. There is also a possibility that tags were not found the month they were brought in from the catch. This could inflate the time at large and therefore the estimated mortality.

I accounted for potential immediate tagging mortality and I assumed all fish had an equal chance of survival after tagging regardless of who tagged them as I had no way of knowing who tagged what fish. I ignored the potential error of constant tag shedding throughout study as it was not likely to occur due to the healing of the incision where the tag was inserted (Ahrenholz, 1981; Kroger & Dryfoos, 1972). I assumed in the model that all tagged fish had the same probability of recovery at each time step regardless of when they were released. I did see the majority of recovered tags in the first two years after tagged fish were released even though age was not known at time of tagging, so while the number of tagged fish available for recovery decreased as time went on, remaining fish still had the same chance of being caught by the fishery. Gulf menhaden are schooling fish, and any tagged cohort remaining together in a school could be recovered together, which would inflate the observed recoveries. However, the distribution of these fish in schools and the size of the schools is still a naturally occurring phenomenon that evens out this chance of inflation. Predators often target these schools of fish, so even in the wild schooling fish would be removed together during a feeding event. The data also shows that fish from the same release cohort were caught throughout the fishing season for several years. The fishery lands mostly age-1 and age-2 fish, as is consistent with the life history of Gulf menhaden, a fish that lives no more than 7 years at most and only rarely occurring in the national biostatistical data bases at age-6 (SEDAR63, 2018). Therefore, my natural mortality estimate is likely to represent more closely age-1 and age-2 Gulf menhaden natural mortality. To differentiate between adult fish and young-of-the-year, a study focused on larval and juvenile Gulf menhaden mortality rates might be useful to distinguish between the age-0 mortality and the adult mortality. Many fish species experience greater mortality as larvae and juveniles (Anderson, 1988; Fennie et al., 2020; Hoey & McCormick, 2004; Houde, 1987; Robert et al., 2010) so ascertaining if this holds true for Gulf menhaden could provide more information for the assessment and ecosystem models. Ecosystem models are now being frequently used to understand the effects of the fishing of multiple species in the ecosystem (Collie et al., 2016; De Mutsert et al., 2008) and creating a system management plan. These models use data from several fisheries to create a holistic model (National Research Council, 2006) that manages all of that ecosystem's fisheries while acknowledging each fisheries' impact on each other. To do this, information regarding all life stages of a fish species is needed. I estimated the adult natural mortality of Gulf menhaden, but a study on juvenile and larval menhaden would provide information about the early life stages.

Several datasets and studies were used in the application of this model to estimate the mortality rates besides the mark-recapture study. Separate testing of the magnets installed in the reduction plants were carried on throughout the 19-year mark-recapture study to better quantify the probability of observing the tags once they were recovered through the fishery. Previous tag shedding and methodology studies were done to determine the effect the actual tagging had on menhaden (Ahrenholz et al., 1991; Byars, 1981; Kroger & Dryfoos, 1972; Pristas et al., 1976; Pristas & Willis, 1973). Landings from the whole reduction fishery for each month were necessary as a proxy for fishery effort. Ahrenholz's 1970 tagging study with data from Pristas et al., 1976 gave an informative prior for the model to incorporate (Ahrenholz, 1981), and a previous tagging study from 1969 gave us insight into what the taggers classified as a 'juvenile' and what was an 'adult' sized fish.

Historical data sets can be useful in statistical analysis even for current populations. This model was built to analyze historical tagging data collected in the 1970s and 1980s, during the peak of the Gulf menhaden fishing efforts. This period of time covers the highest fishing mortality and events leading up to it, allowing the model to estimate a natural mortality for a period of increasing fishing mortality. The tagging study relied on the fishing industry for tag recoveries, and so was able to recover the greatest possible amount of tags for a Gulf menhaden tagging study during the years of the highest fishing industry effort. A current tagging study with the same number of tagged fish would objectively recover a lower number of tags due to lower fishing effort. My study based on historical data is an important contribution to the stock assessment, which has no tagging data incorporated in the BAM model except for the natural mortality parameter I am validating.

It is important for science-based stock assessments to continuously improve and validate their parameters as computing capability and scientific methodology grow and we gain better understanding of population functions and ecosystems. My model validates the instantaneous natural mortality range given by Ahrenholz and provides an updated natural mortality estimate that has been estimated using modern statistical methods. Unlike a similar model used for Atlantic menhaden, I was able to estimate the overdispersion value *k* (Liljestrand et al., 2019a). While I could estimate annual and monthly mortality rates, those models had a maximum gradient greater than 0 and were able to produce a determinant of the Hessian but could not converge, making their estimates unreliable. I was able to use the natural mortality rate estimated by Ahrenholz in the 1970s as a weak informative prior for my Bayesian model and I estimated an accurate and precise constant natural mortality of 1.08 y^{-1}, with 95% CI of 1.04 y^{-1} to 1.13 y^{-1}, which I propose as the new natural mortality value for the stock assessment.

2.6 Tables and Figures

Table 2.1 *Mathematical Symbols*

Descriptions and definitions of symbols used in models and equations.

Symbol	Definition and Description	Units	Values
Coefficients			
T	Time of release		1-228; January, 1970 to December of 1985
t	Time of recovery		1-228; January, 1970 to December of 1988
m	Month		1 - 12
y	Year		1 - 19
a	Stage of recovery		Adult
A	Stage of release		Juvenile or Adult
j	Recovery reduction plant		1 to 11
i	Number of magnet efficiency trials		1 to 11
Data			
R	Releases		Table A1.1, Figure 1.1b
r	Observed Recoveries		Figure 1.1a
E	Fishing effort in landings	1,000 metric tons	
H	Trial recoveries		
Stated Values			
G	Tag mortality	Percentage	Adult: 0.17, Juvenile: 0.30
s	Variance of normal distribution penalties		1
n	Total amount of fish batches in magnet trials		100
v	Number of magnet efficiencies		
M_p	Prior of M (Ahrenholz, 1981)	m^{-1}	0.09
Estimated Values			
k	Overdisperion value		
M	Instantaneous natural mortality	m^{-1} or y^{-1}	
M_c	Monthly constant instantaneous natural mortality	m^{-1}	
q	Constant catchability	1,000 metric tons	
c	Deviations in catchability		
Likelihood Function Components			
NLL	Total negative log likelihood		
NLL_r	Negative log likelihood for recoveries		
P_M	Prior penalty for constant natural mortality		
P_q	Log initial penalty for catchability		
NLL_ε	Negative log likelihood for magnet efficiency rates		
D	Log penalty for deviation from constant monthly M		

Table 2.1 (continued).

Symbol	Definition and Description	Units	Values
Calculated Values			
ε	Magnet efficiency	Percentage	Table 1.3
F	Instantaneous fishing mortality	m^{-1}	
Z	Instantaneous total mortality	m^{-1}	
Q	Total catchability		
N	Abundance of fish after tagging mortality		
$ř^{1}$	Expected recoveries from month of release		
$ř^{*}$	Expected recoveries for months after release		
ř	Expected recoveries after accounting for probability of observation		

Table 2.2 *Model Permutations*

Base model and sensitivity model combinations. M deviations indicate presence of time varying deviations on the constant M_c estimated parameter. The same is true for q deviations, which indicate if that sensitivity run had any time variations of the parameter q. SD stands for standard deviation, which applies to the normal distribution of the P_M likelihood component. The table indicates whether or not the month of April was censored or not estimated separately like other months for the runs where months were estimated separately. I also indicate whether fishing effort was censored in some runs, if there was any change in the adult tagging mortality value, or if the informative prior for M_c was altered from the base value.

Model	M deviation	q deviation	SD	April Presence and Estimation	Fishing Effort	Adult Tagging Mortality	Prior
Base	-	-	1	Y	No change	0.17	0.09
C	-	c	1	Y	No change	0.17	0.09
C_m	-	c_m	1	Y	No change	0.17	0.09
C_y	-	c_y	1	Y	No change	0.17	0.09
$C_{m,y}$	-	c_{my}	1	Y	No change	0.17	0.09
M_m	M_m	-	1	Y	No change	0.17	0.09
M_y	M_y	-	1	Y	No change	0.17	0.09
$M_{m,y}$	M_{my}	-	1	Y	No change	0.17	0.09
P0.07	-	-	1	Y	No change	0.17	0.07
P0.08	-	-	1	Y	No change	0.17	0.08
P0.1	-	-	1	Y	No change	0.17	0.1
P0.11	-	-	1	Y	No change	0.17	0.11
G.1	-	-	1	Y	No change	0.1	0.09
G.2	-	-	1	Y	No change	0.3	0.09
NE	-	-	1	Censored Recoveries	No April effort	0.17	0.09
AE.c	-	c	1	No separate April estimation	No change	0.17	0.09
AE.c_m	-	c_m	1	No separate April estimation	No change	0.17	0.09
B.1970	-	-	1	Y	No effort for 1970	0.17	0.09

Table 2.3 *Magnet Efficiency Rates*

Estimate magnet efficiency rates for all years. Years that had no magnet tests were assigned the mean annual efficiency rate of 0.42.

Year	**Magnet efficiency, ε**
1971	0.25
1972	0.42
1973	0.36
1974	0.42
1975	0.33
1976	0.69
1977	0.48
1978	0.51
1979	0.52
1980	0.49
1981	0.48
1982	0.42
1983	0.57
1984	0.46
1985	0.42
1986	0.32
1987	0.33
1988	0.42

Table 2.4 *Model Estimates for All Parameters*

Base model estimated values for parameters *M*, *q*, and *k* with log scale standard deviation and bounds of the 95% confidence interval (CI).

Parameter	Estimate	Standard deviation in log scale	Upper 95% CI	Lower 95% CI
M	0.09	0.021	0.094	0.086
q	0.000322	0.094	0.000387	0.000268
k	0.72	0.073	0.83	0.62

Figure 2.1 *Released and Recovered Fish*

a) Total fish recoveries per year from Ahrenholz's mark-recovery study (Ahrenholz, 1981), b) Number of tagged fish released each year.

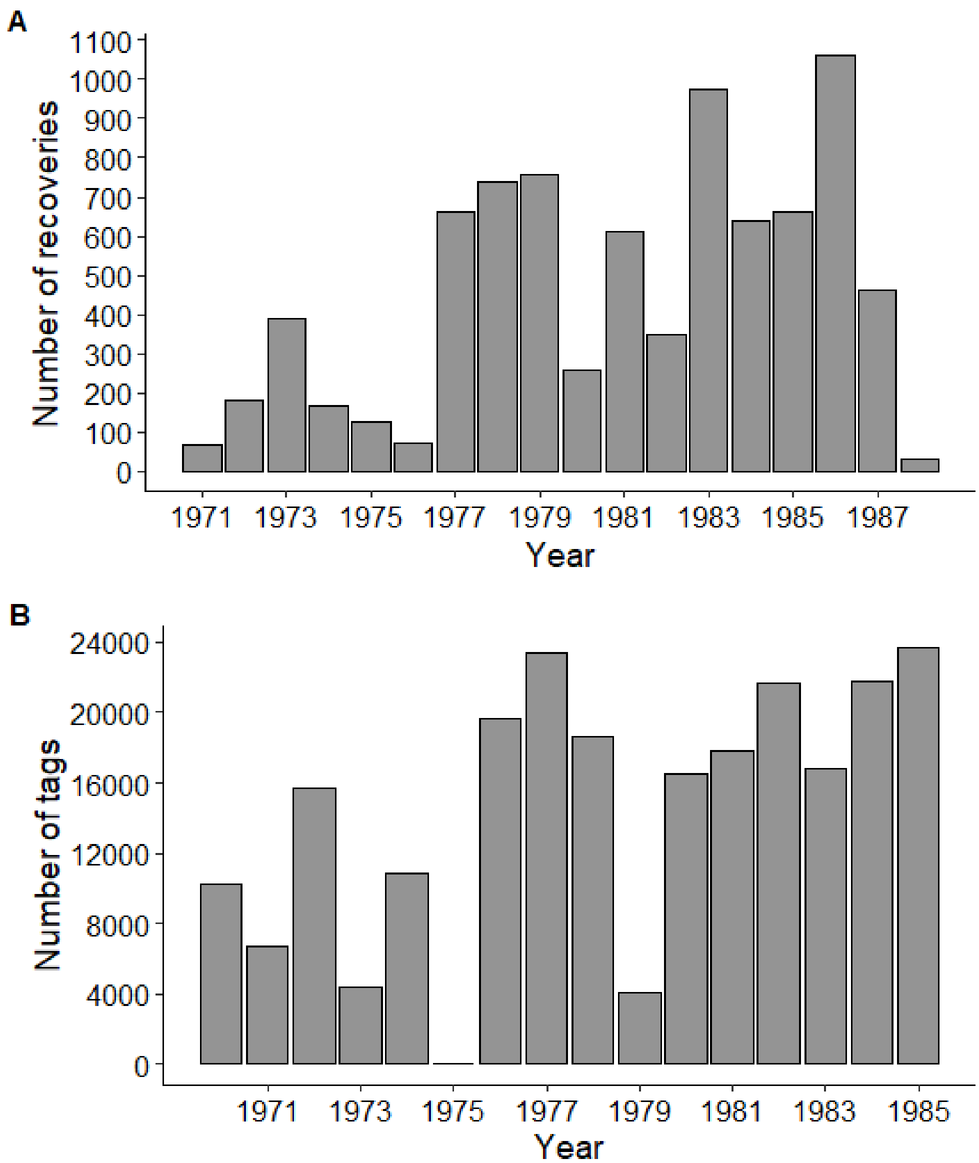


Figure 2.2 *Fishing Mortality*

Estimated yearly instantaneous F y^{-1} values from the Base model run.

