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The economic importance and management of mercury contamination in pelagic fisheries

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THE ECONOMIC IMPORTANCE AND MANAGEMENT OF MERCURY
CONTAMINATION IN PELAGIC FISHERIES

A Dissertation
Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements for the degree of
Doctor of Philosophy

in

The Department of Agricultural Economics and Agribusiness

by
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For Nate

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ABSTRACT

The presence of mercury and other contaminants in the U.S. fish supply is a growing public health concern. At high levels, these substances can be harmful to humans and ecosystems, and thus represent a growing threat not only to public health, but also to the economic and ecological viability of many fisheries. This research examined the economic issues surrounding mercury contamination in fish, developed a population dynamics and bioeconomic model to investigate the problem, and compared a variety of management actions to reduce consumer exposure to contaminants.

The dissertation begins with an overview of contamination issues in U.S. fisheries, including a review of the historical public health impacts and their related economic costs. Management strategies for dealing with fish and shellfish contamination were discussed, along with an examination of the efficacy of these actions and their implied economic cost to the fishing industry.

Given that mercury concentration is shown to increase with fish length, this study continues by examining the implications of harvesting smaller (and less contaminated) fish. This was accomplished through the development and application of an age-structured bioeconomic model for king mackerel, a species experiencing particularly large concentrations of mercury contamination. First, a population dynamics model of the Gulf of Mexico and Atlantic king mackerel stocks was constructed and validated. Using the population dynamics model as a base, a comprehensive bioeconomic model was created through the incorporation of the economic characteristics of the fishery and mercury contamination relationships. Forward simulations were used to examine the plausibility of different management alternatives for the king mackerel stocks in the presence of mercury contamination. The simulations demonstrated the possibility of reducing the amount of mercury that reaches consumers by altering the age composition of the

commercially marketed catch. Furthermore, the simulations illustrated that it may be possible for this to occur without seriously impacting the long-run stability of the stock. There are tradeoffs, however, in terms of the economic viability of the fishery. In the case of both the Atlantic and Gulf stocks, reductions in mercury came at the price of reduced fishery profits and losses in the aggregate net present value of the fishery over a 25 year time horizon.

CHAPTER 1: INTRODUCTION

High in protein, low in saturated fat, and containing Omega-3 fatty acids, fish and shellfish are important components of a healthy human diet (U.S. EPA 2004b). In particular, evidence suggests that Omega-3 fatty acids protect against coronary heart disease and stroke, and also aid in the neurological development of fetuses (McMichael and Butler 2005). These benefits, however, need to be weighed against the potential health risks associated with the consumption of fish and shellfish, including contamination from harmful algal blooms (HABs). Heavy metals like mercury, organic pollutants like polychlorinated biphenyls (PCBs) and dioxins, and pesticides such as chlordane and dichlorodiphenyl-trichloroethane (DDT) also pose health risks for seafood consumers. These latter contaminants have been labeled by the U.S. Environmental Protection Agency (EPA) as persistent, bioaccumulative, and toxic (PBT). PBT substances can build up in the food chain to levels that are harmful to humans and ecosystems (U.S. EPA 2004a), and thus represent a growing threat not only to public health, but also to the economic and ecological viability of many fisheries. In addition to increased public health costs, losses to commercial fisheries, reduced recreation and tourism, and increased monitoring and management costs, the negative publicity and public awareness concerning fish contamination can also impact economic viability of fisheries through reduced demand, not only for the contaminated species but also for other fish products due to the consumer's tendency to interpret advisories and warnings as applying to broad food groups (if they pay attention to them at all) (Shimshack et al. 2004). Within this context, there is a need to examine the options available to public health and fisheries managers, not only to protect the public health from contaminated seafood, but also to continue to provide both the public and private benefits that are generated from the fishing industry.

BACKGROUND AND MOTIVATION

The widely publicized mercury poisoning incident in Japan's Minamata Bay, and the resulting health impacts, ignited a public interest in the consumption of contaminated fish. Mercury is a persistent metal that is distributed throughout the environment and originates from both natural sources and human activities. Its organic form, methylmercury, accumulates in the tissues of fish and, once ingested, can cause irreversible human health effects (U.S. EPA 2001). Mercury has been found in many fish species throughout world, and dietary intake through fish consumption is the dominant source of mercury exposure for the general population. Fish consumption has been linked to elevated mercury levels in humans (Bjornberg et al. 2003; Schober et al. 2003). The human nervous system is very sensitive to all forms of mercury, and exposure to high levels of methylmercury can permanently damage the brain, kidneys, and developing fetus (ATSDR 1999).

The deleterious health impacts that may result from mercury exposure have led to considerable efforts to reduce the levels that reach the population. These efforts have focused primarily on the issuance of consumption advisories and on long-term pollution reduction. Consumption advisories are recommendations for voluntary action, informing the public that excessive concentrations of chemical contaminants have been found in local fish. These advisories may include recommendations to limit or avoid eating certain fish species or fish caught in specific water bodies. An advisory may be issued for the general population or for sensitive subpopulations such as pregnant women, nursing mothers, and children (U.S. EPA 2005c). Consumption advisories are only successful in reducing exposure if consumers are aware of the advisory and respond in the appropriate manner. However, consumer response to advisories is often unpredictable.

While almost all fish contain traces of methylmercury, larger fish that have lived longer have the highest levels due to the persistent and bioaccumulative nature of this contaminant (U.S. EPA 2004b). The 2004 joint federal advisory issued by the U.S. EPA and the Food and Drug Administration (FDA) advises pregnant women, women who may become pregnant, nursing mothers, and young children to avoid consumption of shark, swordfish, tilefish and king mackerel and limit albacore tuna consumption due to high mercury levels. Not coincidentally, these are all large predatory fish. Recent studies have examined the relationship between fish size and mercury concentration in a variety of species from various waterbodies and found a significant positive relationship. Examples include king and spanish Mackerel in the Atlantic and Gulf of Mexico (Adams and McMichael, 2007), swordfish and bluefin tuna from the Mediterranean Sea (Storelli and Marcotrigiano, 2001), tunas from offshore waters of the Florida Atlantic Coast (Adams 2004), swordfish, yellowfin and skipjack tuna, wahoo, and dolphinfish in the Indian Ocean (Kojadinovic et al. 2006), and various commercially important species in Japan, including bluefin tuna (Yamashita, Omura, Okazaki 2005).

At the same time, the U.S. Environmental Protection Agency (EPA) has made it a priority to reduce risks to human health and the environment from existing and future exposure to priority pollutants, such as mercury (U.S. EPA 2004a). The EPA has taken considerable action to reduce mercury pollution, including issuing stringent regulations for industries that contribute to U.S. mercury emissions. While the aim is to significantly reduce the new deposition of mercury into the environment, its persistence makes it likely that mercury will remain in the nation's fish stocks indefinitely, even as emissions are greatly reduced. As an example of the difficulties faced by regulators, consider the prominent case of mercury. Attempts to limit exposure to mercury through normal regulatory emissions controls is confounded by uncertainty concerning the relative importance of anthropogenic versus natural sources and the lack of

quantitative estimates of the relationship between mercury deposition and mercury concentrations in fish (U.S. EPA 1997). This latter point is highlighted by the fact that, although U.S. mercury emissions have been greatly reduced since 1990, levels of methylmercury in seafood have not changed substantially over recent decades. Since the available evidence suggests that even deep cuts in domestic mercury emissions are unlikely to bring benefits to public health or ecosystems (Lutter and Irwin 2002), alternative approaches may be needed in order to reduce the public's long-term exposure beyond that achieved through voluntary responses to health advisories. Health advisories themselves are problematic in that an advisory can only be effective if consumers are aware of it and are willing and able to translate that awareness into behavior. For example, Shimshack et al. (2004) examined response to the 2001 FDA methylmercury fish advisory and found that a large group of at-risk consumers (infants, small children, pregnant or nursing mothers, and women who may become pregnant) did not respond to the advisory, particularly in the case of less educated and less informed consumers. Alternative approaches may be needed in order to reduce the public's long-term exposure beyond that achieved through voluntary responses to consumption advisories. One potential alternative that has not yet been considered is to reexamine the way size-based fisheries management is conducted.

As currently implemented, most management plans focus on supporting recruitment to the fish stocks and survival to reproductive age by imposing minimum size limits on captured fish. The bioaccumulative property of many contaminants often results in a positive relationship between fish size and the levels of contaminant concentration, thus paradoxically leading to a situation where management plans designed to protect stocks for ecosystem purposes and for future human use actually increase the levels of contaminant exposure experienced by consumers. At the current time, no pre-harvest methods are used to control the amount of

contaminants that reach fish consumers. While a complete ban on the harvesting of a contaminated species is conceptually possible (although unlikely), an alternative might be a more directed, size-based management scheme that explicitly accounts for the economic and public health dynamics of harvesting in the presence of contamination. Intuitively, this approach might require the harvesting of younger, smaller fish with the goal of allowing older, larger fish to serve as both a breeding stock and contaminant sink. The development and analysis of an empirical bioeconomic model can be used to investigate these issues, in the process combining the complex sets of population and toxicology information necessary for analyzing the relevant economic tradeoffs.

RESEARCH OBJECTIVES

The goals of this research can be captured in four distinct objectives:

1. Summarize the major historical feature of contaminated fisheries, focusing not only on the public health and economic implications of contamination, but also on the various approaches that have attempted to manage contamination and its private and public effects;
2. Using the historical summary as motivation, develop a realistic, multiple-cohort population dynamics model for a pelagic fish species¹ that can be used in an investigation of contaminant management scenarios;
3. Incorporate the population dynamics model into a broader bioeconomic model that not only accounts for the harvesting sector and its economic structure, but also specific contaminants and their potential exposure to human consumers; and
4. Apply the bioeconomic model to the investigation of fishery management scenarios that might have the potential for mitigating the deleterious effects of contamination on humans

¹ Although not true for all pelagic species, many are predatory and near the apex of the marine food web, thereby being exposed to the bioaccumulative effects associated with a number of both man-made and natural contaminants.

while, at the same time, preserving the public and private benefits associated with the fishing industry.

The first major objective of this research is to present an overview of contamination issues in U.S. fisheries by summarizing the important historical evolution of the contamination problems, including a review of the public health impacts and the related economic costs. Management strategies for dealing with fish and shellfish contamination will be described, along with an examination of the apparent efficacy of these actions and their implied economic cost to the fishing industry. Building on this context, the second major objective of this study is to develop a theoretically sound population dynamics model of a fish species that can be used to investigate the potential for reducing human exposure to specific contaminants. The pelagic species king mackerel (*Scomberomorus cavalla*) was chosen for parameterizing the model because it is relatively well studied from a biological perspective, is currently undergoing management revisions at the federal level, and is contaminated by the heavy metal mercury. A multiple cohort approach was used in this study because the expected level of mercury contamination in king mackerel is size, and therefore age, dependent, requiring the tracking through time of each recruited class. Once developed, the multiple cohort population dynamics model was augmented with the necessary economic, contamination, and exposure relationships so as to provide a comprehensive bioeconomic framework from which to analyze various potential management approaches (objective 3). A comparison of the different scenarios is then used to generate policy relevant management suggestions (objective 4).

ORGANIZATION OF THE DISSERTATION

The main body of this dissertation contains three chapters that illustrate the economic issues of fish contamination ranging from economic costs, current management, and proposed management to reduce consumer exposure to contaminants. The first, Chapter 2, highlights the

public health and economic implications of contaminated U.S. fisheries, with a focus on harmful algal blooms (HABs) and several primary persistent, bioaccumulative, and toxic (PBT) contaminants. Chapter 3 develops a multiple cohort population dynamics model for the U.S. king mackerel fishery, validating its use against information collected and analyzed during the recent king mackerel SouthEast Data, Assessment, and Review (SEDAR), a cooperative Fishery Management Council process designed to improve the quality and reliability of fishery stock assessments in the South Atlantic, Gulf of Mexico, and U.S. Caribbean. Chapter 4 expands on this biological model by incorporating economic characteristics of the fishery, contamination relationships, and exposure assumptions. In addition, it examines potential management alternatives for the king mackerel fishery in the presence of mercury contamination. A summary of the entire dissertation topic is then presented in Chapter 5, highlighting the conclusions from the three previous chapters and discussing future directions for the research.