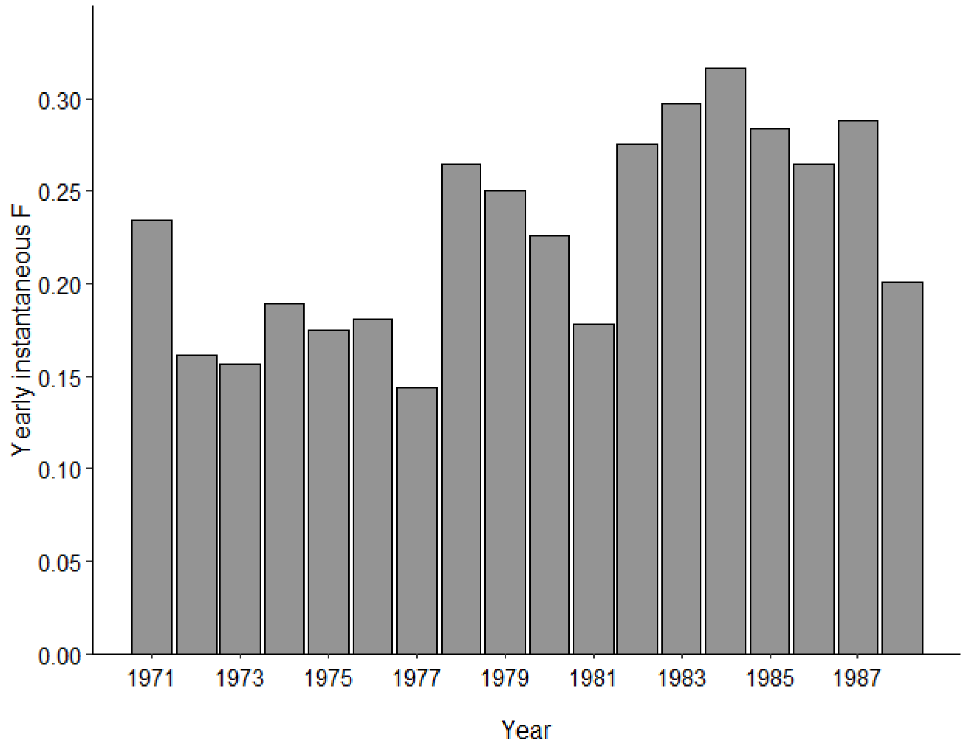


Figure 2.3 *Base Model Monthly Expected Recoveries*

Comparison of the Base model monthly estimated and observed recoveries throughout the tagging study. The open points represent the observed recoveries and the black lines are the estimated recoveries.

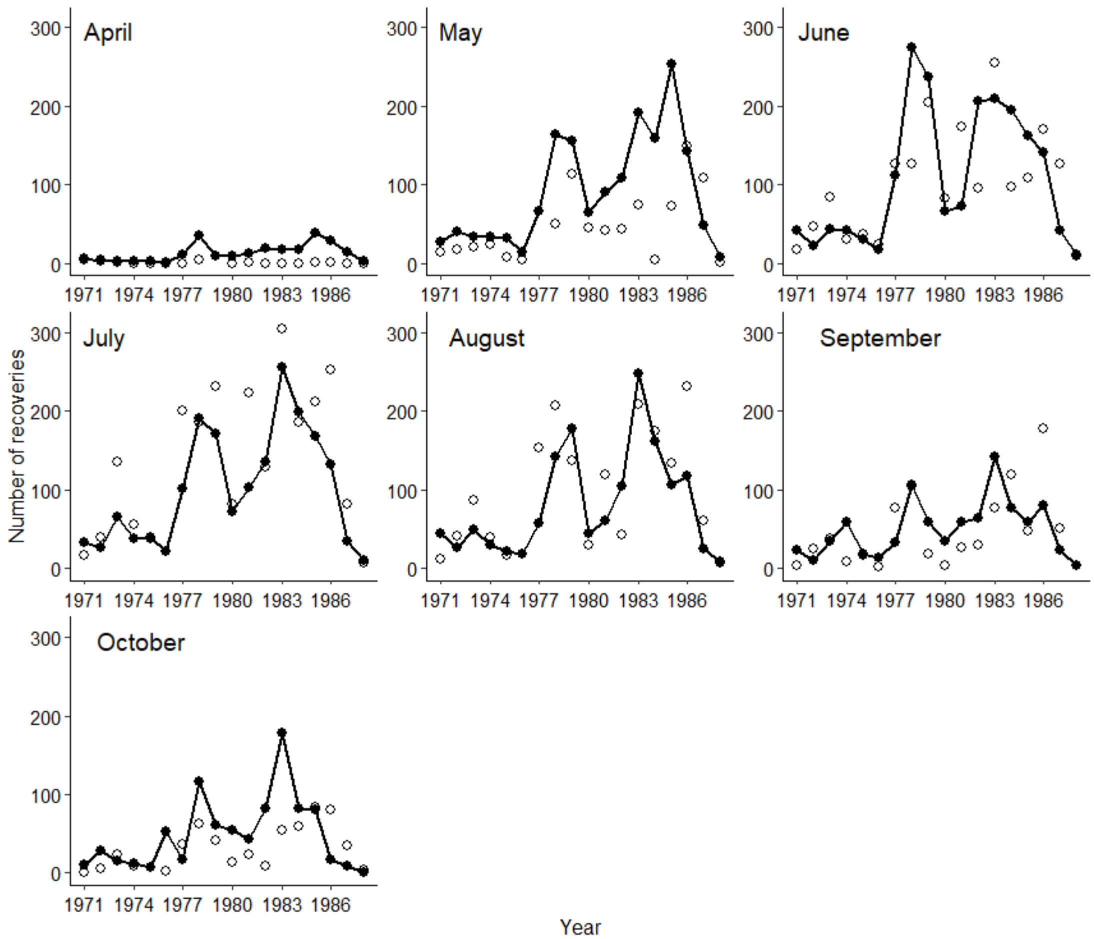


Figure 2.4 *Base Model Annual Expected Recoveries*

Comparison of the Base model annual estimated and observed recoveries throughout the tagging study. The open points represent the observed recoveries and the black lines are the estimated recoveries.

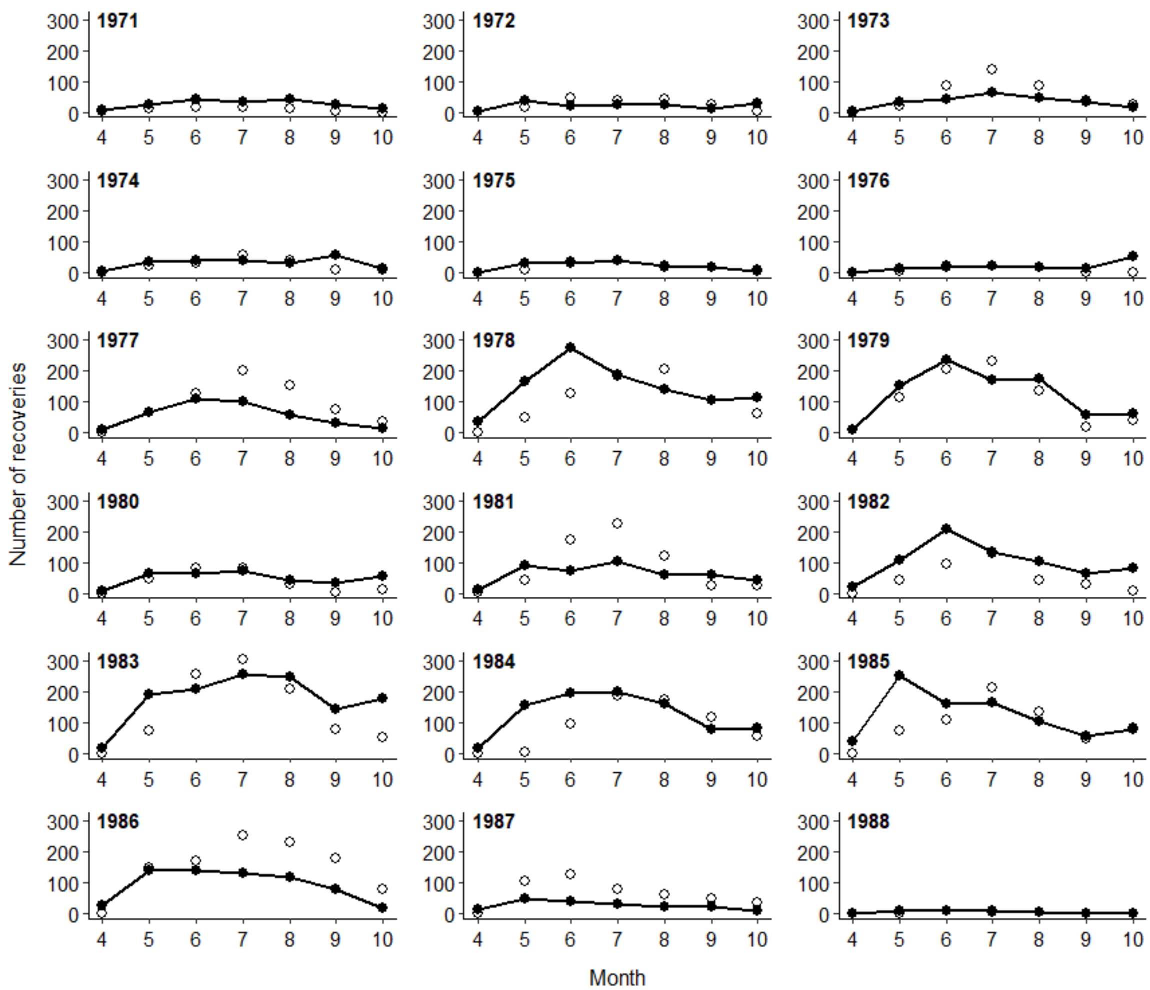


Figure 2.5 *Monthly Expected Recoveries for c Deviations*

Comparison of monthly estimated recoveries of sensitivity models with *c* deviations and the Base model. The Base model is in red points, model AE.c in teal triangles, model C in light blue diamonds, model C_m in black circles, and model C_y in dark blue squares. Sensitivity runs that did not converge are not included.

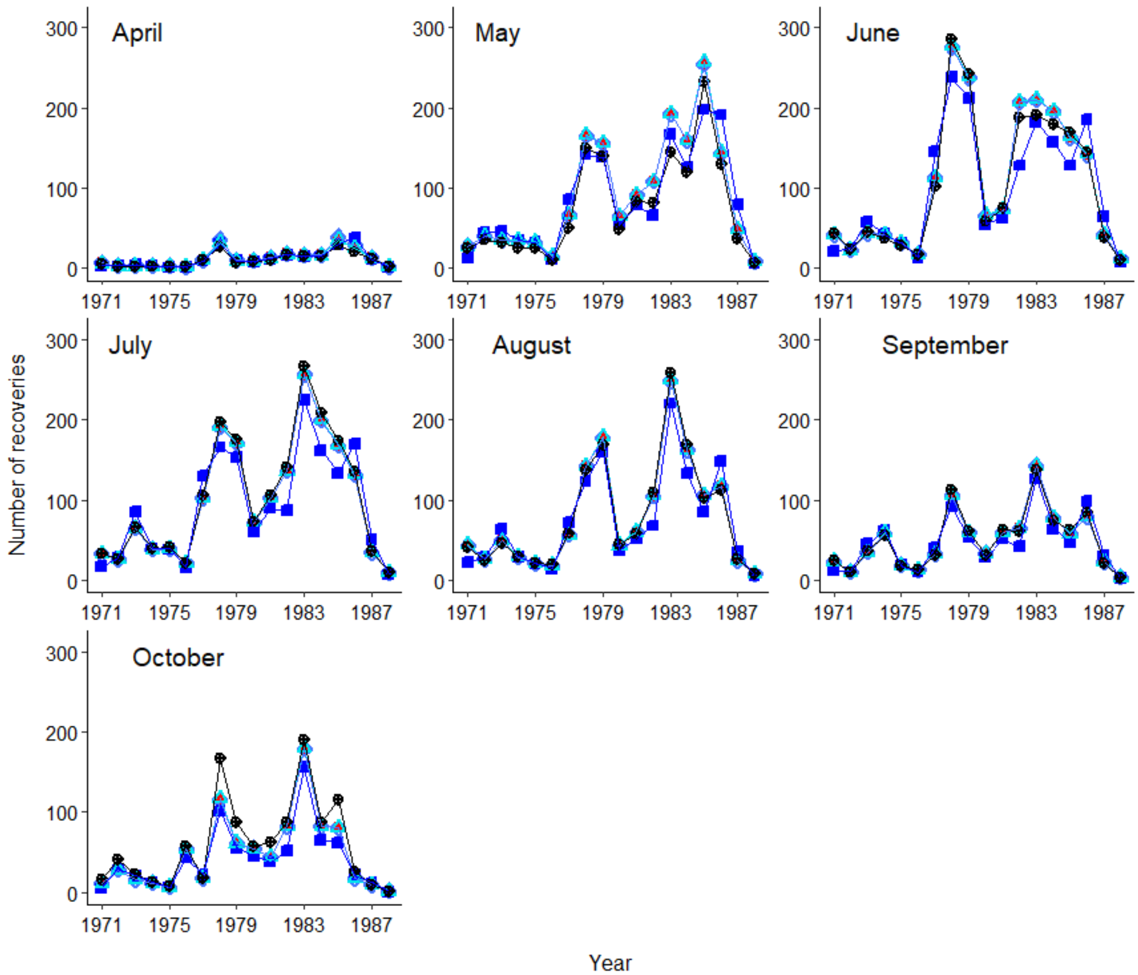


Figure 2.6 *Annual Expected Recoveries for c Deviations*

Comparison of annual estimated recoveries of sensitivity models with *c* deviations and the Base model. The Base model is in red points, model AE.c in teal triangles, model C in light blue diamonds, model C_m in black circles, and model C_y in dark blue squares. Sensitivity runs that did not converge are not included.

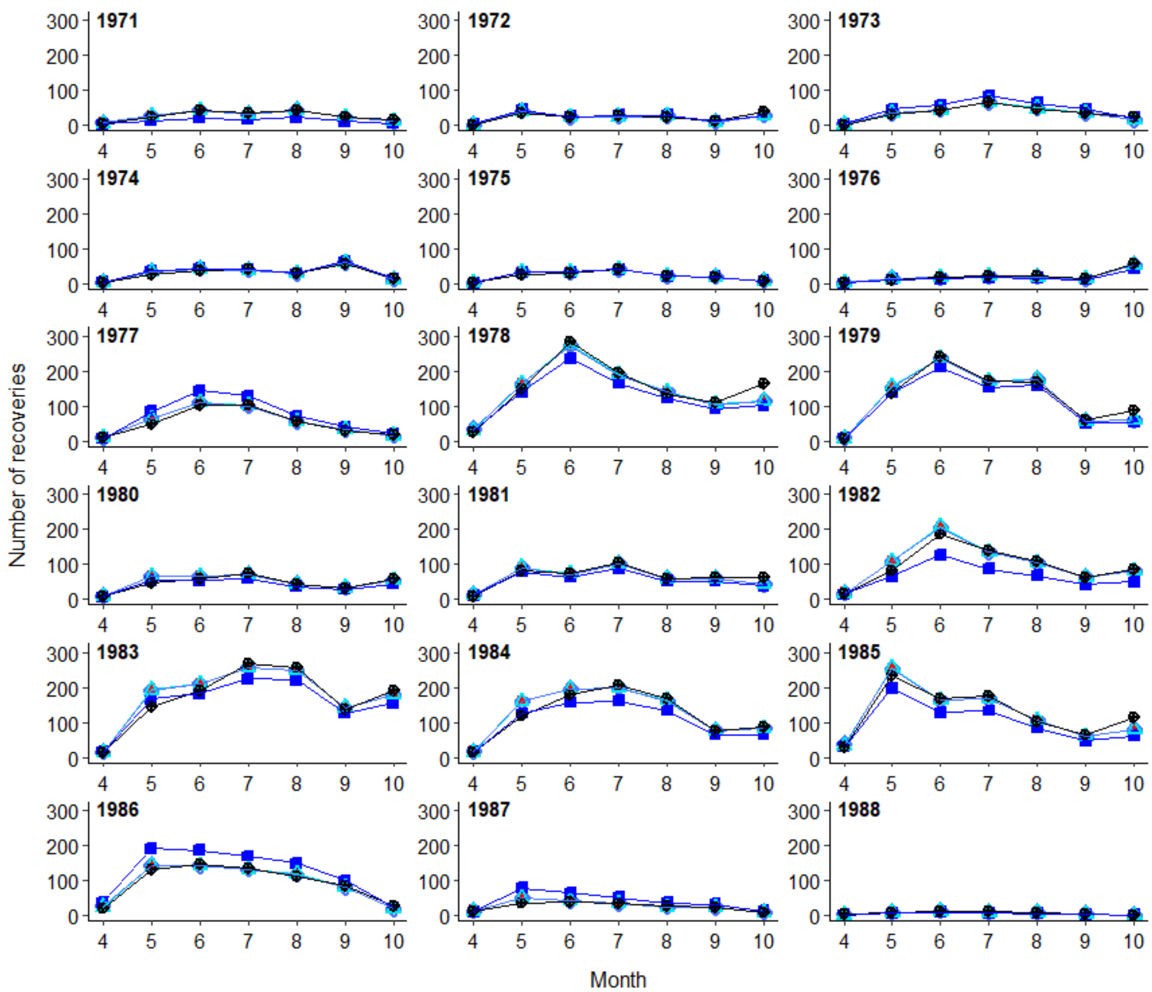


Figure 2.7 *Annual Expected Recoveries for Other Sensitivity Models*

Comparison of annual estimated recoveries of sensitivity models with all other deviations and the Base model. The Base model is in red points, model B.1970 in purple triangles, model G.1 in pink diamonds, model G.3 in black circles, and model NE in light blue squares. Sensitivity runs that did not converge are not included.

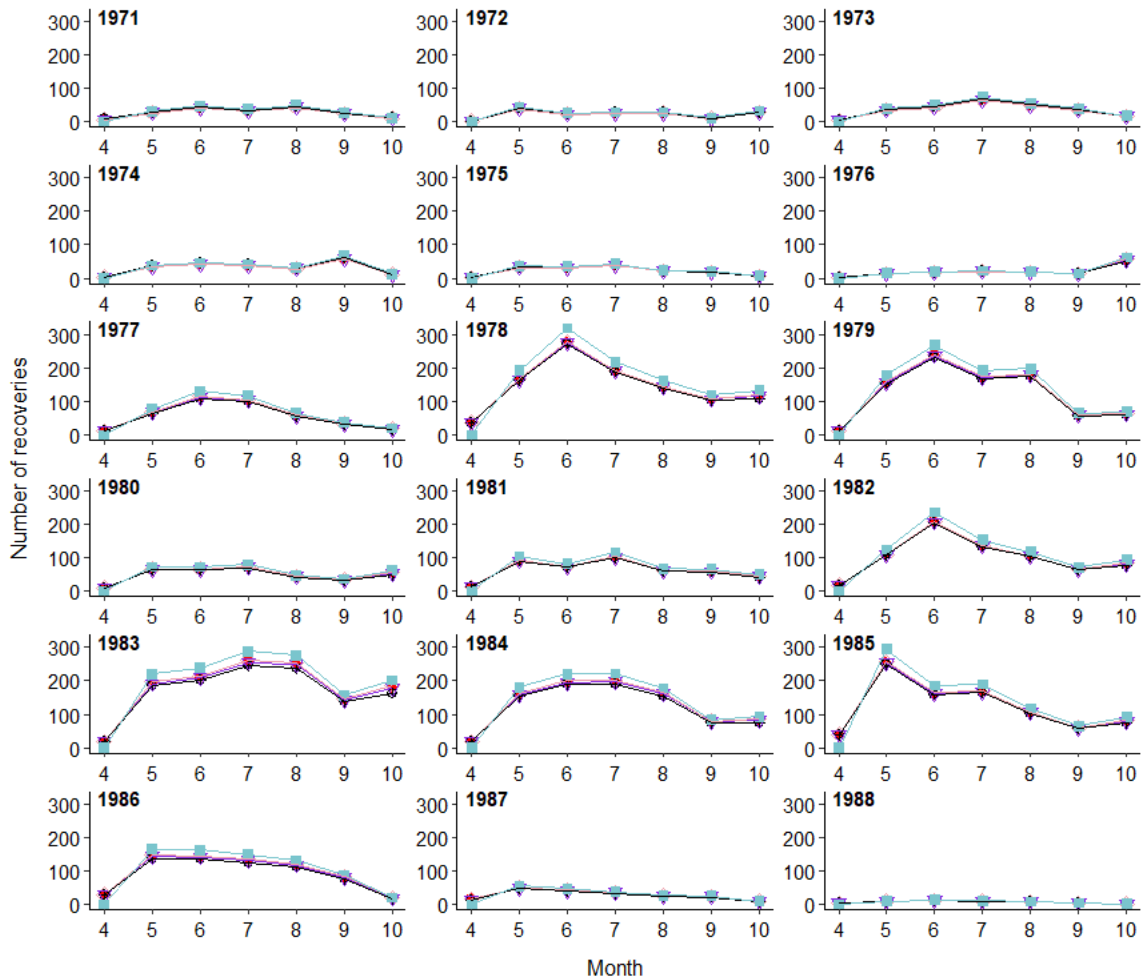


Figure 2.8 *Monthly Expected Recoveries for Other Sensitivity Models*

Comparison of monthly estimated recoveries of sensitivity models with all other deviations and the Base model. The Base model is in red points, model B.1970 in purple triangles, model G.1 in pink diamonds, model G.3 in black circles, and model NE in light blue squares. Sensitivity runs that did not converge are not included.