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CHAPTER 2: THE PUBLIC HEALTH AND ECONOMIC IMPACTS OF CONTAMINATED U.S. FISHERIES²

INTRODUCTION

High in protein, low in saturated fat, and containing Omega-3 fatty acids, fish and shellfish are important components of a healthy human diet (U.S. EPA 2004b). In particular, evidence suggests that Omega-3 fatty acids protect against coronary heart disease and stroke, and also aid in the neurological development of fetuses (McMichael and Butler 2005). These benefits, however, need to be weighed against the potential health risks associated with the consumption of fish and shellfish, including contamination from harmful algal blooms (HABs). Heavy metals like mercury, organic pollutants like polychlorinated biphenyls (PCBs) and dioxins, and pesticides such as chlordane and dichlorodiphenyl-trichloroethane (DDT) also pose health risks for seafood consumers. These latter contaminants have been labeled by the U.S. Environmental Protection Agency (EPA) as persistent, bioaccumulative, and toxic (PBT). PBT substances can build up in the food chain to levels that are harmful to humans and ecosystems (U.S. EPA 2004a), and thus represent a growing threat not only to public health, but also to the economic and ecological viability of many fisheries.

This article highlights the public health and economic implications of contaminated U.S. fisheries, with a focus on HABs and several primary PBT contaminants. A brief history of contamination problems in fisheries will be presented, including a review of the public health impacts and the related economic costs. Current monitoring, notification, and management strategies for dealing with fish and shellfish contamination will be described, along with an

² Reprinted by permission of BRILL Academic (See Appendix A). The paper originally appeared under the same title in 'Ocean Yearbook 21' edited by Aldo Chircop, Scott Coffen-Smout, Moira McConnell. Leiden: Martinus Nijhoff Publishers, 2007, pp. 307-337. ISBN 9789004157552

examination of the efficacy of these actions and their implied economic cost to the fishing industry. The article will conclude with a discussion of the future of commercial fisheries in the presence of contamination, including some suggestions for alternative strategies that have the potential for reducing public exposure to contaminants and improving the management of wild fish stocks.

CONTAMINATION IN FISHERIES

The contamination of wild fish stocks, and the ensuing public health problems, is a worldwide phenomenon that began as far back as 800 B.C. (Tyson et al. 2004). In modern times, the first widely publicized case involved the PBT methylmercury and the associated poisonings that resulted from industrial discharges into southern Japan's Minamata Bay. Contamination of the Bay itself began in the 1930s; the first instances of human poisoning were not reported until the 1950s when mothers who had experienced a lifetime of exposure to contaminated seafood gave birth to deformed babies (Powell 1991). As of early 2005, Japan had officially recognized 2,955 poisoning victims, of which 1,784 had already died (although not necessarily directly due to methylmercury poisoning; Dateline Tokyo 2005). An additional 15,000 individuals have registered as victims of the contamination, highlighting the long-term nature of public health and economic impacts when contamination problems do not manifest themselves immediately after exposure (Grinel 2001).

In addition to toxic chemicals, seafood consumers face exposure to pathogens and naturally occurring toxins. Human and industrial wastewater discharged into coastal waters can carry many infectious pathogenic microorganisms, most commonly viruses and bacteria (Knap et al. 2002), with some estimates suggesting that this source generates over 800 million potentially contaminated seafood meals annually (Shuval 2003). Most of the ensuing illnesses result from the consumption of raw or lightly steamed molluscan shellfish harvested from waters

contaminated by sewage (IOM 1991). The majority of these infections lead to mild gastroenteritis with no associated mortality (IOM 1991). In other cases, naturally occurring organisms such as the bacteria *Vibrio vulnificus* have been associated with high mortality levels for subpopulations with liver disease or compromised immune systems (IOM 1991). As a general rule, most pathogens pose considerable risks only to the immuno-compromised and thorough cooking can eliminate virtually all microbial and parasitic pathogens (IOM 1991).

The ability of consumers to manage the risk associated with pathogen consumption is qualitatively different than their ability to manage HAB and PBT contaminated seafood, in which case even proper handling and preparation do not significantly reduce the health risks. Management of the health problems associated with these latter contaminants generally requires public intervention to prevent the seafood from reaching consumers in the first place. Although severe Minamata-type contamination incidents have not arisen in the U.S., the number of potential problems is large and expands as scientists come to better understand the biological and ecological consequences of exposure to both natural and manufactured chemicals. Two areas are of specific concern with respect to public health and the economic viability of capture fisheries: HAB and PBT contamination.

HAB Contamination in U.S. Fisheries

HAB events are common along U.S. shorelines, from the Gulf of Maine through the Gulf of Mexico and from California north to Alaska (Turgeon et al. 1998). The past three decades have seen an increase in the frequency and geographic distribution of HAB events from known sources, along with intoxication from algae not previously identified with problems (Van Dolah et al. 2005). Prior to 1970, only a few regions were affected by HABs, but now virtually every coastal state has reported major blooms (Gliebert et al. 2005).

The microscopic algae that cause HAB events play an important role in marine and freshwater ecosystems. They can become harmful, however, when they accumulate in sufficient numbers due to their production of endogenous toxins, their sheer biomass, and/or their physical shape (Gliebert et al. 2005). When these algae are used as a food source by shellfish such as clams, mussels, oysters and scallops, their toxins can accumulate in the shellfish tissues (Turgeon et al. 1998). Toxic shellfish cannot be distinguished from nontoxic ones, and the toxins are heat-resistant and not destroyed by standard cooking or processing (IOM 1991). Human consumers of these shellfish are exposed to potential illness, including paralytic shellfish poisoning (PSP), neurotoxic shellfish poisoning (NSP), diarrhetic shellfish poisoning (DSP), and amnesiac shellfish poisoning (ASP). Ciguatera fish poisoning (CFP), although not an actual algal bloom phenomenon is another illness that occurs when toxic algae on coral reef seaweeds are consumed by herbivorous fish, which pass the toxins to the larger predator fish consumed by humans (Anderson et al. 2000). *Pfiesteria piscicida*, a recently identified species of dinoflagellate, has also generated significant interest because of its potential effects on fish stocks and humans who come in direct contact with the organism (Buck et al. 1997).

PSP is perhaps the most widespread and best understood of the HAB syndromes (Van Dolah et al. 2005). PSP results from the consumption of shellfish contaminated by saxitoxins or its derivatives.³ The adverse effects of PSP are generally mild, and begin with numbness or tingling of the face, arms, and legs, followed by headache, dizziness, nausea, and loss of muscular coordination. Muscle paralysis, respiratory failure, and death can occur in cases of high-level exposure. PSP has been linked to deaths in the Pacific Northwest as far back as 1798, and is now widespread in the U.S., affecting the West Coast from Alaska to California, coastal

³ An excellent resource for information concerning saxitoxins and the other marine biotoxins mentioned in this article is provided by *ARNAT: Australian Research Network for Algal Toxins*, available online: <<http://www.aims.gov.au/arnat/arnat-00001.htm>>.

New England, and New York (National Research Council 1999). PSP events have been documented as early as 1903 in California, and they once were common on the West Coast. On the East Coast, only Maine had experienced a PSP event prior to 1972, but it has since spread throughout New England, out to Georges Bank, and currently affects more U.S. coastline than any other HAB related syndrome (Turgeon et al. 2005).

Although PSP can be found in fisheries off most U.S. coastal states, NSP events have historically been isolated in the Gulf of Mexico, with incidents dating back to the mid-16th century in western Florida and Texas coastal waters. NSP results from the consumption of shellfish contaminated by brevetoxins; the acute adverse effects of NSP include numbness, gastrointestinal upset, lack of coordination, and tingling in the mouth, arms and legs. NSP rarely results in death, and symptoms usually subside in a few days. Mississippi, Alabama, and Louisiana experienced their first NSP outbreak in 1996, and NSP events have recently extended to North Carolina.

ASP, caused by domoic acid, is a rare syndrome characterized by dizziness, headache, disorientation, and permanent short-term memory loss. In severe cases of ASP, seizures, focal weakness or paralysis, and death may occur. ASP currently is found along the western U.S. coast and in Alaska, but the organism responsible for domoic acid production has been identified in Massachusetts and in the northern Gulf of Mexico. Somewhat less severe are DSP events, which are caused by the presence of okadaic acid and result in nausea, vomiting, and diarrhea that generally last less than a week. There is also evidence, however, that okadaic acid is a tumor promoter, prompting concerns about the effect of chronic low-level exposure in humans (Van Dolah et al. 2001). DSP is not a current public health threat in the U.S. but has been documented in Nova Scotia, Canada, and should be considered a potential emerging threat (National Research Council 1999).

CFP results from the consumption of tropical reef fish contaminated by ciguatoxins. Ciguatoxins become progressively concentrated as they move up the food chain and reach particularly high concentrations in large predatory tropical reef fish. CFP affects both the gastrointestinal and neurological systems (IOM 1991). Common symptoms include nausea, vomiting, diarrhea, cramps, excessive sweating, headache, muscle aches, the sensation of burning or “pins-and-needles,” weakness, itching, and dizziness. Temperature dythesia, unusual taste sensations, nightmares, or hallucinations may also occur. CFP is rarely fatal, but some symptoms may persist for weeks, months, or even years. CFP is prevalent in virtually all tropical and subtropical U.S. waters, including Florida, Hawaii, Guam, the U.S. Virgin Islands, and many Pacific Island Territories. Ciguatoxin has been documented in at least 400 species of fish and is responsible for over half of all reported seafood-borne illnesses worldwide (Knap et al. 2002).

Recent concerns regarding *Pfiesteria* in the U.S. have garnered much more attention than other HAB issues (Kleindinst and Anderson 2001). While major fish kills were attributed to *Pfiesteria* as early as 1991, the organism was not formally identified until 1996. The health effects resulting from *Pfiesteria* exposure are still being investigated. *Pfiesteria* toxins have been blamed for causing adverse health effects in people who have come in close contact with waters where this organism is abundant, but there is no evidence of illness from the consumption of fish or shellfish exposed to *Pfiesteria* (Buck et al. 1997). Symptoms of exposure to the toxins include headaches, dizziness, a burning sensation on the skin or eyes, skin lesions or sores, nausea, intestinal distress and, in some people, short-term memory loss (N.C. DHHS). *Pfiesteria* has only been a problem during the warmer months of the year, usually between April and October, where salt waters and fresh waters mix (i.e., tidal estuaries, sounds and rivers near the coast; N.C. DHHS). *Pfiesteria* has been identified along the Atlantic coast from Delaware to Florida.

While many of the illnesses related to HABs are known, there is much that needs to be investigated. Questions remain about the diagnosis, treatment, chronic effects, and other characteristics of these poisonings (Backer and McGillicuddy 2006), and the underlying causes and triggers for most HABs are not well understood (Tyson et al. 2004). This renders it virtually impossible to accurately assess the overall health risks from exposure to HABs (Knap et al. 2002). The increase in distribution, incidence, duration and severity of HABs in recent decades suggest that HABs are an expanding public health threat (U.S. Commission on Ocean Policy 2004).

PBT Contamination in U.S. Fisheries

While HAB events are primarily the result of natural environmental processes that might be indirectly exacerbated by human activities, PBT contamination is a direct result of the industrialization of society. The presence of harmful PBT contaminants in U.S. fisheries is a growing environmental and public health concern. PBT contaminants have been discharged into U.S. waters from a variety of industrial sources for decades, and they accumulate in the tissues of fish and other aquatic organisms, with top predators in the food chain often having PBT concentrations a million times higher than that found in the water (U.S. EPA 2005c).

The geographical extent of PBT contamination in U.S. fisheries is best illustrated by examining current fish consumption advisories. These advisories inform the public that unacceptable concentrations of chemical contaminants have been found in local fish, and may include recommendations to limit or avoid eating certain fish or fish caught from a specific waterbody type (U.S. EPA 2005c). Each year the EPA compiles the National Listing of Advisories, which catalogues the fish advisory information provided to the EPA by states, tribes, territories, and local governments. The most recent listing was published in September 2005 based on 2004 data, and it included 3,221 advisories covering approximately 14 million lake

acres, or 35 percent of the nation's total lake acreage (U.S. EPA 2005c). In addition, the listing identified nearly 840,000 river miles as being under advisories, representing 24 percent of the nation's total. These figures do not include the Great Lakes and their connecting waters, all of which were under some type of advisory in 2004. With respect to marine systems, almost 65 percent of the U.S. coastline is currently under at least one advisory. Alabama, Connecticut, Florida, Georgia, Hawaii, Louisiana, Maine, Massachusetts, Mississippi, New Hampshire, New Jersey, New York, North Carolina, Rhode Island, South Carolina, and Texas all issued advisories for all of their coastal waters. In aggregate, the entire coast of the Gulf of Mexico and over 90 percent of the U.S. Atlantic coast are under advisories for at least one species of fish; specific species generally vary by state. Among the Gulf Coast states (with the exception of Florida), however, statewide coastal advisories are in effect only for king mackerel (*Scomberomorus cavalla*) because of mercury contamination (U.S. EPA 2005b). The Pacific coast has several local areas under advisory, but no statewide advisories have been issued. Hawaii also has a statewide advisory in effect for the PBT contaminant mercury in several fish species.

Under programs currently active, states, tribes, territories, and local governments issue advisories for 36 different PBT contaminants, with almost 98 percent of the advisories involving only five PBT contaminants: mercury, chlordane, dioxin, PCBs and DDT. These five contaminants have received increased public attention because they pose considerable threats to public health. Specifically, these five contaminants have been linked to adverse effects on the human nervous and reproductive systems, and they are known to cause problems in the form of irregular fetal development, human cancer, and other genetic abnormalities (U.S. EPA 2004a). In addition to the public health impacts, these contaminants can also significantly affect the economic viability of capture fisheries when harvesting prohibitions are instituted for

contaminated areas. Given their importance to PBT contamination issues, the remainder of this section will focus solely on these five contaminants.

Mercury is a persistent metal that is distributed throughout the environment and originates from both natural sources and human activities. Its organic form, methylmercury, accumulates in the fatty tissues of fish and, once ingested, can cause irreversible human health effects (U.S. EPA 2001). The human nervous system is very sensitive to all forms of mercury, and exposure to high levels of methylmercury can permanently damage the brain, kidneys, and developing fetus (ATSDR 1999). Dietary intake is the dominant source of mercury exposure for the general population, and over 76 percent of the 2004 National Listing of Fish Advisories focused on mercury contamination, including freshwater incidents in the states of Connecticut, Florida, Illinois, Indiana, Kentucky, Maine, Maryland, Massachusetts, Michigan, Minnesota, Missouri, Montana, New Hampshire, New Jersey, North Dakota, Ohio, Pennsylvania, Rhode Island, Vermont, Washington, and Wisconsin (U.S. EPA 2005c). In addition, the coastal states of Alabama, Florida, Georgia, Louisiana, Maine, Massachusetts, Mississippi, New Hampshire, North Carolina, Rhode Island, South Carolina, and Texas issued statewide advisories for mercury in their coastal marine systems, while Hawaii instituted a statewide advisory for mercury in marine fish.⁴ Humans with heavy dietary reliance on seafood have the highest concentrations of methylmercury in their tissues; individuals classified as poisoned in the Minimata case had mercury concentrations as high as 50–100 parts per million (ppm) compared to less than 1 ppm in those who consume only 10–20 grams of fish per day (Dewailly and Knap 2006).

⁴ Two tribes have also issued mercury advisories in 2004. The Mi'kmaq tribe of Maine had two tribal statewide advisories in effect for mercury in freshwater and marine fish, including lobster, while the Cheyenne River Sioux Tribe had one tribal statewide advisory for mercury in rivers, lakes, and stock ponds.

In contrast to the naturally occurring element mercury, PCBs are a group of synthetic organic chemicals that can also cause a number of harmful effects in humans. Once widely used as coolants and lubricants in transformers, capacitors, and other electrical equipment, PCB production ceased in the U.S. in 1977. Nonetheless, PCBs are still found in the environment and have been associated with acne-like skin conditions in adults and neurobehavioral and immunological changes in children (ATSDR 2000). Studies implicate PCBs in a variety of adverse human health effects on reproduction, neurobehavioral development, liver function, birth weight, and immune response (Dewailly and Knap 2006). The major source of human PCB exposure is through the consumption of contaminated seafood, and the National Listing of Fish Advisories reported that there were over 4.6 million lake acres and more than 110,000 river miles under PCB advisories in 2004 (U.S. EPA 2005c). Indiana, Minnesota, New York, and the District of Columbia issued statewide freshwater advisories for PCBs, while Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New York, and Rhode Island issued PCB advisories for all of their coastal marine waters.

Chlordane, another synthetic chemical and widely used as a pesticide in the U.S. prior to 1983, can cause damage to the nervous system and liver in humans at high levels of exposure.⁵ Exposure occurs primarily from eating contaminated foods, such as root crops, meats, fish, and shellfish, or from touching contaminated soil (ATSDR 1994). The National Listing of Fish Advisories identified chlordane advisories for nearly 850,000 lake acres and 54,000 river miles in 2004, even though many advisories have been rescinded in recent years because the chemical is no longer used and continues to degrade in the environment (U.S. EPA 2005c).

⁵ The U.S. EPA banned all uses of chlordane in 1988 over concerns of harm to the environment and human health.

Dioxins, chemicals that are formed during combustion processes such as commercial or municipal waste incineration and from burning fuels such as oil, wood, and coal, have been at the root of some of the more highly publicized environmental and human contamination incidents from PBTs, ranging from the use of dioxin-containing defoliants during the Vietnam war to the inadvertent contamination of groundwater at New York's Love Canal. High levels of exposure to dioxins can result in a number of adverse health effects, including chloracne, skin rashes, skin discoloration, excessive body hair, and liver damage. Although most dioxin exposure occurs through dietary intake of animal fats, it is thought that the majority of the U.S. population has a relatively low-level of exposure to dioxins (CFSAN 2006). Consequently, the geographic extent of dioxin advisories is less widespread when compared to that of the other four major contaminants, accounting for only approximately 23,000 lake acres and slightly more than 2,300 river miles under advisory in 2004 (U.S. EPA 2005c).

The last of the five primary PBTs, DDT is a pesticide once widely used to control insects that damaged crops and carried diseases such as malaria. The use of DDT in the U.S. was banned in 1972 because of damage to wildlife, but DDT is still used in some countries. Exposure to DDT, and the chemicals it breaks down to in the environment (DDE and DDD), occurs mostly from eating meat, fish, and poultry that contain small amounts of these compounds. Exposure to high levels of DDT can affect the nervous system and cause excitability, tremors and seizures. In women, DDE can lead to a reduction in the duration of lactation and an increased chance of premature birth (ATSDR 2002). Although the use of DDT has been banned in the U.S. since 1972, the 2005 National Listing of Fish Advisories reported advisories for DDT, DDE, and DDD that covered more than 840,000 lake acres and 69,000 river miles (U.S. EPA 2005c). California had the greatest number of DDT advisories in effect, followed by Maine and Massachusetts.

Economic Consequences of Past HAB Contamination

The potential economic consequences of a HAB contamination event are vast in scope, ranging from increased public health costs and direct losses in commercial fisheries to reduced revenues from curtailed recreation and tourism. Public health costs include direct medical costs, lost workdays, and lost productivity. Commercial fishery impacts include shellfish bed closures or quarantines, wild or farmed fish mortalities, and the loss of income due to these closures and mortalities (WHOI 1998). An additional effect on commercial fisheries is the reduced fish and shellfish demand resulting from consumer fear of contaminated seafood, and these impacts may stretch to seafood products unaffected by contamination issues. Algal blooms that result in dead fish or shellfish washing up on beaches, discolored water, noxious odors, and/or human respiratory problems from toxins released into the air can be significantly costly in terms of lost recreation and tourism opportunities (WHOI 1998). Additional economic costs are also incurred in maintaining monitoring and testing programs designed to detect algal toxins and in cleaning up fish or shellfish kills when they do occur.

Few studies have attempted to quantify the economic impacts of HAB events in the U.S. Most coastal states have neither conducted economic analyses of HABs nor collected data that might be used to estimate reliable economic impacts. The most comprehensive effort to date was done by Anderson et al. (2000) and focused on direct and indirect costs for a subset of HAB events occurring during the years 1987–1992. Estimated average annual total economic impacts for HABs were calculated at US\$46 million, with public health effects comprising the largest proportion of the economic impacts at almost US\$20 million (Hoagland et al. 2002).⁶ Commercial fishery losses accounted for an additional US\$18 million annually, while recreation

⁶ Hoagland et al. (2002) summarizes the results from Anderson et al. (2000) and also updates the previous estimates. All values reported in this article were adjusted to reflect year 2000 U.S. dollars.

and tourism impacts were estimated at US\$7 million annually (Hoagland et al. 2002). In addition, monitoring and management costs contributed another US\$2 million annually (Hoagland et al. 2002). These estimates, however, are considered to be highly uncertain given the lack of information about the overall effects of many HAB events and the difficulty in assigning a dollar cost to many of the impacts associated with the events (Anderson et al. 2000). While it may be impossible to quantify all economic impacts, there is little doubt that HABs can have significant and serious effects at local and regional levels (Anderson et al. 2000).

Of all the economic consequences of HAB events, the public health impacts have been the most clearly documented. For example, cases of sickness and death from shellfish poisoning are recorded by state and federal public health agencies. Based on these reports, shellfish poisonings due to HAB events appear to be a minor cause of seafood poisoning in this country, although it is widely believed that poisonings are underreported because the resulting illnesses are attributed to other causes or are not severe enough to prompt the victim to seek medical attention. For those shellfish poisonings that are reported, estimates of economic impacts usually include costs associated with lost worker productivity, medical treatment, transportation, and investigation of the incident. Estimates of the economic impact of unreported cases can also be calculated, but they obviously would not include the cost of medical treatment and transportation. Using estimates of \$1,400 per reported illness, \$1,100 per unreported illness, and US\$1 million per death, Hoagland et al. (2002) suggest that the average annual combined PSP, NSP, and ASP costs are US\$400,000. In contrast, CFP accounts for the majority of public health impacts from HABs and is estimated to cost the nation approximately US\$19 million annually (Hoagland et al. 2002). This estimate is conservative, as it does not include the additional liability insurance purchased by many seafood companies to help protect them from ciguatera-related litigations. Under current law, seafood sellers can be held liable for damages due to

exposure to toxic fish, even if care was exercised in the preparation and sale of the product (Steidinger et al. 1999). For example, victims have successfully litigated against restaurant operators to recover CFP-related damages (Nellis and Barnard 1986).

The economic effects of HABs on commercial fisheries can be as substantial as the public health impacts, and they range from the direct impacts associated with wild harvest closures, harvesting delays, and lost aquaculture production to the indirect impacts associated with untapped resource exploitation opportunities. HAB events during the period 1987 to 1992 were estimated to directly cost the U.S. commercial fishing sector between US\$7 million and US\$19 million per year (Hoagland et al. 2002). Some of the more significant events included a 1987 NSP outbreak in North Carolina resulting in a lost harvest of clams, oysters, scallops, and finfish worth US\$8.27 million, a recurring brown tide in New York that led to bay scallop mortality and lost harvests of US\$3.27 million annually, US\$17.64 million in losses due to phytoplankton-related mortality in Washington's farmed Atlantic salmon during 1987, 1989, and 1990, and recurring closures of Alaska's surf clam fishery due to PSP that led to estimated annual losses as high as US\$9.14 million (Anderson et al. 2000). Indirectly, repeated incidents of HAB intoxication have led to opportunity costs associated with shellfisheries that were unable to be developed commercially. One example of this is the potential Alaskan shellfishery, where the recurring presence of PSP toxins in most coastal areas has prevented the commercial development of the industry. Estimates of the forgone benefits due to this presence of PSP range from a high of US\$50 million to a low of US\$6 million annually (Neve and P.B. Reichardt 1984; Anderson et al. 2000).