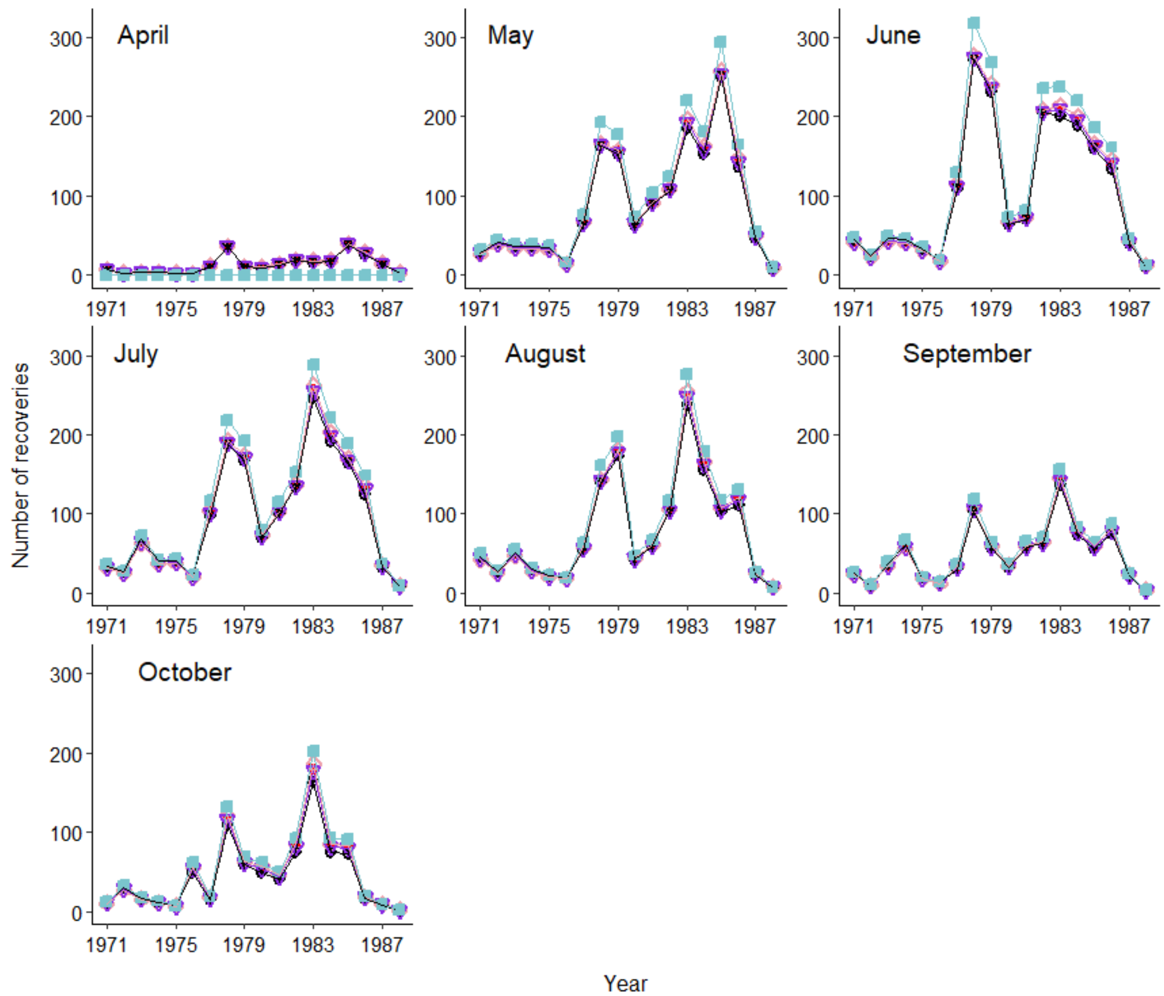


Figure 2.9 *Monthly Expected Recoveries for Prior Sensitivity Models*

Comparison of monthly estimated recoveries of sensitivity models with prior variations and the Base model. The Base model is in red points, model P.1 in lime green triangles, model P.11 in orange diamonds, model P0.07 in black circles, and model P0.08 in light pink squares.

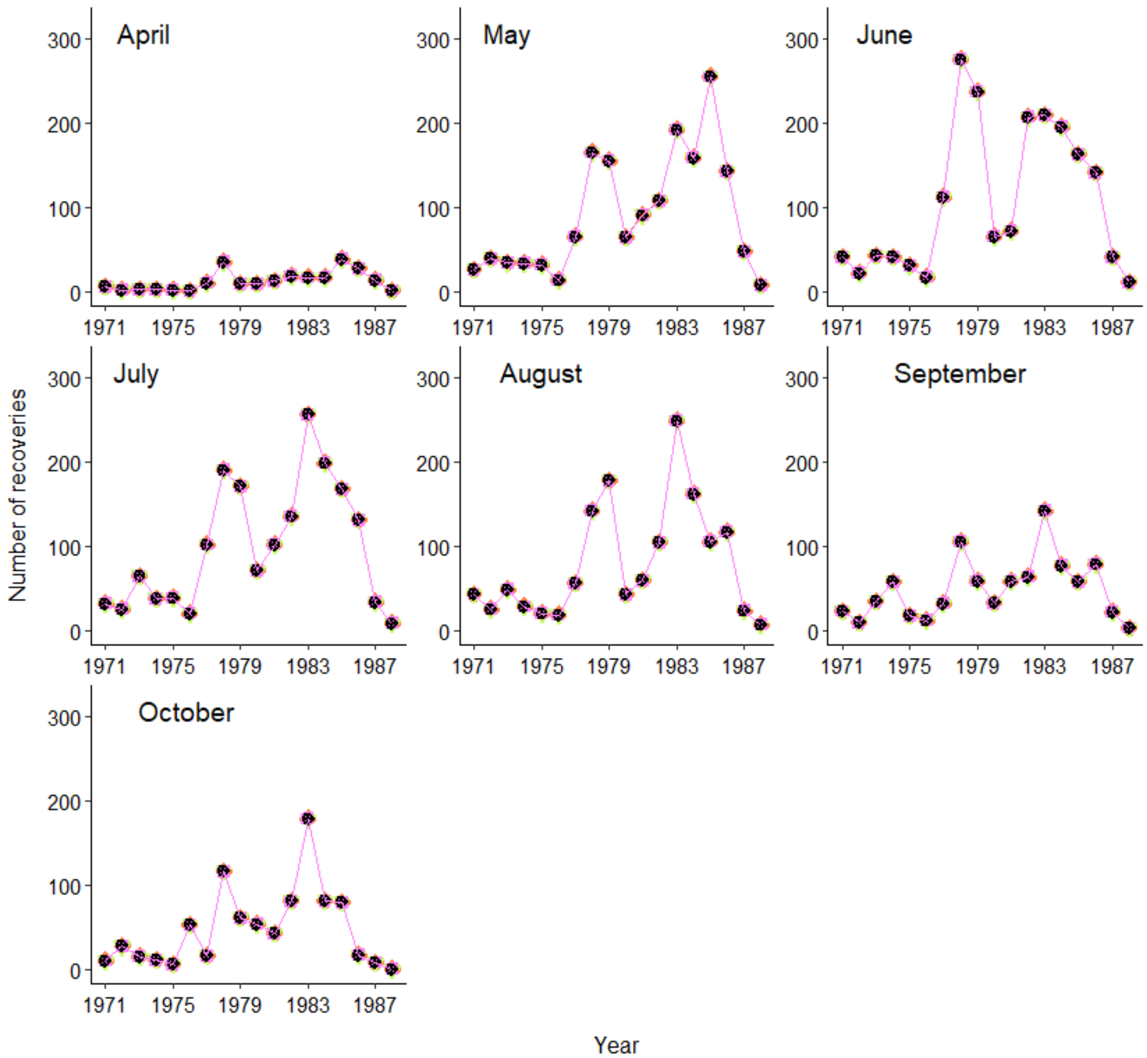


Figure 2.10 *Annual Expected Recoveries for Prior Sensitivity Models*

Comparison of annual estimated recoveries of sensitivity models with prior variations and the Base model. The Base model is in red points, model P.1 in lime green triangles, model P.11 in orange diamonds, model P0.07 in black circles, and model P0.08 in light pink squares.

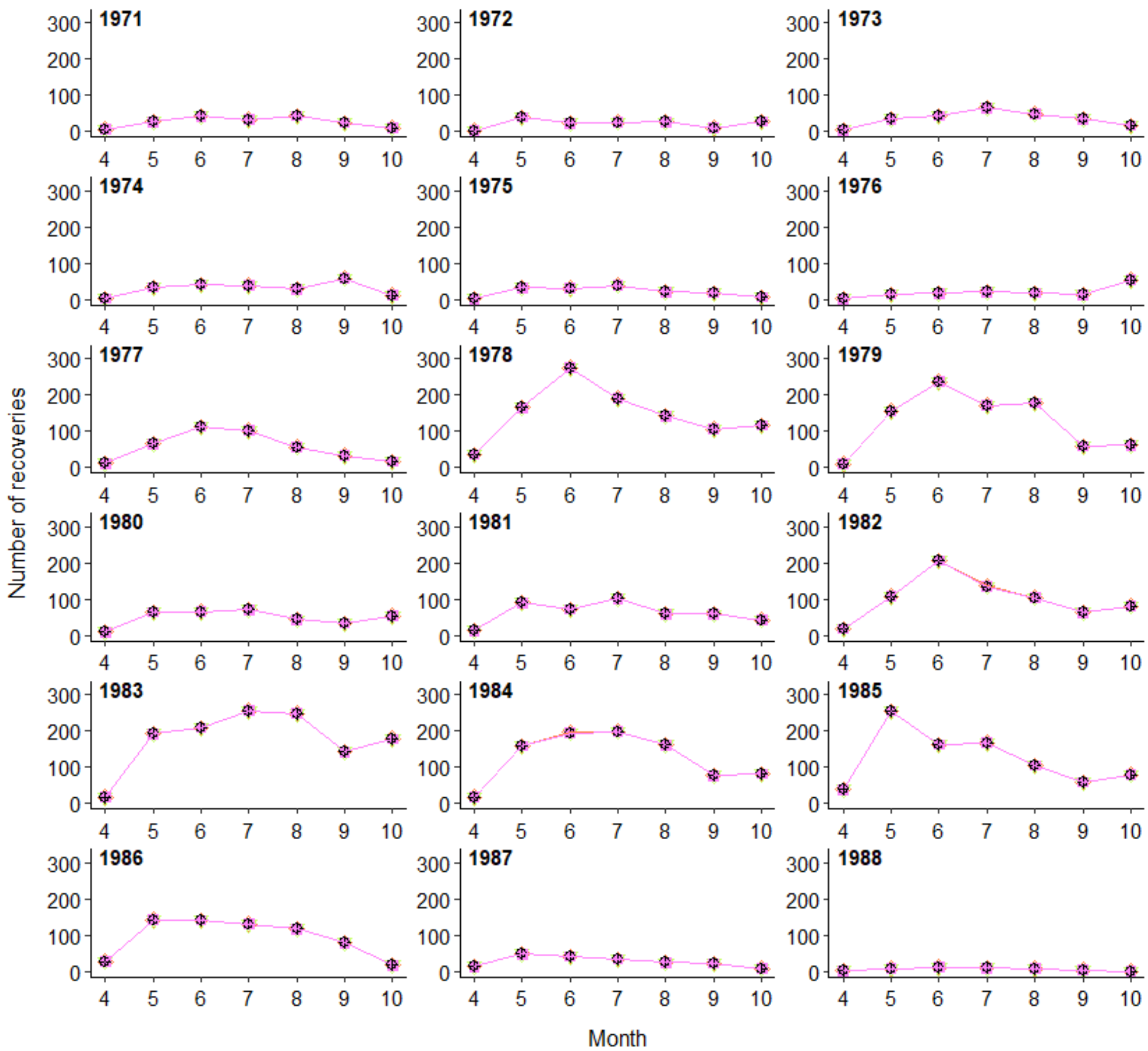
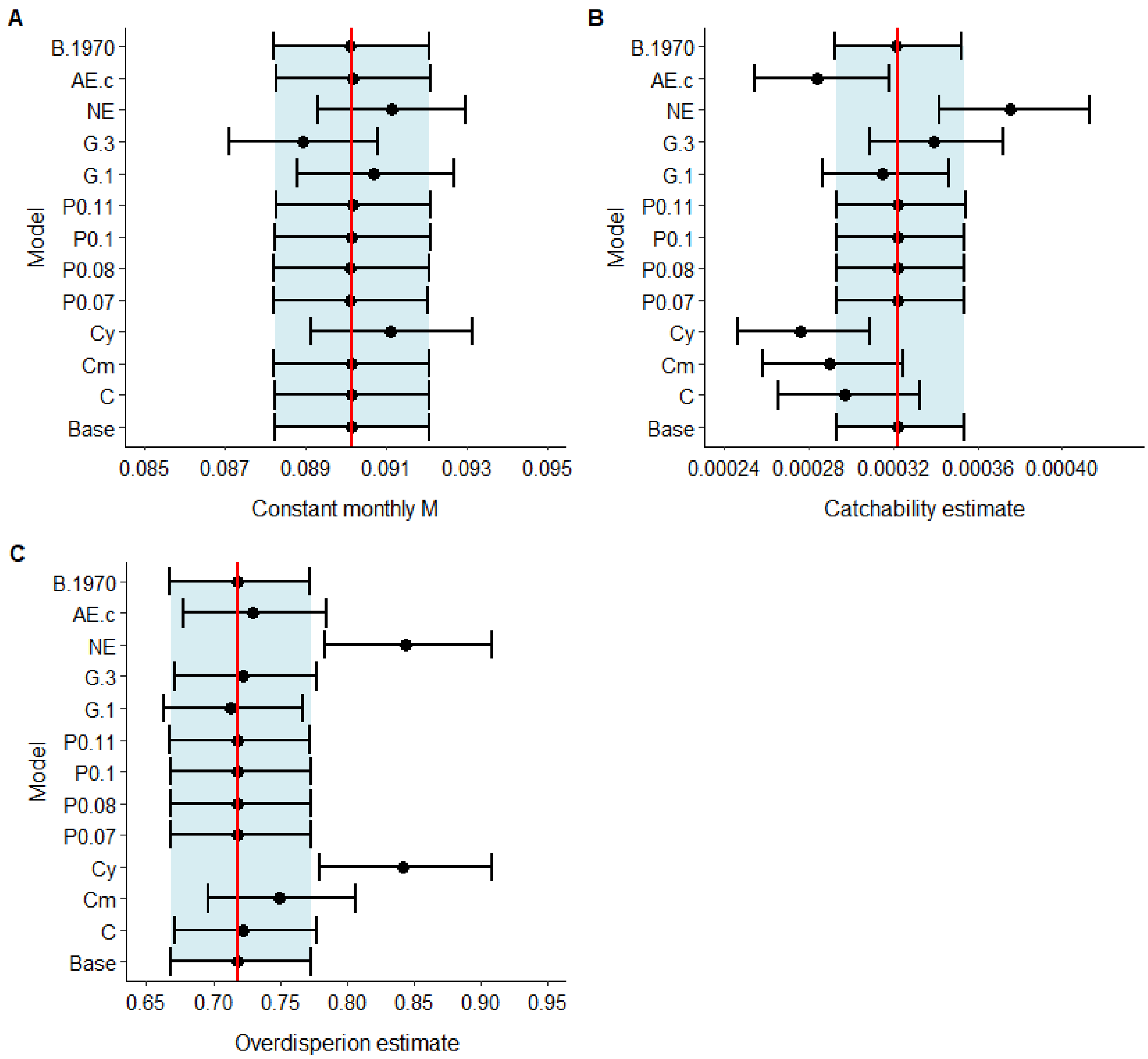


Figure 2.11 *Comparison of All Model Parameter Estimates*

a) Comparison of Base and sensitivity model estimates of monthly instantaneous natural mortality *M*; b) Comparison of Base and sensitivity model estimates of *q*, the constant catchability; c) Comparison of Base and sensitivity model estimates of *k*, the overdispersion factor. Models that did not converge were not included. The red vertical line indicates the Base model estimate.



REFERENCE POINT IMPLEMENTATION FOR THE GULF MENHADEN FISHERY

3.1 Abstract

Regulation and management are needed in many commercial fisheries to maintain a sustainable fish stocks and economic viability. Many fisheries develop reference points to provide indication to fishers and managers about the sustainability levels of their fishery. The Gulf State Marine Fisheries Commission (GSMFC) developed an index-based reference point through an algorithm created and tested in simulation by Dr. Doug Butterworth and Rebecca Rademeyer (Butterworth & Rademeyer, 2019). The approach uses a standardized and weighted time series model to provide estimates of stock status. Their proposed fishery reference point incorporates the indices of relative abundance to form a management target. I developed a web-based application for the use and implementation of this reference point. The online application uses processed fishery independent survey data that has undergone limited quality assurance and control, and creates interactive elements for stakeholders and industry to understand if the current stock status is less than the reference point limit.

3.2 Management and Regulation

Fishery reference points are developed for managed stocks to provide guidance to managers (Quinn & Deriso, 1999; SEDAR63, 2018; Vaughan et al., 2006). Reference points indicate to managers if limits or thresholds are exceeded and thus whether management goals can be attained and if regulatory action is warranted (Quinn & Deriso, 1999; Vaughan et al., 2006). Regulatory action can include limiting fishing effort or limiting harvest. Examples of effort limitation include gear, temporal, and spatial restrictions. Gear restrictions such as regulating minimum mesh size and banned fishing gear (specific types of nets, hooks, etc.) change the selectivity of that gear (Vanderkooy & Smith, 2015). Seasonal fishery closures, trip limits, and area closures reduce fishery effort in space and time. Limitations on harvest, or harvest controls, are another type of regulation that controls the number of fish (or the weight) that can be harvested by the fishery. Such limitations include weight or trip limits and quotas (Quinn & Deriso, 1999).

The Gulf menhaden purse seine fishery has several state-specific effort and harvest regulations. The whole fishery operates during a fishing season, which runs from the third Monday in April to November 1. In the state of Texas, the fishery has a seasonal catch limit in state waters out to nine nautical miles. The current Total Allowable Catch (TAC) in Texas waters was set in 2008 at 14,288.2 metric tons, the five-year average of 2002-2006. All fish nets and traps (excepting purse-seine for Gulf menhaden) are prohibited from use in Texas waters. Purse-seines must be at least ¾ inch square mesh. Gulf menhaden are restricted from harvest within 0.5 miles from the shoreline, with an unofficial agreement to not fish within one mile of the Gulf Beaches. Commercial Gulf menhaden fishing in Louisiana is prohibited inside (inland of) the inside-outside shrimp line (Revised Statutes 56:495) and within 500 ft of the shoreline at Grand Isle from May 1 to September 15, excluding the Chandeleur and Breton Sounds (Title 76 §307). In Mississippi, purse-seine nets are limited at 1,500 ft. Other nets that are used for menhaden, such as gill and trammel, have several restrictions imposed on them. Commercial nets are not allowed within 1,200 ft of public/hotel piers and the Deer Island Shoreline, within 300 ft of ≥ 75 ft private piers, within 1,500 ft of the shoreline between Bayou Caddy in Hancock County and U.S. Highway 90 Bridge, or within 100 ft of the mouths of water sources to marine systems. Gulf menhaden cannot be caught within one mile of the shoreline of Hancock and Harrison Counties. Purse-seines in Alabama must have a minimum mesh size of ¾ inches bar with a maximum length of 2,400 ft, a length which applies to other gill and trammel nets. Entangling nets are required to have at minimum a stretched mesh size of 2.5 inches. Purse-seines for Gulf menhaden are only allowed west of a line defined from South Rigolets to Bayou LaBatre Channel marker “19” and south, not including waters one mile around Dauphin Island. All fishing is prohibited one mile from Gulf Beaches. Florida has banned the use of purse-seine net use for menhaden in state waters since 1995 (Vaughan et al., 2006). Purse-seine nets are restricted from inshore waters, within 3 miles of the shoreline, and in all state waters in Regions 2,3,4 respectively. Florida law also prohibits the use of gill or entangling nets in all waters and restricts all menhaden fishing to the weekdays.

3.3 Commercial Reduction Fishery

The Gulf menhaden reduction fishery is a major purse seine fishery in the Gulf of Mexico and has an annual ex-vessel value of $419.3 million (Murray, 2022). It operates nearshore from April to October every year (Schaaf et al., 1975). The products of the fishery consist of fertilizer and animal feed made from fish meal and refined oil for supplements (Schaaf et al., 1975; Smith, 1991). Fishery harvest peaked during the 1980s, with a maximum of over 980,000 metric tons (Smith, 1991; Vaughan et al., 2006). Since that time, catch has steadily decreased and varies between 600,000 and 300,000 metric tons per year (SEDAR63, 2018). Effort followed the trend of landings, peaking in the 1980s (Figure 3.1). Fishery infrastructure followed a similar expansion and contraction.

The number of reduction plants grew to 14 in the late 1960s, opening in Texas, Louisiana, Mississippi, and Florida. During the 1970s the number of plants held steady at 11, and after 1984 began to decline until today where only 3 plants remain in operation (SEDAR63, 2018; Smith, 1991). Some of this decline may be attributed to the advent of more technologically advanced and larger reduction plants. The same may be said of the number of vessels. The number of vessels greater than 75 net tons overtook those under 75 tons by 1956 (Nicholson, 1978) and the total number of vessels ranged between 65 and 82 from the 1960s to 1980s (Vaughan et al., 2006). The fleet steadily decreased after 1990 and is now stable at 35 to 30 vessels. Landings have been consistent in the last 10 years, oscillating between a range of 360,500 mt to 535,700 mt, with 60% of landings staying in the 400,000 mt scope (SEDAR63, 2018).

3.4 Stock Assessment

To provide information to managers about the fishery and the stock, the interstate resource management agency Gulf States Marine Fisheries Commission (GSMFC) coordinates stock assessment activities. This includes a formal stock assessment every 3 years. The assessment is a collaboration between the GSMFC, the Southeast Data, Assessment and Review (SEDAR), and the individual states' regulatory agencies in the Gulf of Mexico. The process is conducted by NOAA Fisheries' Southeast Region (SEDAR63, 2018). The SEDAR process is a well-publicized, transparent, and open process of the stock assessment to discuss the data, the model, and it includes rigorous external peer review. During the data workshop in the assessment process, data are gathered from fishery-independent and dependent sources. Fishery-independent data for the Gulf menhaden stock are primarily from state surveys and used to develop indices of relative abundance (SEDAR63, 2018). Fishery-dependent data includes the magnitude of harvest from the fishing industry and length, weight, and age data from fish sampled from the catch. These data are used in a forward-projecting age structured model, the Beaufort Assessment Model (BAM), to estimate population metrics for stock status determinations (SEDAR63, 2018; Williams & Shertzer, 2015). The model used to assess the Gulf menhaden stock incorporates an age-specific instantaneous natural mortality rates (M y^{-1}) estimated from a 1970s tagging study (Ahrenholz, 1981; Nelson & Ahrenholz, 1986; SEDAR63, 2018), a time series of landings, age compositions of catch, an adult index and a recruitment index, length compositions for the adult abundance index, and a Beverton-Holt stock curve for recruitment. Selectivity in the model is specified as dome-shaped for the fishery and flat-topped for the adult abundance index (SEDAR63, 2018; Williams & Shertzer, 2015).

3.5 Need for Tactical Management

The Gulf menhaden fishery currently uses two biological reference points, one based on fishing mortality (F) and one based on spawning stock biomass (SSB) in the form of fecundity (SEDAR63, 2018). If F is greater than the F_{limit}, then the stock is experiencing overfishing (Quinn & Deriso, 1999; SEDAR63, 2018). If SSB (or population fecundity FEC) is less than SSB_{limit}, then the stock is considered overfished (SEDAR63, 2018). The limits are meant to be the points at maximum sustainable yield (MSY) of the fishery or a proxy of that value (Quinn & Deriso, 1999). The current limits and targets for the Gulf menhaden stock are $F_{\text{F=M}}$ and $F_{\text{F=0.75M}}$ and $SSB_{\text{25\% at F=0}}$ and $SSB_{\text{50\%}}$

$_{\text{at F=0}}$ (SEDAR63, 2018). Based on the BAM model output, the Gulf menhaden stock is considered not overfished nor is it experiencing overfishing (GDAR03, 2021; SEDAR63, 2018).