While Anderson et al. (2000) highlighted the economic consequences of HAB events from 1987–1992, there were significant events in other years that illustrate the enormity of the potential economic impacts. For example, the entire Maine coastline was closed to shellfishing

following a PSP event in September 1980. Direct harvest losses were estimated at slightly over US\$5 million, with total economic impacts (direct and indirect) estimated at over US\$15 million (Shumway, Sherman-Caswell, and Hurst 1988). PSP also affected the west coast in 1980, closing oyster harvesting for one month in California, Oregon, and Washington and generating an estimated US\$1.3 million in total losses (Nishitani and Chew 1988). A 1997 *Pfiesteria* bloom in the Chesapeake Bay was particularly costly, having resulted in a fish kill of 30,000 menhaden and physical and neurological problems in fishermen that prompted the closure of several Chesapeake tributaries to fishing and recreation (Lipton 1999). The negative publicity from this event significantly decreased demand for Maryland seafood despite the state's promotional efforts to restore consumer confidence. Lipton (1999) estimated that over US\$45 million in lost seafood sales were directly attributable to the 1997 *Pfiesteria* outbreak. These losses are especially large given that the events were confined to a relatively small area, only a few commercially important species were affected, and there was no scientific evidence that *Pfiesteria* posed a significant public health threat to the consumers of Chesapeake Bay seafood products (Lipton 1999).

As demonstrated by the Chesapeake Bay *Pfiesteria* experience, the economic impacts of contamination can reach far beyond the immediate geographic area of contamination. Negative publicity surrounding a contaminated seafood product often adversely affects other seafood products, even if those products are only remotely associated with the contamination event. This phenomenon, termed the “halo effect,” was illustrated in an early study by Jensen (1975) that examined the economic consequences of a 1972 New England red tide and PSP event on the New York shellfish industry.⁷ The harvest and sale of soft shell clams, hard shell clams, and mussels were banned from Maine to Massachusetts resulting in significant loss of income for

⁷ Jensen (1975) appears to have been the first to use the term “halo effect” for this market perception phenomenon.

shellfish diggers and their dealers, who were forced to destroy stocks of shellfish on hand. To compound the impact, however, the negative publicity led to decreased consumer demand for fish, lobsters, and sea scallops even though these products were never linked to PSP. In addition to these local New England effects on the shellfish and seafood markets, New York was also negatively impacted by the adverse publicity, as consumer demand fell for hard shell clams in the city and this led to reduced landings from state waters (Jensen 1975).

Economic Consequences of PBT Contamination

The economic impacts of HAB events, while potentially large for isolated geographic areas, are typically (although not always) limited in duration. In contrast, the persistent, toxic nature of PBT contaminants suggests that they have the potential for significant economic implications over long periods of time. Like HABs, PBT contamination leads to public health costs, losses to commercial fisheries, reduced recreation and tourism, and monitoring and management costs. But, while PBT contaminants pose significant threats to public health, their effect on public health through fisheries is different than that observed for HABs. Where the symptoms of most HAB-related shellfish poisonings subside relatively quickly, the health effects of PBT exposure are often irreversible. Even so, little research exists concerning the linkage between PBT concentration and economic losses from poor health, primarily because most scientific efforts have examined linkages between exposure and health, not health and the ensuing impacts such as lost work days and reduced productivity (Brook 2002).

One study that did attempt to examine the linkage between health and the economic impacts of PBT contamination was a 1996 investigation of children born to women who had eaten Lake Michigan fish contaminated with PCBs. The study demonstrated that prenatal exposure to PCBs led to lower full-scale and verbal IQ scores, with the strongest effects related to memory and attention (Jacobson and Jacobson 1996). The most highly exposed children were

three times more likely to have low average IQ scores and twice as likely to be at least two years behind in reading comprehension. If the exposure of women to PCB contamination occurs on a wide enough scale, the magnitude of the economic costs to society would be tremendous given that a single point decrease in the average IQ for the population can potentially lead to lost earnings in excess of US\$31 billion annually (Muir and Zegarac 2001).⁸ In a separate study that examined the relationship between childhood development and methylmercury exposure, researchers found that 316,000 to 637,233 children each year have cord blood mercury levels higher than levels that have been associated with loss of intelligence, measured in IQ points (Trasande, Landrigan, and Schechter 2005). This potential loss of intelligence may cause reduced economic productivity over the lifetime of these children, a cost that was estimated to be US\$8.7 billion annually ((Trasande, Landrigan, and Schechter 2005). Although this latter study did not consider specific sources of exposure, the primary means by which humans are exposed to methylmercury is through fish consumption.

Several studies examining the economic effects of exposure to PBTs contaminating the New York Bight-Hudson River Estuary were summarized by Ofiara and Seneca (2001). Estimates of the excess cancer risk, and the associated economic losses, from the consumption of PBT-contaminated seafood were examined for a variety of species, contaminants, contamination levels, and rates of seafood consumption. Excess risk and the resulting economic impacts were highest for PCB-contaminated white catfish and white perch, each ranging from US\$5.3 billion to \$70.4 billion in losses. The net economic costs associated with excess cancer mortality from consuming PCB contaminated striped bass ranged from US\$3.7 billion to \$21.7 billion (assuming low consumption rates) up to \$8.8 billion to \$51 billion (assuming a high

⁸ As with the values reported for HABs, reported PBT-associated impacts were adjusted to reflect year 2000 U.S. dollars.

consumption rate). For contaminated bluefish, impacts ranged from US\$3.7 to \$50.4 billion depending on consumption rate. Dioxin-associated risk was smaller in magnitude than PCB-related risk, and as a result dioxin-contaminated striped bass economic impacts ranged from \$1.3 billion to \$9.1 billion. Additive risks were also examined by the authors because many species are affected by more than one contaminant. Striped bass, a predatory species, not surprisingly exhibited the highest levels of excess risk and additive risk, with PCB accounting for the highest individual contaminant risk in this species, followed by DDT and chlordane. Taken together, the additive risks of contamination in striped bass were estimated to generate public health losses that ranged in value from US\$1.7 billion to \$34.6 billion. Given the breadth of the studies surveyed by Ofiara and Seneca (2001), their estimates were at times necessarily based on imprecise data. Regardless, the reported values highlight the potentially sizeable public health losses that may be caused by the consumption of contaminated seafood over a lifetime.

In addition to the public health-related economic impacts, commercial and recreational fishery closures resulting from PBT contamination can also have significant economic consequences. Waterbodies in New Bedford Harbor and the Buzzards Bay areas of Massachusetts have been closed to lobster harvesting since 1979 as a result of PCB contamination, forcing local lobster fishers to travel greater distances or discontinue harvesting (McConnell and Morrison 1986). Those who steam to unclosed areas for harvesting were estimated to incur an increase in costs of US\$1,749 annually ((McConnell and Morrison 1986). While no economic estimates of losses are available, the Hudson River commercial and recreation fishery has also been subject to closures and fish consumption advisories for most of the past 30 years due to the presence of high levels of PCB contamination in fish (NYSDEC 2001). For example, recreational fishing in the upper Hudson River was prohibited from 1976 until 1995, after which the fishery was designated as catch and release only. Considering this,

along with the continuing closure and harvest restrictions on a number of potentially important commercial species, it is realistic to assume that untapped fishery resources and reduced recreational fishing opportunities have led to significant economic losses.

While the direct economic effects associated with PBT contamination in commercial and recreational fisheries are often large and extend over a long period of time, the indirect economic impacts may also be significant. For example, a hypothetical study by Jackus, McGuinness, and Krupnick (2005) estimated the surplus losses from decreased demand following negative publicity and public awareness concerning mercury contamination. After estimating supply and demand models for striped bass in the Chesapeake Bay, the effect of a potential consumption advisory was modeled as a leftward shift of the demand curve. The resulting combined producer and consumer surplus losses exceed US\$500,000 solely for the commercial striped bass market in the Maryland portion of the Chesapeake Bay.

MANAGING CONTAMINATED FISHERIES

The scope and scale of HAB and PBT contamination problems in fisheries is difficult to quantify on an aggregate basis, but they are clearly expanding and having an impact on how the harvesting and processing sectors operate and on how consumers perceive seafood products. As a result, managers must confront the threats that contamination poses to public health and the economic viability of the fishing industry. Faced with the negative effects of HABs, one approach has been to attempt to minimize the impacts through routine monitoring programs and harvesting closures when necessary. Thus, successful management of fish and shellfish resources in the presence of contamination has traditionally depended on an active monitoring strategy that is able to detect the toxins that pose a threat to human health.

The National Shellfish Sanitation Program (NSSP) was developed in 1925 as a response to public health concerns after typhoid fever outbreaks were traced to sewage contaminated

oysters (Herwig 2001). In this program, individual states monitor their shellfish growing water to determine safety before harvesting, and the FDA periodically audits the states' efforts to guarantee safety. The NSSP requires tagging and labeling of all shellfish to ensure that only shellfish harvested from approved waters reach market and to allow the tracing of product that might later be determined contaminated. The *2003 Guide for the Control of Molluscan Shellfish* is the manual of operations for the NSSP, and represents current science on safe and sanitary control of the growing, processing, and shipping of molluscan shellfish for human consumption (NSSP 2003). With respect to HABs, the guide provides action toxin levels for PSP, NSP, and ASP, and also presents guidelines for developing a marine biotoxin contingency plan (NSSP 2003). For example, the manual suggests that shellfish areas are to be closed when PSP toxin levels equal or exceed 80 micrograms (μg) per 100 grams (g) in the edible portion of raw shellfish, when ASP levels equal or exceed 20 ppm in the edible portion of raw shellfish, when any NSP toxin is found in shellfish meat, or when the cell counts for members of the genus *Karenia* in the water column exceed 5,000 cells per liter (NSSP 2003). In developing a contingency plan, the guidelines suggest gathering and evaluating intelligence and surveillance information, implementing early warning systems, establishing procedures to define severity of occurrence, and identifying the steps necessary to eventually return contaminated growing areas to harvestable status.

Under NSSP guidelines, individual states are responsible for the detection and monitoring of marine toxins in their coastal waters. Programs vary widely with differing points of responsibility and experience, from long-established monitoring programs to those in development.⁹ An example of a typical monitoring effort is the PSP monitoring program in

⁹ NOAA's Coastal Services Center maintains links to various state and regional HAB programs on its website, available online: <<http://www.csc.noaa.gov/crs/habf/resources.html>>.

Massachusetts, where the Division of Marine Fisheries conducts annual coastline PSP surveys from April until November (Massachusetts Division of marine Fisheries). Shellfish samples are collected on a weekly basis from 16 locations. When biotoxin levels rise above 50µg per 100g,¹⁰ sampling is conducted more frequently at the affected sites. If shellfish toxin levels exceed 80µg per 100g, the area is closed to shellfishing. Notices of closures are made to shellfish constables and town officials in the affected areas, and a computerized e-mail notice is sent to state personnel responsible for monitoring PSP events and to the state's Environmental Police. Every week following the first event, an informational e-mail is sent to select state agencies informing them of the ongoing status of the contamination. When toxin levels fall below 80µg per 100g for three consecutive samples, notices reclassifying the affected areas are prepared to ensure rapid re-opening of shellfish growing areas.

The success of monitoring programs like the one in Massachusetts is dependent on many variables. The toxins responsible for contamination must be known, levels must be established for harmful concentrations, and their health impacts must be understood. An effective communication strategy must also be in place to inform necessary state personnel of the current status of the contamination as well as to make the public aware of any potential health threats. The NSSP guidelines are designed to control those variables and ensure successful monitoring.