The commercial Gulf menhaden fishery seeks to implement a process for annual status determination in part to satisfy the mandates of Marine Stewardship Council (MSC). The MSC is a third-party sustainability certification that provides, to the fishing industry, a certification that the fishery meets certain standards in the governance, regulation, and conduct of the fishery (e.g. limiting bycatch and using non-destructive fishing practices). To meet the requirements of the MSC, the GSMFC worked with stakeholders, the industry, and contracted scientists Butterworth and Rademeyer to develop an index-based fishery reference point as one of the requirements for certification (Butterworth & Rademeyer, 2019). The GSMFC hosted a series of public meetings in the Fall of 2019 to develop the approach. These meetings were attended and mediated by a diverse range of people from different aspects of the fishery and stock, industry liaisons, federal scientists, state scientists and managements, academic scientists, and stakeholders. The reference point that was developed is a statistically based forecasting tool which will allow the commercial fishing industry to understand what constitutes a sustainable level of fishing for the upcoming season and allows management agencies to have near real-time ability for management. The indices will be used as the reference for population health to specify a potential need for total allowable catch (TAC) for years which the threshold index is exceeded. Here I note that the approach for the TAC is still being developed.

The index-based approach provides an annual management target for the fishery to compare current status to, and will provide direct information on how to approach the next season. This approach uses fishery-independent indices of relative abundance. These indices are standardized, weighted, and then used to understand stock status relative to a recognized threshold reference point. The index-based method for this approach uses species catch from scientific surveys to estimate relative abundance of the stock. Data on Gulf menhaden adults and juveniles are collected during these surveys. Mississippi (MS), Alabama (AL), and Louisiana (LA) have state gill net surveys, but only LA's is used for the adult index of abundance due to variations in gear and deployment (SEDAR63, 2018). All 3 states also have state seine surveys, but due to the difference in sampling methods and my lack of MS and AL survey data I did not combine MS and AL surveys with the LA survey during estimation.

All LA survey data used for the Gulf menhaden stock assessment are sampled from fixed stations. Samples collected with seines in specified CSAs (coastal study areas) are measured for total length (TL). Measurements are taken for up to 30 specimens of targeted species and all species collected are identified and counted. Total length measurement of fish is from the tip of the snout to the farthest ends of the caudal fin. Non-target species are weighed together. The LA gill net is fished as a wraparound net survey at fixed stations. All organisms are removed and counted, and TL is recorded. Each sample is weighed per gill net panel. The gill nets are 750 ft long and 8 ft deep monofilament nets with five 150 ft panels. The panels go from 1 to 2 in bar mesh over the five panels increasing ¼ in every panel. Stretch mesh size goes from 2 to 4 in. Large floats are attached to the float line and anchor weights are attached to the lead line. Floats and weights are thrown into the water and the gill net is released from the 1 in bar mesh side over the back of the boat. The juvenile index of abundance is calculated using data from state finfish surveys in LA, MS, and AL (SEDAR63, 2018; Vaughan et al., 2006). Seine data from these stations during the months of January to June are used for the juvenile or recruitment index in the Beaufort Assessment Model. According to the Southeast Data, Assessment, and Review (SEDAR63) Gulf Menhaden Stock Assessment Report in 2018, each state seine survey is performed independently of the other and generally does not target any particular species of finfish or marine organism.

3.6 Objective

There has recently been an influx of management practices that include stakeholders in the regulation decisions (Linke & Jentoft, 2016; Reed, 2008). Inclusion of the communities and stakeholders in management policy is an important part of a well-regulated fishery (Lundquist & Granek, 2005). For several fisheries, this has been implemented through the use of interactive tools and access to the science used by fishery management. Due to common use of and access to the internet and computers, online tools have become substantial in sharing information and providing an easy interactive device. Web-based dashboards have become a frequently used tool in science and fishery management practices as a way to facilitate inclusion and public outreach (Schug et al., 2020). The interactive nature of such applications allows authors to reach a wide audience and more effectively communicate topics. To provide the operationalization of the reference point developed and tested by D. Butterworth in an online application to create an interactive method for users to identify periods of change in stock status. The online application and adjacent program will 1) process sampling data from state agencies, 2) perform some limited quality assurance and control to fix data errors, 3) create two time series from LA fishery-independent data, a seine juvenile index and an adult gillnet index, 4) create a combined adult index of abundance time series, 5) incorporate an interactive element comparing the terminal index of abundance, historic catch, and the current threshold index.

3.7 Methods

I used two sets of data in this study: fishery independent data and LA environmental data. The fishery independent data came from the state of LA gill and seine net complete scientific surveys, with all stations and all taxa. All data and code scripts were set up in a Dropbox folder for storage. The LA gillnet and seine survey data were filtered to only include Gulf menhaden and then split into separate R files (R Core Team, 2023) according to their gear and for error management and data analysis. LA water temperature and salinity were used as covariates in the model. The MS and AL data were not used due to my model set up and lack of MS and AL survey data sets.

I used the Gulf menhaden LA survey data to calculate standardized indices of relative abundance. My model used a generalized linear modeling procedure that combined binomial modeling of presence/absence with the Gamma-distributed positive catch model accounting for relative abundance (Lo et al., 1992). Annual coefficients of variation, standard error, and mean indices for the predicted relative abundance of Gulf menhaden were generated using a jackknife resampling approach. Post-hoc weighted model averaging was performed on the jackknife model predictions, as well as a mean smoothing across the time series. Processed LA gillnet and seine index of abundance data tables were saved to separate csv files in the Dropbox folder. The two index data sets were then combined using the weighted method designed by Butterworth and Rademeyer. Gillnet indices were scaled around 0 and then given a 0.8 weight and seine indices were given a 0.2 weight (Butterworth & Rademeyer, 2019). This combined data set was also saved as a csv file in Dropbox. The index of the year after the highest catch during the years 1988 to the present year 2023 is currently used as the threshold index. I expect the indices in the time series to change slightly every year as new data is added to the model.

I created a dashboard using the Rshiny package (v1.7.2; Chang et al., 2022) to display the adult index data with interactive figures. I wrote a global script to run in the background of the dashboard script. In the global script, all 3 index data sets are automatically downloaded to the dashboard from Dropbox storage and are then used to compute the probability of a determined threshold index overlapping with the recent terminal index. To compute the probability, I created a sequence between the minimum and maximum combined index value with 0.01 intervals. This sequence was placed in a data table with temporary upper and lower bounds that were filled in with combined index data bounds where values matched, and were given rolling averages where there were no data bounds from the historic indices. Each index was then distributed normally around its value with a standard deviation calculated by subtracting the index value from its temporary upper bound. The 95% confidence interval (CI) values were taken from each distribution and recorded as the upper and lower CI bounds for the sequence of index values. I then took 100 random samples using the function RNORM from a normal distribution with mean value equal to the terminal index value and the difference between the terminal index and its upper bound as the standard deviation. I calculated the overlapping probability as the number of the terminal normal distribution values that were greater than or equal to the random sample from the normal distribution of each individual index in the sequence, divided by the number of values in the normal distribution. These probabilities were smoothed with a rolling average.

3.8 Results

I estimated the 2022 terminal index to be 0.97 with 95% confidence intervals of 0.53 and 1.41. These intervals change slightly in the dashboard figures due to the use of the RNORM function in R to randomly sample the terminal index for the probability curve estimates

The dashboard itself contains 4 tabs 1) an About tab, 2) a Data Information tab, 3) an Index Figures tab, and 4) a Simulations tab. The About and Data Information Tabs contain information concerning the project and the data formatting procedure, with examples of the data that are used in the analysis. The Index Figures tab displays the graphs of the current combined adult index of abundance time series (Figure 3.2c), both the Louisiana gillnet and seine time series of indices (Figure 3.2a and b respectively), and a comparison of catch and the combined index of abundance (Figure 3.3). The Simulations tab has three interactive figures. All three are controlled by a slider that allows the user to pick any threshold index value between -1.5 and 2.5 with 0.01 intervals. The slider input controls a red bar in each figure. The first figure is the graph of the combined index of abundance time series (similar to Figure 3.2c), with a red bar indicating the value of the threshold index to visually compare the past and current terminal index. The second figure is the curve of the probabilities calculated in the global script (Figure 3.4). The red bar intercepts with the probability that the input threshold index overlaps with the CI of the terminal index. The final figure compares the total yearly fishery catch in 1000 metric tons and the indices of abundance (Figure 3.3), the red bar again falling on the threshold index. The current threshold index used by the fishery is the index of the year after the highest catch after 1988, which is presently -1.03. I also included a table that provides the current terminal index with its CI, the user chosen threshold index with its CI, and the probability of overlap (Table 3.1). The CI bounds change slightly every time the global section is rerun due to the random sampling of each normal distribution.

Total fishery catch does not seem to have a significant positive or negative relationship with index of abundance. However, there were more occurrences of years of high catch also exhibiting low indices than years of high catch exhibiting high index values. After 2008, all years except 2010 have greater indices than 0, while 1988 to 2008 have indices lower than 0 (Figure 3.3).

3.9 Discussion

My scaled combined adult index of abundance has a different scale than the stock assessment estimated indices (GDAR03, 2021; SEDAR63, 2018) due to the use of different modeling approaches, different scaling techniques, and the addition of more data since 2018. My adult index of abundance used only LA data like the stock assessment, but I did not include the survey data from MS or AL for the juvenile index of abundance since I did not have it. The LA seine data had the longest running data set and several stations while MS only had 2 long term sampling stations and AL data is only available from 2001 onwards (SEDAR63, 2018). This makes LA the predominant data set used by the stock assessment, so I consider it sufficient to only use LA seine survey data for my index calculations. The seine index of abundance was weighted down in the combined standardized index of relative abundance, accounting for only 20% of the combined index, so it had a lower effect on the combined index trend than the LA gillnet index. However, by not including the other state surveys, I run the risk of not fully accounting for the entire spatial distribution of Gulf menhaden.

My seine index of abundance trend was comparable to the stock assessment trend, though with different index values. My seine index time series (Figure 3.2b) exhibited large juvenile year classes in the same years as the assessment (1996, 2010, 2011, 2014, 2016) (SEDAR63, 2018) but the new data for the following years also exhibited a large year class in 2018, 2020, and 2022. My gillnet index of abundance trend was also comparable to the trends in the assessment gillnet index. Both index time series show a slight increasing trend from the 1990s, a stable oscillation in the early 2000s, and a high abundance from 2008 onwards with 2010 as an exception. This dip in abundance in 2010 is likely due to the Deepwater Horizon oil spill that occurred April 20 as adults were moving nearshore. The adult index of abundance reflects the dip in 2010, while the juvenile index does not (Figure 3.2a and b). It is suggested that juvenile menhaden, already in estuaries from the previous winter spawning, moved further upstream to avoid the spill and experienced less predation than usual, as their predators were caught in the oil spill (Schaefer et al., 2016; Short et al., 2017). This accounts for the large increase in

Gulf menhaden juvenile recruitment in 2010. However, since the standardized index of abundance gives more weight to the adult index, this large juvenile abundance does not come through.

My terminal estimate for 2022, 0.97, is above the mean index value average of 0.9 for the last 15 years, but below the mean index value average, 1.01, of the last 15 years sans 2010. The 2022 terminal index is the 5th highest index in the last 10 years, and fishery landings close to the average for the last 10 years.