As the frequency and geographic extent of contamination has increased, the U.S. has been forced to develop a comprehensive research, notification, and management plan for dealing with HAB events. Management strategies to address the potential public health impacts of HABs depend on a successful research plan to optimize management and mitigation strategies. Effective management of HAB contaminated fisheries requires a collaborative effort among state

¹⁰ The Massachusetts Division of Marine Fisheries web site actually lists this as 50 µg per 100 mg. David Whittaker, a marine fisheries biologist at the Pocasset Office, confirmed that this is incorrect in a phone call on July 11, 2006. The toxin levels should be in terms of µg of toxin per 100 g of shellfish tissue.

and federal agencies, the research community, and regional and local resource managers to develop communication, research, funding, and monitoring programs. To this end, the report *Marine Biotoxins and Harmful Algae: A National Plan* has served as the blueprint for national HAB research activities (Anderson, Galloway, and Joseph 1993). The plan was developed in 1993 with the goal of achieving effective fisheries management, protecting public health and solving ecosystem problems related to marine biotoxins and harmful algae. Research goals of the national plan include the characterization of the chemical structures and pharmacological actions of toxins and their derivatives, the development of specific detection methods based on these characterizations, the development of forecast capabilities, and the determination of the source, fate, and consequences of algal toxins in fisheries. Management goals include the development of mitigation strategies to minimize the impacts of HABs, the identification of and improved access to HAB-related databases, the development of an effective communication program, and the institution of a rapid response to HAB outbreaks.

In order to meet these objectives, a range of national and local programs and agencies have focused their work on various aspects of the HAB problem (e.g., ecology, toxicology, monitoring, mitigation, human health, and education; HARRNESS 2005). The U.S. National Office for Marine Biotoxins and Harmful Algal Blooms, located at the Woods Hole Oceanographic Institution, was established in 1995 to aid in the development of a multidisciplinary HAB agenda and is now the center of a national HAB communication program (Turgeon et al. 1998). The National Office provides access to HAB information, including the latest research developments, workshop reports, research strategies and relevant data. In addition, the National Office helps coordinate the efforts of federal agencies, the academic research community, and resource managers.

Early efforts under the National Plan focused on the development of a strategic research program aimed at understanding the ecology and oceanography of HABs. Termed ECOHAB (for the Ecology and Oceanography of Harmful Algal Blooms), the program was implemented in 1995 to address scientific and monitoring needs, and consisted primarily of a competitive, peer-reviewed research grant program. A critical ECOHAB goal was the development of reliable models to forecast blooms, persistence, and toxicity (Turgeon et al. 1998). Projects funded through ECOHAB have directly led to enhanced abilities to monitor, predict, and mitigate HABs.

MERHAB (for Monitoring and Event Response for Harmful Algal Blooms) was another effort developed under the National Plan. Initiated in 1999, MERHAB's goal was to fund research projects that helped to expand the number of coastal regions benefiting from advancements in algal identification, detection, modeling, and prediction (CSCOR MERHAB). The focus of MERHAB on the development and adoption of new technologies has allowed for the proactive detection of HAB events (CSCOR MERHAB). Initial MERHAB efforts improved HAB monitoring in the Chesapeake Bay and along the Olympic Peninsula in Washington. The Chesapeake Bay project has produced new real-time tools to measure environmental parameters in critical shallow water areas at unprecedented temporal and spatial resolutions (CSCOR MERHAB). The Olympic Region Harmful Algal Bloom (ORHAB) project has successfully integrated knowledge from current ECOHAB research into state and tribal coastal management. ORHAB/MERHAB projects have allowed Washington State to anticipate the need to close recreational and commercial shellfisheries while retaining the public trust necessary to enforce current and future closures. In addition, other projects in this region have focused on rapid, cost-effective, reliable, and highly sensitive toxin detection methods that hold promise for estimating the public health risk from chronic exposure to low levels of algal toxin. MERHAB also includes

those research projects that will enhance monitoring and response capabilities for red tide in the Gulf of Mexico, freshwater toxic algae in the Great Lakes, and domoic acid along the California coast.

Another effort to minimize HAB impacts on coastal communities is the Center for Sponsored Coastal Ocean Research (CSCOR) HAB Event Response program. This program strives to avoid another large-scale HAB-related incident like the 1997 *Pfiesteria* outbreak. In the event of another major episode, CSCOR serves as the National Coordinator of the Federal Event Response Plan for HABs. This plan creates a formal mechanism, initiated by State request, to utilize federal resources on HAB-related events that overwhelm state capabilities (CSCOR HAB). Upon notification of an event, CSCOR and the National Office for Marine Biotoxins and Harmful Algal Blooms work to provide access to the best technology and expertise available, provide supplemental financial support for investigating a unique event, and ensure proper scientific documentation to add to the HAB knowledge base (CSCOR HAB). Since 2003, the CSCOR HAB Event Response program has responded to events impacting states along the East, West, and Gulf Coasts. These responses have addressed a wide range of state and federal management and scientific needs, including assessing human health risks, identifying causes of marine mammal mortalities, offering training opportunities for managers, and establishing baseline conditions for new or re-emerging HABs.

The nature of management response to PBT contamination has differed from those designed for HAB events because PBT pollutants have the ability to travel long distances, to travel easily among air, water, and land, and to linger for generations in people and the environment (U.S. EPA 2004a). These characteristics present challenges in reducing the public health risks from PBT contaminants. While federal, state, local and tribal agencies have various responsibilities for safeguarding the public against effects of PBT contaminants in fish, most

management strategies encompass long-term pollution control, environmental remediation, and the issuance of health advisories with recommendations about limiting fish consumption and/or adopting other risk-reducing behaviors.

The FDA and EPA are the federal agencies most involved with limiting consumer contaminant exposure. The FDA develops advisories and sets maximum allowable contaminant levels for commercially marketed fish. The EPA is also active in many areas relating to fish contamination, particularly with controlling pollutant releases and issuing consumption advisories. Both agencies actively provide technical assistance and guidance to state, local, and tribal agencies. Many states rely on FDA consumption guidelines for advisories and consult frequently with the FDA on addressing particular fish contaminant situations (U.S. EPA 2005a). This cooperative approach makes sense in part because the FDA has the scientific expertise to determine federal tolerances, action levels, and guidance levels for many of the most harmful contaminants present in fish. Examples of action limits above which the FDA will take legal action to remove products from the market include 1 ppm for mercury, 20 ppm for PCBs, 0.3 ppm for chlordane, and 5 ppm for DDT (U.S. FDA 2000). States often use these same action limits for issuing consumption advisories.

To augment the activities of the FDA, the EPA has already taken action against many of the PBT contaminants present in the nation's fish supply, making it a priority to reduce risks to human health and the environment from existing and future exposure to priority PBT pollutants (U.S. EPA 2004a). A four-part strategy was developed by the EPA that includes the development and implementation of national action plans to reduce priority PBT pollutants and prevent new PBT pollutants from entering the marketplace (U.S. EPA 2004a). Mercury emissions have been greatly reduced since 1990, and will be reduced further by the implementation of the Clean Air Mercury Rule (CAMR) to regulate mercury emissions from coal-fired power plants. CAMR, the

first in the world of its kind, created a market-based cap and trade program to permanently reduce mercury emissions. Although these programs aim to significantly reduce the new deposition of contaminants into the environment, their persistence makes it likely that many of them will remain in the nation's fish stocks for some time to come.

As alluded to above, consumption advisories and safe eating guidelines are the primary management strategies that have been used in the U.S. to reduce consumer exposure to contaminants in fish. Simply defined, consumption advisories are recommendations for voluntary action, informing the public that excessive concentrations of chemical contaminants have been found in local fish. These advisories may include recommendations to limit or avoid eating certain fish species or fish caught in specific water bodies. An advisory may be issued for the general population or for sensitive subpopulations such as pregnant women, nursing mothers, and children (U.S. EPA 2005c). Each state or tribe is responsible for developing their own advisory programs and issuing consumption advice. This heterogeneous structure leads to program variability across the U.S., but, in general, there are five major types of advisories and bans that are issued (U.S. EPA 2005c):

1. No-consumption advisories for the general population, issued when contaminant levels in fish pose a health risk to the general public;
2. No-consumption advisories for sensitive subpopulations, issued when contaminant levels in fish pose a health risk to sensitive subpopulations;
3. Restricted-consumption advisories for the general population, issued when contaminant levels in fish may pose a health risk if too much fish is consumed;
4. Restricted-consumption advisories for sensitive subpopulations, issued when contaminant levels in fish may pose a health risk if too much fish is consumed by those in the sensitive subpopulation; and

5. Commercial fishing bans, issued when high levels of contamination are found in fish caught for commercial purposes.

In addition to the advisories issued by individual states, the federal government has also issued fish consumption advisories pertaining to mercury. In the first ever joint advisory, the EPA and FDA recommended that women who might become pregnant, women who are pregnant, nursing mothers and young children avoid eating shark, swordfish, king mackerel, and tilefish because of the high levels of mercury in the fish (U.S. EPA 2004b). The agencies also advised limiting consumption of albacore tuna to 6 ounces per week for the same target population.

Do Current Strategies Work?

The National Plan has served as the U.S. HAB program foundation for the last decade, guiding the planning efforts that have led to implementation of numerous national, regional, state, and local research and monitoring efforts such as ECOHAB and MERHAB (Anderson and Ramsdell 2005). These coordinated programs led to drastic improvements in the capabilities and resources used to detect and monitor HABs and their toxins (Ramsdell, Anderson, and Gliebert 2005). The efficacy of state monitoring for toxins is evidenced by the rarity of fatalities and illnesses from known toxins (Anderson 2002). Federal-state partnerships have proven successful in rapidly responding to serious events such as the 1997 *Pfiesteria* outbreak in Maryland. Access to information has also greatly improved due to the development and maintenance of HAB-dedicated Web sites by agencies such as the National Office of Marine Biotoxins and Harmful Algal Blooms and various state Sea Grant Offices.

In contrast to the progress made with HABs, the FDA/EPA's PBT strategy's level of success is still unknown. The PBT programs have made progress in minimizing the use of these contaminants and reducing the amounts that are released into the environment. These reductions,

however, do not necessarily lead to decreases in the contaminant concentrations found in fish, at least in the short run. In order to provide a baseline for tracking progress in dealing with PBT contaminants, the EPA conducted the 2000–2003 National Lake Fish Tissue Study to estimate the national distribution of 268 PBT chemicals in fish tissue from lakes and reservoirs in the contiguous United States (U.S. EPA 2005d). Given their persistence in the environment, it may be a decade or more before progress on reducing human exposure to fish-borne PBTs can be definitively demonstrated.

Perhaps one of the difficulties the national PBT programs will encounter when trying to demonstrate reduced human exposure relates to the potential effectiveness of consumption advisories that depend on voluntary consumer behavior. Consumer reactions to advisories have previously been inconsistent, and the advisories ultimately will only be effective if consumers are aware of them and are willing/able to translate awareness into behavior (Shimshack, Ward and Beatty 2005). Shimshack et al. (2005) examined consumer response to the 2001 FDA methylmercury fish advisory and found that a large group of at-risk consumers did not respond to the advisory, particularly in the case of less-educated and less-informed consumers.¹¹ Additionally, they found that providing public information may lead to a broader response than intended, as non-targeted consumers also reduced fish consumption after the mercury advisory. These unintended responses can have significant effects on overall public health. Fish consumption advisories raise the possibility of the classic risk-risk trade-off: by avoiding one risk, that of contaminant exposure, consumers may incur another risk, adverse health consequences due to lower Omega-3 fatty acid intake (Cohen et al. 2005). Trade-offs from consumption-altering policies were recently examined in a study by the Harvard Center for Risk

¹¹ The FDA released this advisory in January 2001. The advisory singled out infants, small children, pregnant or nursing mothers, and women who may become pregnant, advising them to limit consumption of all fish to no more than 12 ounces per week, and to avoid entirely fish known to contain high levels of mercury.

Analysis (Cohen et al. 2005). If women of childbearing age shift their consumption from higher mercury fish to lower mercury fish (i.e., adhere to recommendations), positive public health benefits are realized (Cohen et al. 2005). If non-targeted consumers also reduce their level of fish consumption, substantial overall public health losses can occur, particularly with respect to the sub-population of elderly men (Cohen et al. 2005). This study brings up another interesting, though currently unaddressed question: if subpopulations such as women of childbearing age reduce their consumption of fish with high mercury concentrations, then will other groups, particularly the poor, increase their mercury exposure (Willett 2005)? If the informed public reduces demand for mercury contaminated fish, market forces will lead to reduced prices for those species, thereby making it more likely that they will be consumed by the poor and/or less informed consumers (Willett 2005).

The Costs of Managing Contamination

Economic costs are incurred through management, monitoring and testing programs designed to detect algal toxins and clean up fish or shellfish kills when they do occur. Hoagland et al. (2002) report average annual costs of US\$2 million for monitoring and managing HABs. This figure is based on estimates from only 12 states (Alaska, California, Connecticut, Florida, Maine, New Hampshire, Massachusetts, North Carolina, New Jersey, New York, Oregon, and Washington) and is likely conservative given the difficulty in obtaining cost data. Many coastal states do not have a regular monitoring program, and among those that do, monitoring tasks are spread across different state and local agencies. This leads to problems in trying to identify the costs that pertain specifically to monitoring and management. It should be noted, however, that the measured and presumed costs of HAB management are likely to be significantly smaller than the estimated US\$20 million in public health impacts. Cost estimates associated with managing PBT contamination are even more difficult to ascertain, and it does not appear that any focused

studies of this issue have been undertaken. Given that management of these toxins is spread across many federal, state, and local agencies, data aggregation problems are likely to be at least as severe as in HABs. In addition to sampling and testing costs, the development of advisory programs, compilation of data, maintenance of databases, and communication of advice are also costly and should be included in any future cost estimations.

CAN CONTAMINATION MANAGEMENT BE IMPROVED?

Although HAB contamination management has greatly improved over the past few decades, and many recommendations of the original National Plan have been met, other recommendations remain partially or completely unfulfilled. Given that the HAB problem continues to grow and change in terms of geographic extent and the emergence of new poisoning syndromes, it is necessary to continually update and expand the National Plan so that future financial, human, and physical resources are directed to priority HAB topics (Ramsdell, Anderson, and Gliibert 2005). The current update, termed Harmful Algal Research and Response: A National Environmental Science Strategy 2005–2015 (HARRNESS), is intended to serve as a framework for research and management actions over the next decade (Ramsdell, Anderson, and Gliibert 2005). HARRNESS addresses priority topics in four focus areas—bloom ecology and dynamics, toxins and their effects, food webs and fisheries, and public health and socioeconomic impacts. HARRNESS recommendations include establishing standard reporting procedures for HAB incidents, developing rapid, field-based detection methodologies, early warning systems, and effective techniques to control and reduce HAB impacts, as well as modeling long-term HAB risk exposure and socioeconomic impacts and the cost of mitigating HAB events at local and regional scales. In contrast to previous efforts, HARRNESS focuses not only on marine HABS, but also on the growing HAB problem in freshwater systems. HARRNESS provides the conceptual framework for improved HAB management, but needs an

implementation plan to make this a reality. A combination of existing programs such as ECOHAB and MERHAB, along with restructured and new programs, will be required to meet the goals of HARRNESS. After full implementation, HARRNESS is expected to produce significant benefits across several areas, including improvements in the ability to detect HAB species and analyze HAB toxins, the monitoring and forecasting of HABs, the protection of public health, the creation of prevention and mitigation strategies, the estimation of economic costs associated with HAB events, and the calculation of the economic impacts on aquaculture and shellfish safety.

A recent approach to managing PBT contaminant exposure that fits into the general objectives of HARRNESS is the introduction of the nation's first line of certified low-mercury fish under the Safe Harbor brand name (Hirsch 2008). In a current test, Safe Harbor is marketing a low-mercury line of fresh fish in Northern California supermarkets to see if consumers would increase fish purchases if they were provided with more information about the product's mercury content. Safe Harbor utilizes a new analytical device that measures mercury content in less than a minute, and they aim to only market fish that test well below the FDA's recommended action exposure level of 1 ppm. While labeling that conveys nutritional or environmental information to consumers is not new in the U.S. seafood market,¹² this is the first time labels have been used in an attempt to understand how individual consumers respond to specific information about mercury contamination in their potential purchases. More investigation will be needed to determine if this labeling scheme will lead to significant reductions in mercury exposure among consumers, and ultimately to improvements in public health. A mirror-image of the Safe Harbor market approach occurred in 1991 when California began requiring that Gulf of Mexico oysters

¹² An early example of labeling in the U.S. seafood market is the "dolphin-safe" tuna label. A more recent example is the law requiring retailers to provide country-of-origin information for seafood they sell, as well as whether the product is wild or farm-raised.

be labeled with a warning about potential contamination from *Vibrio vulnificus* (Keithly and Diop 2001). In that instance, consumer reaction to the oyster label led to significantly depressed market prices for Gulf oysters, not only in California but across the nation (Keithly and Diop 2001). Given this experience, it is plausible that market prices for seafood products not labeled as low mercury could similarly fall, an outcome that would heighten Willett's (2005) concern that the poor may ultimately be exposed to higher contaminant levels as a result of public dissemination of contamination information. Perhaps equally likely, however, is that the Safe Harbor fish will either be awarded a price premium by consumers, or that consumers on the whole will disregard the label.

As previously suggested, information and perception play important roles in consumption decisions. Public health gains could be realized through the design and implementation of a focused education and information campaign. Advisory information needs to be presented in ways that are not confusing to the consumer. Consumption guidelines need to be specific. Oceans Alive, a campaign by the Environmental Defense Fund, presents consumption advice based on species and population group.¹³ For example, women and children are advised to avoid consuming swordfish, while men can safely consume one swordfish meal per week. The information is presented in a color-coded manner, and can be printed for easy reference. Education efforts like this show promise, and may reduce the unintended responses to consumption advisories. However, the information needs to be widely available to all fish consumers. Information regarding contaminated seafood is most often disseminated through television or print media, and consequently does not reach all of its target audience. Shimshack, Ward, and Beatty (2005) suggest public transportation advertising and in-store signs as potential methods for improved educational outreach.

¹³ Available from the Oceans Alive Web site: <<http://www.oceansalive.org/eat.cfm?subnav=healthalerts>>.

Although the EPA has striven to reduce the amounts of PBT contaminants in the environment, the persistent and bioaccumulative nature of these contaminants is problematic. Even significant reductions in new releases of PBT pollutants may not result in significant decreases in the level of PBT contaminants present in fish. Alternative approaches may be needed in order to reduce the public's long-term exposure beyond that achieved through voluntary responses to state consumption advisories. One potential alternative that has not yet been considered is to reexamine the way size-based fisheries management is conducted. As currently implemented, most management plans focus on supporting recruitment to the fish stocks and survival to reproductive age by imposing minimum size limits on captured fish. PBT contaminants are bioaccumulative, however, and that often results in a positive relationship between fish size and the levels of contaminant concentration. This paradoxically leads to a situation where management plans designed to protect stocks for ecosystem purposes and for future human use actually increase the levels of PBT exposure experienced by consumers. An alternative would be a more directed, size-based management of contaminated marine fisheries that explicitly accounts for contamination and public health issues when determining optimal harvesting regimes. Intuitively, this approach might require the harvesting of younger, smaller fish with the objective of allowing older, larger fish to serve as both a breeding stock and PBT sink. How this type of management might work needs to be explored within the context of an empirical bioeconomic model that combines population, toxicological, and economic information into the decision-making process. A model such as this could be used to generate policy relevant management suggestions under varying management objectives and ultimately reduce the amount of contaminants reaching consumers.

Recently, increased attention has been given to the connection between the oceans and human health.¹⁴ Humans affect the oceans in many ways and conversely, the oceans affect human health (Sandifer et al. 2004). Knowledge about these connections is continually expanding, and the public health implications are addressed in plans for the Integrated Ocean Observing System (IOOS). Integrated and sustained coastal monitoring efforts such as IOOS hold dramatic potential for reducing the public health impacts that fish consumers face. The IOOS is a multidisciplinary “system of systems that routinely and continuously provides quality-controlled data and information on current and future states of the oceans and Great Lakes from the global scale of ocean basins to local scales of coastal ecosystems (Ocean.US, 2006).” Envisioned as a partnership between state, local, and federal agencies, the private sector, and academia, IOOS is designed to provide decision-makers with timely, necessary information in addressing seven societal goals, including the reduction of public health risks (Nowlin and Malone 1999). As part of the effort to reduce public health risks, the IOOS will aim to establish nationally standardized measures of the risk of illness or injury from exposure to pathogens and toxins, and establish nationally standardized measures of the risk of illness from consuming seafood (Oceans.US 2006). The report, entitled *The First US Integrated Ocean Observing System (IOOS) Development Plan 2006*, addresses the challenge of integrating all current programs and aligning them to current needs through IOOS design and implementation (Ocean.US 2006). As this plan evolves, the integration of many once disparate monitoring systems holds the potential to reduce the public health risks associated with seafood consumption.

¹⁴ Examples include the *Oceanography* 19, no. 2 (2006) special issue on the Oceans and Human Health and *Environmental Health Perspectives* 112, no. 8 (2004).

CONCLUSION

Contamination of U.S. fisheries is a growing threat not only to public health, but also to the economic and ecological viability of many fisheries. The economic impacts of contamination can be staggering, ranging from increased public health costs and direct losses in commercial fisheries to reduced revenues from curtailed recreation and tourism. Current strategies to reduce human exposure are reactive in nature, but the potential for serious loss suggests a greater need for proactive management to prevent contamination. Research is ongoing to develop preventive measures for HABs, and the EPA is working to significantly reduce PBT contaminants in the environment. However, it could be many years before substantial improvements are seen from these efforts. In the meantime, management should be focused on reducing human exposure to contaminants. In particular, management options that reduce perception and bias in decision making, proactively control the contaminant levels that reach the marketplace, and provide for integrated analysis and coordinated action across political boundaries should be considered in an attempt to reduce public health risks from seafood consumption.

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CHAPTER 3: THE POPULATION DYNAMICS OF KING MACKEREL

As currently implemented, most fisheries management plans focus on supporting recruitment to the fish stocks and survival to reproductive age by imposing minimum size limits on captured fish. Given the positive relationship between size and mercury concentration in some fish species, this paradoxically leads to a situation where management plans designed to protect stocks for ecosystem purposes and for future human use actually increase the levels of mercury exposure experienced by consumers. An alternative would be a more directed, size-based management of contaminated marine fisheries that explicitly accounts for contamination when determining optimal harvest regimes. A necessary first step towards modeling potential management strategies is to develop a population dynamics model of the fishery being investigated.

BACKGROUND

Historically, the main priority in fisheries management has been to maintain fish stocks (Grafton et. al 2006) although protecting the economic position of specific groups in the fishery is sometimes a consideration (Anderson 1977). Fishery economists and policymakers have been concerned with control of total catch in order to avoid excessive harvesting of common property resources (Schott 2001). Common management strategies include size, gear and effort restrictions, quotas, closed areas, shorter seasons, and limited entry.

Bioeconomic models provide an integrated approach to evaluate alternative fishery management strategies (Thunberg, Helser, and Mayo 1998). Fishery bioeconomic models combine models of fish biology, or population dynamics, with an economic model of the fishery. The most commonly used models of fish biology in the economic study of commercial fisheries are the lumped-parameter models of Gordon (1954) and Schaefer (1954) and the Ricker (1958) and Beverton-Holt (1957) age-structured models. The lumped-parameter models, also known as

single cohort models, track one age class through time without distinguishing between age classes. A single cohort model, although analytically and empirically more tractable, is unsuited for this study because of the need to explicitly model the variations in contamination and harvestability across age classes. Multiple cohort, or age-structured, models are more applicable for studying many management problems because they track more than one cohort through time and can explicitly distinguish the varying characteristics of each cohort (Schott 2001).