I performed a Pearson's correlation test between total fishery landings and the index time series, correlating the yearly index with the next season's landings. I estimated a -0.35 correlation value with 95% CI of -0.615 to -0.014 and a pvalue of 0.04239. My significance level was 0.05. The wide confidence interval indicates that the relationship between total fishery catch and index value is uncertain in its magnitude, though it is negative. From this I can say it is uncertain if higher levels of catch will have a strong negative impact on the index of abundance estimation, or will cause the fishery to fall under the threshold index. The stock assessment found correlation between age-2 catch and the adult index of abundance (SEDAR63, 2018), but since I did not have the age based landings data I was unable to test this correlation. The higher correlation between year and index suggests that other temporal factors play a larger role in adult Gulf menhaden abundance. The past 15 years since 2008, except 2010, have much higher indices than the early 2000s and 1990s. This is an indicator of a higher abundance of Gulf menhaden in recent years. This recent high relative abundance could be a result of changing environmental factors due to anthropogenic effects and climate change. Average temperatures in the Gulf of Mexico have been increasing due to oceanic warming (Z. Wang et al., 2023), and it has been shown that juvenile gulf menhaden grow faster in a range of higher temperatures (Adams et al., 2018; Deegan, 1990). This high relative abundance could also be a result of cumulative years of stable lower fishing after 1984 (year of highest catch).

I operationalized the fishery reference point and index of abundance for use by agencies and stakeholders. I did not assess performance of the harvest control rule that uses these indices and the fishery reference point since that was completed by Dr. Butterworth and Rademeyer (Butterworth & Rademeyer, 2019). My process will allow stakeholders to understand the effect of the new reference point on catch prior to the fishing season (equal to or below the reference point threshold) and to interact with and improve their understanding of the reference point. My dashboard is a useful tool to industry and regulators through its interactive nature and annual updates of the abundance. The dashboard provides a way for the stakeholders to change the threshold index themselves and see how close the terminal or historic indices interacts with the chosen threshold index. The probability of overlap calculated in my dashboard gives the stakeholders and industry a value to use when discussing the risks of different threshold index values to be used for management.

3.10 Tables and Figures

Table 3.1 *Current Index Values*

Values of current threshold index with its 95% confidence intervals and the current terminal index and its confidence intervals. The table also includes the probability that the threshold index overlaps with the intervals of the terminal index.

Threshold Index	Lower	Upper	Terminal Index	Lower Terminal	Upper Terminal	Probability (%)
-1.03	-1.24	-0.74	0.97	0.53	1.41	0

Figure 3.1 *Gulf Menhaden Fishing Effort*

Historic Gulf menhaden fishery landings in 1,000 metric tons and catch per unit effort in 1,000 vessel-ton-weeks.

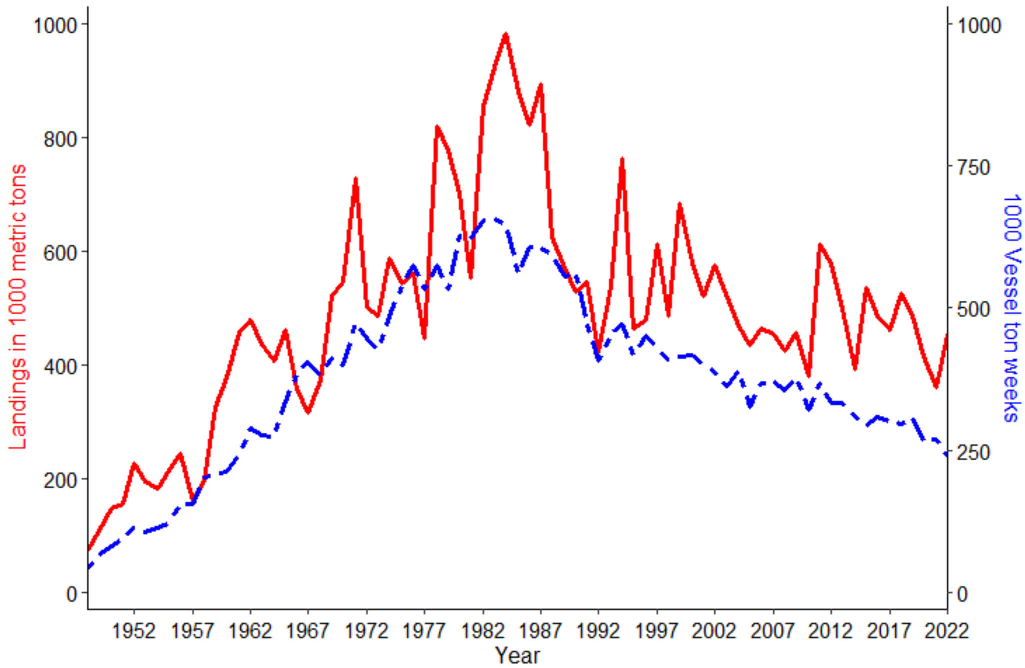


Figure 3.2 *Time Series of Adult Index of Relative Abundance*

a) Calculated time series of adult index of abundance from the Louisiana gillnet survey, b) calculated time series of adult index of abundance from the Louisiana seine survey, c) calculated time series of adult index of abundance from combined gillnet and seine indices. The red point indicates the terminal index.

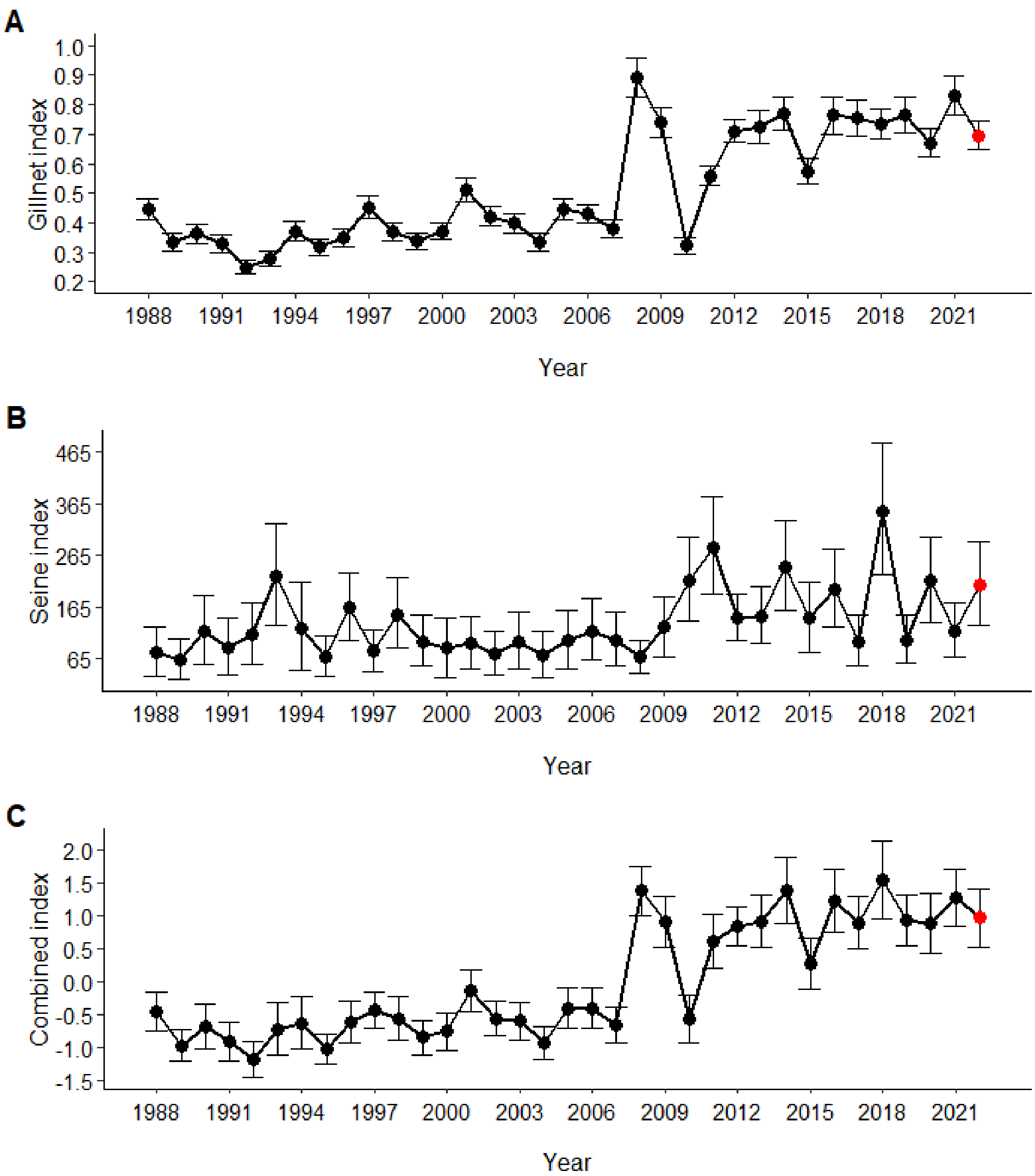


Figure 3.3 *Catch Distributed Over Index of Relative Abundance*

Total yearly Gulf menhaden commercial landings in 1,000 metric tons and their respective index. Data points are colored by their year, with lighter colors pertaining to recent years. The red bar marks the current threshold index.

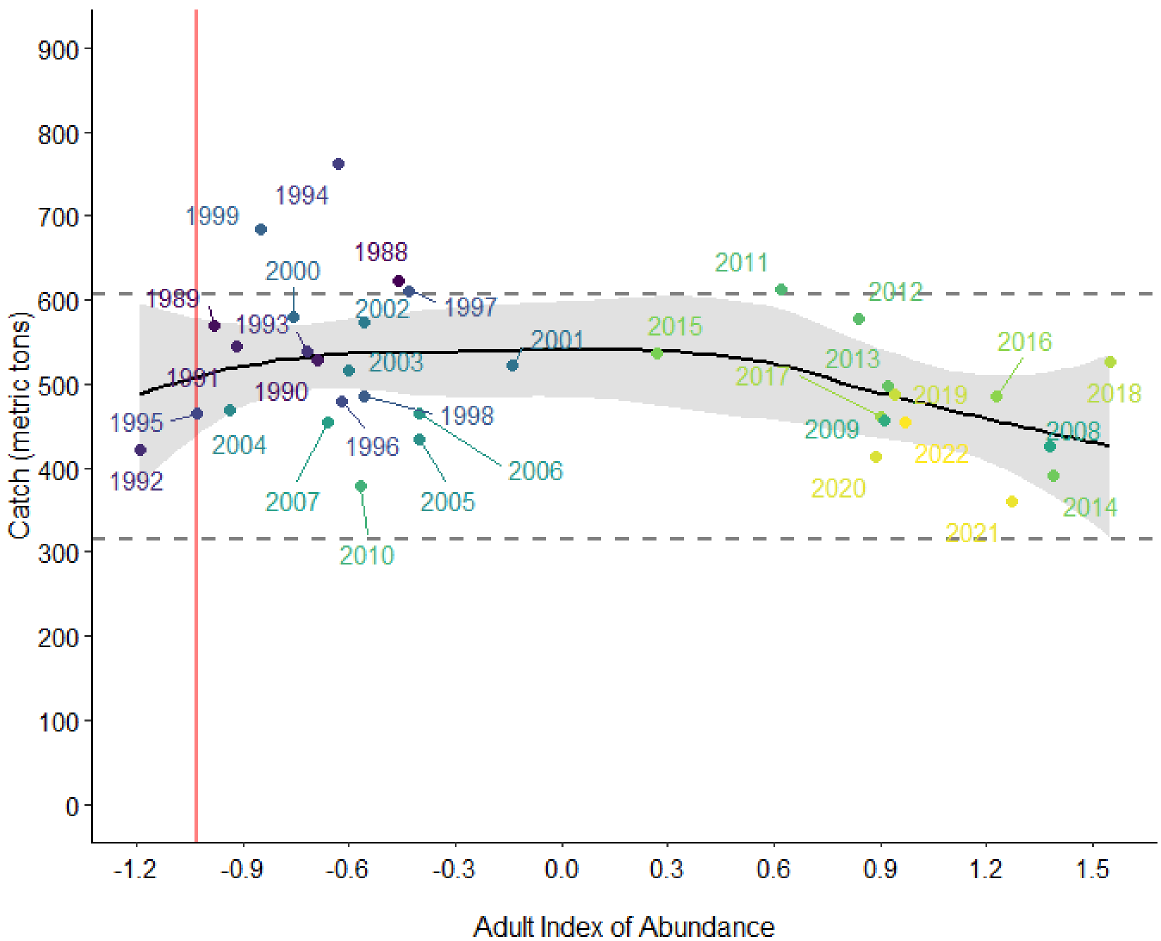


Figure 3.4 *Probability Curve of Overlap with Terminal Index*

Sequence of the probability of different threshold indices overlapping with the margin of error of the terminal index. The red bar marks the current threshold index. The dashed lines indicate the approximate start and end of the curve.