Dynamic age-structured models are the preferred approach to evaluate the impacts of management policies that affect a subset of cohorts, provided that detailed stock information is available (Lee, Larkin and Adams 2000). Recent studies that utilize dynamic age-structured models include Thunberg, Helser, and Mayo (1998), Lee, Larkin and Adams (2000), Bertignac et al. (2000), Pintassilgo and Costa Duarte (2002), Bjørndal, Ussif, and Sumaila (2004), and Kulmala et al (2008). It is from this literature base that a conceptual multiple cohort model was developed for this study, incorporating not only varying contamination characteristics by age/size class, but also temporal and (to some extent) spatial variability in fishing mortalities. Before examining the construction of the population dynamics model, however, it will prove useful to detail the specifics of the king mackerel fishery.

THE FISHERY

King mackerel is a coastal pelagic that is distributed in the western Atlantic and in the Gulf of Mexico and Caribbean Sea, with substantial commercial and recreational catches occurring in U.S. waters. In the southeast U.S., king mackerel is currently managed under the Coastal Migratory Pelagic Species Fishery Management Plan (FMP). The FMP recognizes two stocks for the purpose of management (Gulf migratory stock and Atlantic migratory stock). Management under the two-group model is complicated due to migrations within the Gulf of Mexico group and the mixing that occurs between the Atlantic and Gulf populations during the

winter. The Atlantic migratory stock management area extends from New York to Florida while the Gulf migratory group management area extends from Florida to Texas. For management and assessment purposes, a mixing zone was specified off southeast Florida to assign stock identity to landings captured there (Figure 3.1). The mixing zone boundaries are defined by the Volusia\Flagler County border on the east coast of Florida and the Monroe\Collier County border on the Southwest coast in Florida. Landings taken in this zone from April 1 to November 31 are attributed to the Atlantic stock, while landings taken in this zone from December 1 to March 31 are attributed to the Gulf stock, despite information suggesting that the Atlantic stock likely contributes a significant percentage of winter landings taken there (DeVries et al. 2002, Fable 1990, Patterson et al. 2004, Sutter et al. 1991).

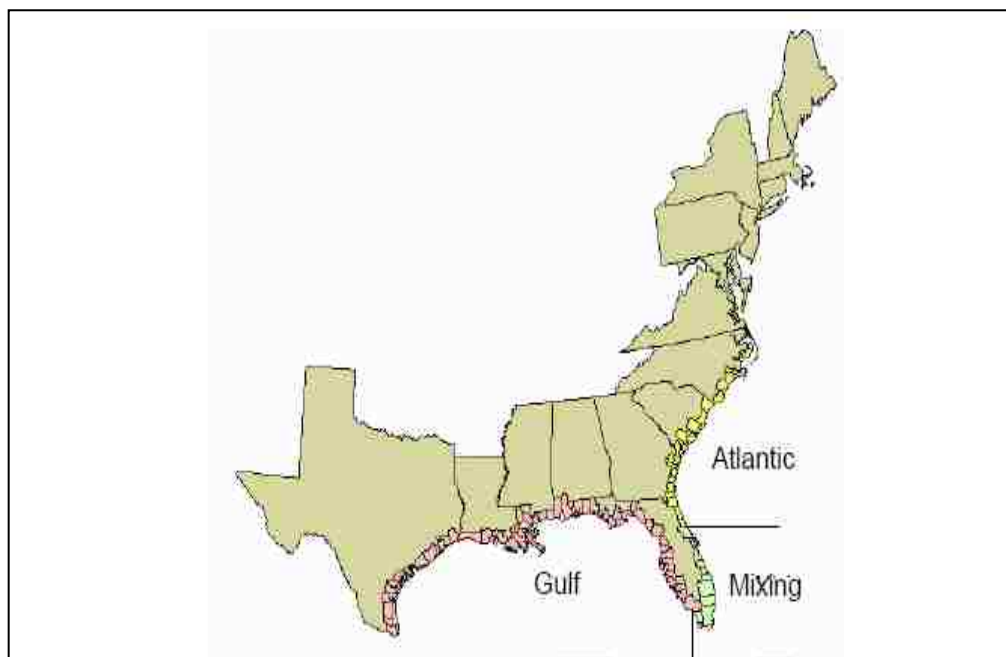


Figure 3.1. Map indicating the Atlantic, Gulf, and Mixing zones for U.S. king mackerel Source: SEDAR 16 2009.

As implied above, king mackerel are managed through a total allowable catch (TAC) calculated for each migratory group and allocated to harvesters based on FMP requirements. Commercial fisheries are typically managed through quotas, possession and trip limits, size

limits, and seasonal closures, while recreational fisheries are typically managed through possession limits and size limits. Limited entry restrictions are in effect for commercial and charter and headboat fisheries. Modifications to TACs and framework adjustments such as trip limits, size limits, and seasonal closures are addressed and documented through regulatory amendments promulgated by the Councils. The most recent framework adjustment for the Gulf Migratory group of king mackerel, approved in 2003, maintained the status quo TAC of 10.2 million pounds with 3.26 million pounds allocated to the commercial sector. The commercial TAC was allocated by geographic zones and gear types, and restricted by trip limits and seasonal closures specific to each zone and gear. The Gulf group king mackerel fishery opens with a new quota every year on July 1. The most recent framework adjustment for the Atlantic Migratory group of king mackerel was approved in 2000. It increased the TAC to 10.0 million pounds, with 3.71 million pounds allocated to the commercial fishery. Commercial fisheries are restricted by a 3,500 pound trip limit from New York to the Brevard\Volusia County line in Florida, 50 fish from that line south to the Dade\Monroe County line in Florida, and 1,250 pounds in Monroe County.¹⁵ Regulations for both migratory groups currently require a minimum size limit of 24 inches for each fish harvested.

The majority of commercially caught king mackerel are landed off the coast of Florida in the mixing zone. Figure 3.2 illustrates the historical landings of king mackerel broken down by the area of landing. While commercial landings of king mackerel have fallen from their early 1980s levels, the gears used to harvest king mackerel have changed in importance over time. For the Gulf of Mexico, gillnet landings previously accounted for more than half of the commercial harvest, but in recent years have accounted for only ten to twenty percent of the landings

¹⁵ The current management routine for king mackerel is complex. In addition to the changing regulatory boundaries already discussed, trip limits for some areas are defined in terms of numbers of fish while others are defined in terms of catch weight in pounds.

(primarily due to increased restriction on gillnet use because of its nonselective nature) (SEDAR16 2009). As shown in Figures 3.3 and 3.4, hook and line gear now accounts for the majority of commercially landed king mackerel in U.S. waters.

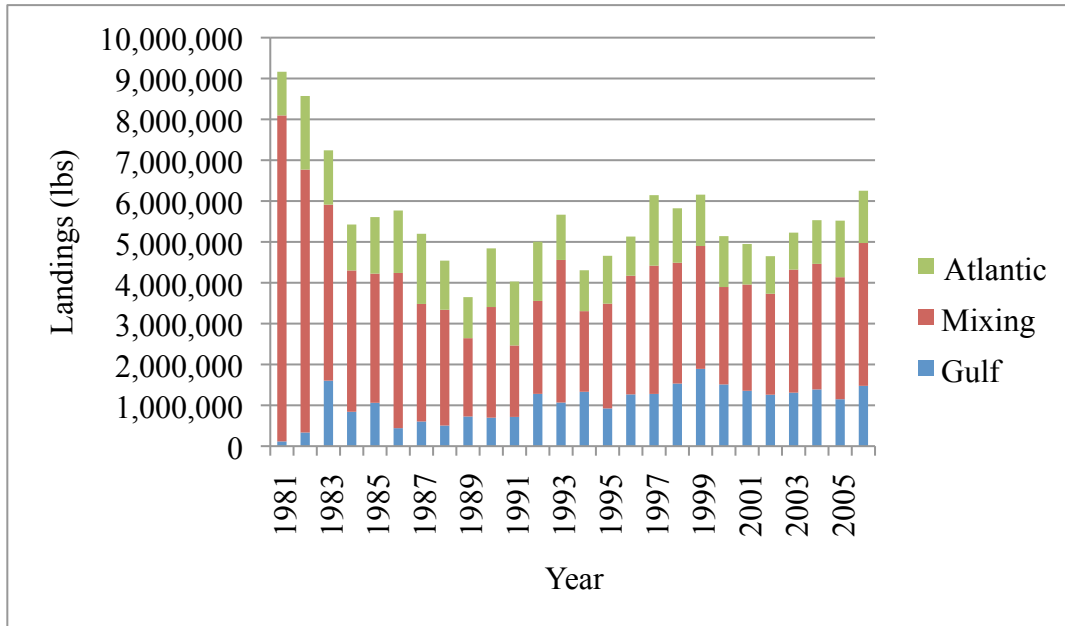


Figure 3.2. U.S. commercial landings in pounds by zone. Source: SEDAR16 2009

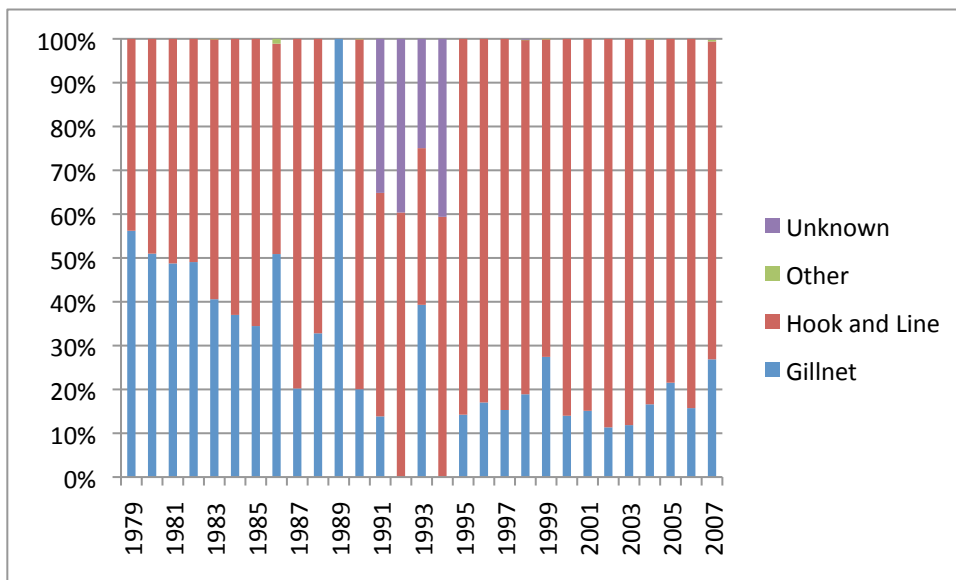


Figure 3.3. Percentage of total U.S. king mackerel landings by gear for Gulf of Mexico stock. Source: Ortiz 2008.

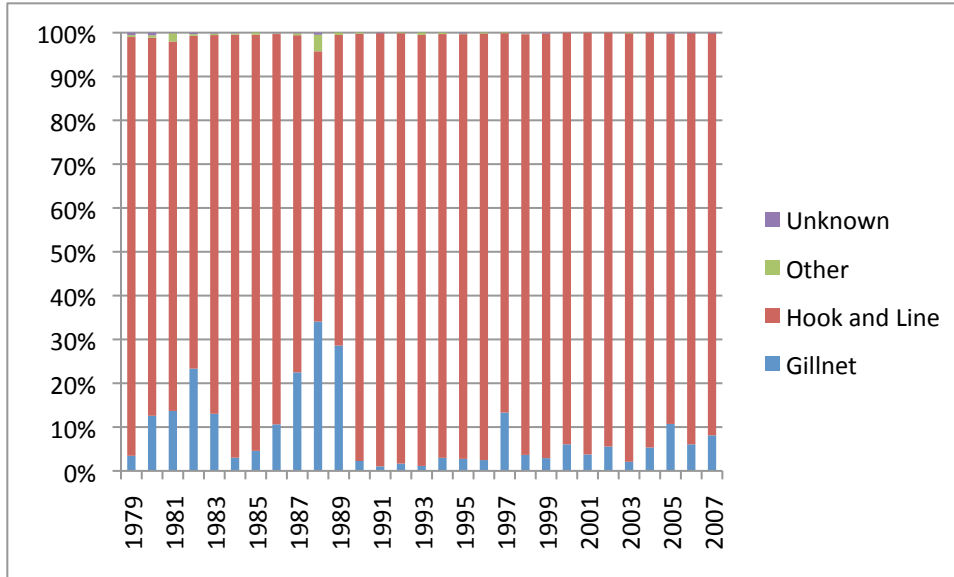


Figure 3.4. Percentage of total U.S. king mackerel landings by gear for Atlantic stock
Source: Ortiz 2008.