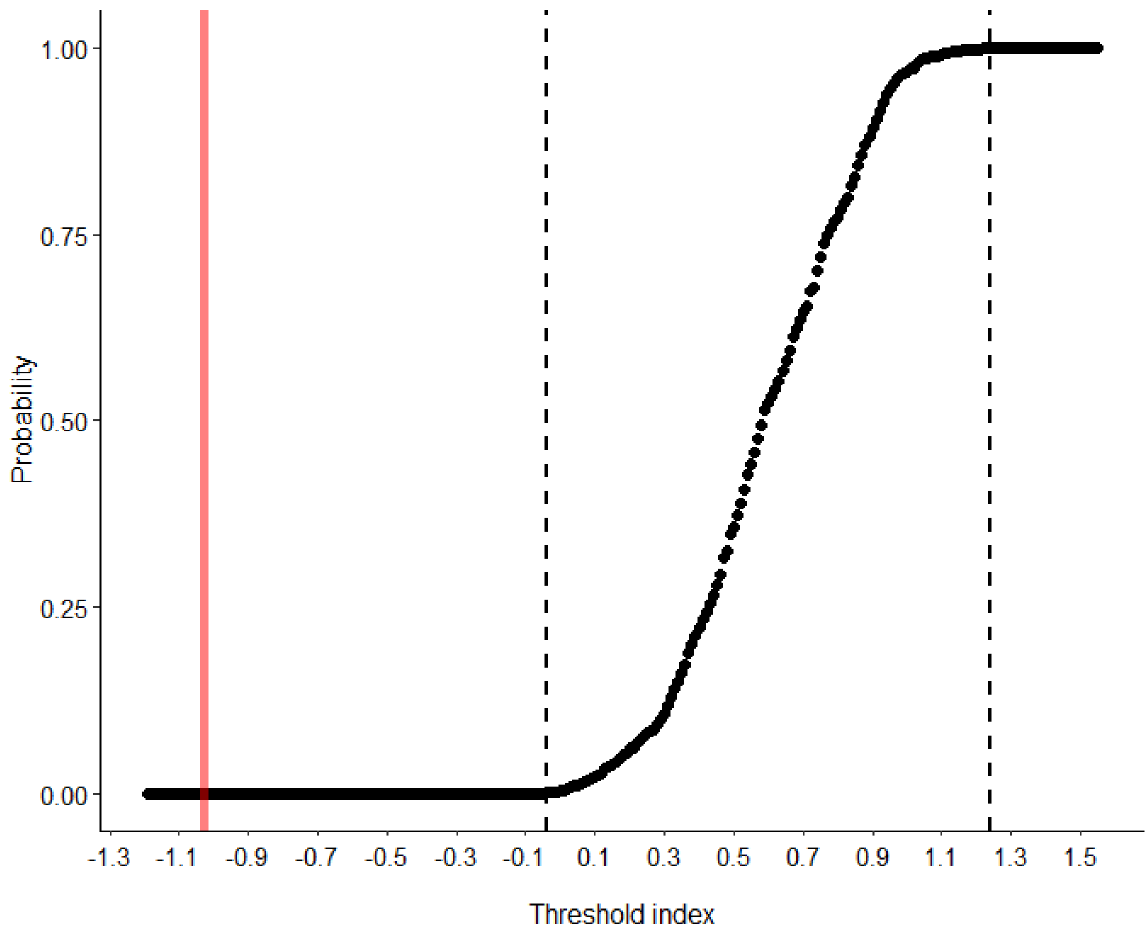
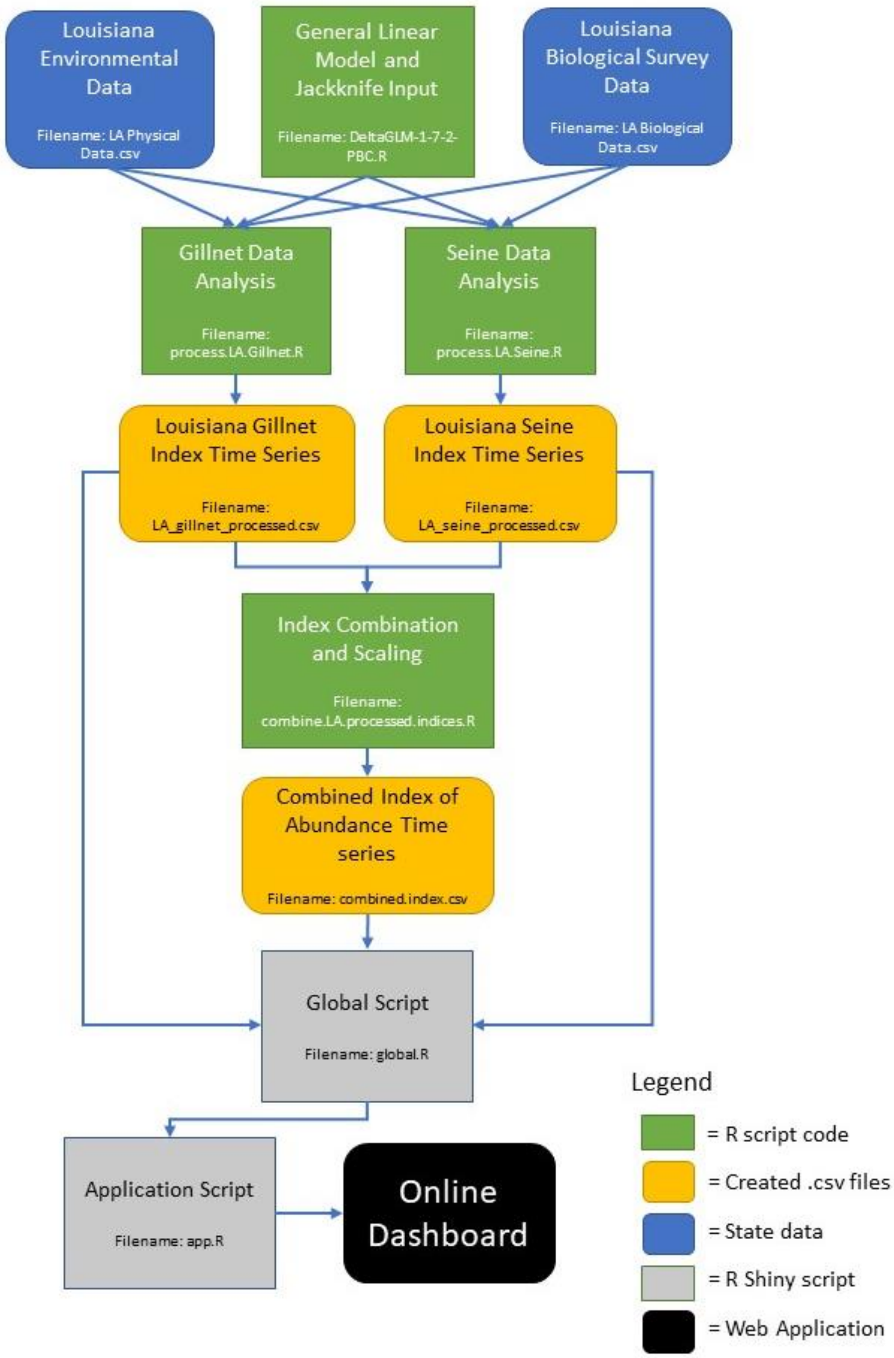


Figure 3.5 *Flow Chart of Proceedings*

Flow chart of total dashboard process. File names of all data sets and code scripts are given. Green boxes are R scripts, yellow boxes are created data files, blue boxes are state datasets, grey boxes are R shiny scripts, and the black box represents the dashboard tool.



APPENDIX

Table A.1 *Total Gulf Menhaden Tags Released*

Total number of Gulf menhaden tagged and released by Ahrenholz from 1970 to 1985.

		Year															
Stage	**Month**	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985
Juvenile	July	0	2798	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	August	0	3896	0	4400	10797	0	0	0	0	0	0	0	0	0	0	0
	September	3998	0	15696	0	0	0	0	0	0	0	0	0	0	0	0	0
	October	600	0	0	0	0	0	7689	15287	9889	2000	10190	11293	8498	7894	13286	8198
	November	5688	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Adult	July	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	August	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	September	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	October	0	0	0	0	0	0	11979	8084	8770	2100	6294	6495	13197	8895	8493	15487
	November	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table A.2 *Constant Natural Mortality Point Estimates and Error*

Point estimates, 1 standard deviation confidence intervals and the standard deviations in the natural log scale of the constant instantaneous monthly natural mortality *M* for all model runs.

Model	M_c	**Upper 68.2% CI**	**Lower 68.2% CI**	**Standard deviation in log scale**
Base	0.09013	0.092	0.088	0.021
C	0.09012	0.092	0.088	0.021
C_y	0.09112	0.093	0.089	0.022
C_m	0.09012	0.092	0.088	0.021
P0.07	0.09010	0.092	0.088	0.021
P0.08	0.09011	0.092	0.088	0.021
P0.1	0.09015	0.092	0.088	0.021
P0.11	0.09017	0.092	0.088	0.021
G.1	0.0907	0.093	0.089	0.022
G.3	0.08892	0.091	0.087	0.021
B.1970	0.09011	0.092	0.088	0.021
NE	0.09112	0.093	0.089	0.02
AE.c	0.09016	0.092	0.088	0.021

Table A.3 *Constant Catchability Point Estimates and Error*

Point estimates, 1 standard deviation confidence intervals and the standard deviations in the natural log scale of the constant monthly catchability *q* for all model runs.

Model	***q***	**Upper 68.2% CI**	**Lower 68.2% CI**	**Standard deviation in log scale**
Base	0.000322	0.00035	0.00029	0.094
C	0.000297	0.00033	0.00027	0.114
C_y	0.000276	0.00031	0.00025	0.113
C_m	0.000289	0.00032	0.00026	0.114
P0.07	0.000322	0.00035	0.00029	0.0941
P0.08	0.000322	0.00035	0.00029	0.0941
P0.1	0.000322	0.00035	0.00029	0.0941
P0.11	0.000322	0.00035	0.00029	0.0940
G.1	0.000315	0.00035	0.00029	0.095
G.3	0.000339	0.00037	0.00031	0.093
B.1970	0.000321	0.00035	0.00029	0.094
NE	0.000376	0.00041	0.00034	0.094
AE.c	0.000284	0.00032	0.00025	0.112

Table A.4 *Overdispersion Factor Point Estimates and Error*

Point estimates, 1 standard deviation confidence intervals and the standard deviations in the natural log scale of the overdispersion factor *k* for all model runs.

Model	***k***	**Upper 68.2% CI**	**Lower 68.2% CI**	**Standard deviation in log scale**
Base	0.718426	0.773	0.668	0.073
C	0.722272	0.777	0.671	0.073
C_y	0.841651	0.909	0.779	0.077
C_m	0.749364	0.807	0.696	0.074
P0.07	0.718512	0.773	0.668	0.073
P0.08	0.718469	0.773	0.668	0.073
P0.1	0.718383	0.773	0.668	0.073
P0.11	0.718339	0.773	0.668	0.073
G.1	0.713373	0.767	0.663	0.073
G.3	0.72226	0.777	0.671	0.073
B.1970	0.71775	0.772	0.667	0.073
NE	0.843723	0.909	0.783	0.074
AE.c	0.729287	0.785	0.677	0.074

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