The choice of king mackerel for this study was prompted by a number of factors. Each of the mackerel fisheries is considered to be biologically distinct with the exception of the mixing interface off of south Florida. Both stocks are currently considered to be recovered from overfishing and are managed through a TAC that divides the harvestable stock between recreational and commercial interests (SEDAR16 2009). Given the current level of management intervention, these fisheries are relatively well documented, both with respect to their biological characteristics and incidence of mercury contamination. Mercury levels in king mackerel harvested off Florida’s Atlantic coast and in the Gulf of Mexico ranged from less than 0.5 ppm for individuals with fork lengths of 600 mm to over 3.0 ppm for individuals with fork lengths approaching 1.2 meters (Axelrad et al. 2004). Similarly, Atlantic king mackerel off the coast of Georgia, South and North Carolina were found to contain mercury levels as high as 3.5 ppm (Bender 2003). Given the U.S. Food and Drug Administration’s recommended current action exposure level of 1.0 ppm (U.S. FDA 2001), these levels of contamination have prompted the issuance of consumption advisories by most of the states bordering the Atlantic Ocean and Gulf

of Mexico. Additionally, the Southeast Data, Assessment, and Review (SEDAR) conducted a stock assessment of the Atlantic and Gulf of Mexico migratory groups of king mackerel in 2008. This current biological data is available for use in constructing a bioeconomic model. In addition, the current active management of the fishery provides real-world relevancy for the project and the opportunity to demonstrate how public health risks can be incorporated into management strategies to minimize mercury exposure.

POPULATION DYNAMICS

“Fish are born, they grow, they reproduce and they die – whether from natural causes or from fishing. That’s it. Modelers just use complicated (or not so complicated) math to iron out the details.” – Andrew B. Cooper in *A Guide to Fisheries Stock Assessment*

While the above quote is a simplification, it touches on the important features that the age-structured fishery population dynamics model must capture. An age-structured population dynamics model includes three basic components: recruitment, mortality and individual fish growth (Quinn and Deriso 1999). This section presents the equations for a discrete time biological model of the Atlantic and Gulf of Mexico king mackerel fisheries that reflect the dynamics of the stocks as a result of mortality, reproduction, and growth. A list of all symbols used in the model is given in Appendix B.

Population Dynamics

The king mackerel population is distributed in age classes, beginning at age 0, with the time step being one year. The terminal group is age 11, and is calculated as an accumulator age class where all fish age 11 years and older are pooled together.¹⁶ The year-to-year change in the number of fish in a cohort, or age class, depends on instantaneous fishing and natural mortality rates. Natural mortality refers to all deaths that are not a result of fishing, including predation,

¹⁶ The use of an accumulator age class, often called a plus group, is common in fisheries models. Scientists define a plus group based on the ability to predict age from length, which becomes more difficult in older fish that may not exhibit much change in length as they age, or based on the age above which very few individuals appear in the data set (Cooper 2006).

pollution, and senility, while fishing mortality refers to removals from the stock due to harvesting. The time-dynamics of the cohorts are modeled using the exponential decline function:

$$(3.1) \quad N_{s,a,t} = \begin{cases} N_{s,a-1,t-1} e^{-Z_{s,a-1,t-1}} & \text{for } a = 1, 2, \dots, 10 \\ N_{s,10,t-1} e^{-Z_{s,10,t-1}} + N_{s,11,t-1} e^{-Z_{s,11,t-1}} & \text{for } a = 11 \end{cases}$$

$$(3.2) \quad Z_{s,a,t} = M_{s,a} + F_{s,a,t}$$

where $N_{s,a,t}$ is the number of fish of age a at the beginning of year t in stock s (s =Atlantic, Gulf), $Z_{s,a,t}$ is the total instantaneous mortality rate of fish of age a during year t for stock s , $M_{s,a}$ is the instantaneous rate of natural mortality on fish of age a for stock s , and, $F_{s,a,t}$ is the total instantaneous fishing mortality rate of fish of age a during year t for stock s . The number of fish in each cohort in the initial year, denoted $N_{s,a,0}$, are assumed known at the beginning of a simulation.

In addition to accounting for losses due to natural and fishing mortality, it also necessary to account for recruitment of new fish to the stock. Recruitment is often assumed to be a function of the spawning stock, or the fish in a stock that are old enough to reproduce. In particular, the commonly used Beverton and Holt (1957) stock recruitment function relates the number of recruits in a year to the previous year's spawning stock fecundity:

$$(3.3) \quad N_{s,0,t} = \frac{\alpha_s SSF_{s,t-1}}{(\beta_s + SSF_{s,t-1})}$$

$$(3.4) \quad SSF_{s,t} = \sum_{a=1}^{11} Mat_{s,a} N_{s,a,t} Fec_{s,a} Fem_{s,a,t}$$

where $N_{s,0,t}$ is the number of recruits (age-0 fish) in year t for stock s , $SSF_{s,t-1}$ is the spawning stock fecundity in year $t-1$ for stock s , $Mat_{s,a}$ is the proportion of age a fish in stock s that are mature enough to spawn, $N_{s,a,t}$ is the number of fish of age a at the beginning of year t in stock s ,

and $Fec_{s,a}$ is the fecundity or number of eggs produced by a fish of age a in stock s , and $Fem_{s,a,t}$ is the proportion of age a fish in year t from stock s that are female. α_s and β_s are positive recruitment function parameters for the stock s .¹⁷

The model also tracks the biomass, or total weight of the stock. Biomass is important in fisheries models because it is often used to determine the status of a stock. It is calculated by taking the number of fish in each age class, multiplying by the weight at age, and then summing across ages as follows:

$$(3.5) \quad B_{s,t} = \sum_{a=0}^{11} N_{s,a,t} \cdot W_{s,a,t}$$

where $B_{s,t}$ is the biomass of stock s in year t , $N_{s,a,t}$ is the number of fish of age a at the beginning of year t in stock s , and $W_{s,a,t}$ is the average weight of an individual fish of age a in year t for stock s .

Total removals from the stock are accounted for in equation 3.1, but it is also necessary to separate the removals due only to fishing. Catch is modeled as a function of fishing mortality, total mortality, and numbers of fish:

$$(3.6) \quad CN_{s,a,t} = \frac{F_{s,a,t} \cdot N_{s,a,t}}{Z_{s,a,t}} (1 - e^{-Z_{s,a,t}})$$

where $CN_{s,a,t}$ is the number of age a fish caught in year t from stock s , $F_{s,a,t}$ is the total instantaneous fishing mortality rate of fish of age a during year t for stock s , $N_{s,a,t}$ is the number of fish of age a at the beginning of year t in stock s , and $Z_{s,a,t}$ is the total instantaneous mortality rate of fish of age a during year t for stock s . It is also useful to have a measure of the total weight of the fish caught. This is modeled as:

¹⁷ The Beverton-Holt recruitment function is often reparameterized for estimation and interpretation purposes as illustrated in Haddon (2001). In the form of equation 3.3, the parameter α is the maximum number of recruits produced and β is the spawning stock needed to produce an average recruitment equal to half of the maximum, although their interpretation is not vital to this research.

$$(3.7) \quad C_{s,t} = \sum_{a=0}^{11} CN_{s,a,t} \cdot W_{s,a,t}$$

where $C_{s,t}$ is the total weight of all fish caught in year t from stock s , $CN_{s,a,t}$ is the number of age a fish caught in year t from stock s , and $W_{s,a,t}$ is the weight of an age a fish in year t from stock s . Equations 3.6 and 3.7 account for all removals of the stock due to fishing. This includes both commercial and recreational king mackerel fishing as well as dead recreational discards and bycatch from the shrimp (and other) fishing industry. While this measure of fishing mortality is vital for tracking the overall dynamics of the stock, it is also necessary to explicitly model the commercial catch. To accomplish this, total fishing mortality F is partitioned into commercial fishing mortality and the remaining fishing mortality due to recreational fishing and bycatch:

$$(3.8) \quad F_{s,a,t} = FComm_{s,a,t} + FRem_{s,a,t}$$

where $F_{s,a,t}$ is the total instantaneous fishing mortality rate of fish of age a during year t for stock s , $FComm_{s,a,t}$ is the instantaneous fishing mortality rate of fish of age a resulting from commercial king mackerel fishing activity during year t for stock s , and $FRem_{s,a,t}$ is the remaining instantaneous fishing mortality rate of fish of age a during year t for stock s . $FRem_{s,a,t}$ accounts for aggregate stock removal resulting from the recreational king mackerel fleet, including dead discards, and bycatch of king mackerel occurring in fishing activities targeting other species.

The partitioned fishing mortality can be used to model commercial catch. Substituting equation 3.8 into equation 3.6 into yields:

$$(3.9) \quad CN_{s,a,t} = \frac{(FComm_{s,a,t} + FRem_{s,a,t}) \cdot N_{s,a,t} (1 - e^{-Z_{s,a,t}})}{Z_{s,a,t}}$$

Rearranging equation 3.9 allows the partition of total catch into that of commercial catch plus the remaining catch from recreational catch and bycatch:

$$(3.10) \quad CN_{s,a,t} = \frac{FComm_{s,a,t} \cdot N_{s,a,t}}{Z_{s,a,t}} (1 - e^{-Z_{s,a,t}}) + \frac{FRem_{s,a,t} \cdot N_{s,a,t}}{Z_{s,a,t}} (1 - e^{-Z_{s,a,t}})$$

Equations 3.11 and 3.12 then give the commercial catch in numbers and weight, respectively:

$$(3.11) \quad CommCN_{s,a,t} = \frac{FComm_{s,a,t} \cdot N_{s,a,t}}{Z_{s,a,t}} (1 - e^{-Z_{s,a,t}})$$

$$(3.12) \quad CommCW_{s,t} = \sum_{a=0}^A CommCN_{s,a,t} \cdot W_{s,a,t}$$

where $CommCN_{s,a,t}$ is the number of age a fish commercially caught in year t from stock s and $CommCW_{s,a,t}$ is the total weight of the commercial catch in year t from stock s .

The population dynamics parameters were obtained from the latest king mackerel stock assessment as outlined in the SEDAR 16 Stock Assessment Report (SEDAR16 2009). The stock assessment makes use of Virtual Population Analysis (VPA) to estimate the yearly numbers of fish in each age class ($N_{s,a,t}$) and the annual fishing mortality at age ($F_{s,a,t}$). Tables of these parameters are included in Appendix B. VPA is a commonly used modeling technique that reconstructs historical fish numbers at age through backward projections. VPA assumes that catch at age is known with certainty for all years covered by the stock assessment and requires “tuning” through the incorporation of relative indices of abundance during the estimation process (Butterworth and Rademeyer 2008)¹⁸. While classical VPA is not a statistical analysis, it serves as a basis for the adaptive framework VPA (ADAPT) that is used in the king mackerel stock assessment (Lassen and Medley 2001). ADAPT, introduced by Gavaris (1988), is one of the most popular tuning models and involves the minimization of the sum-of-squares over any number of indices of abundance to find best-fit parameters (Lassen and Medley 2001). The VPA base model parameters were used for the Atlantic stock, while the VPA final model results were

¹⁸ Tuning a model involves adjusting parameter estimates to minimize differences between predicted population catches and observations from indices of population abundance (NRC 1998).

used for the Gulf stock.¹⁹

It should be noted that because of management definitions, the stock assessment used fishing year rather than calendar year. The fishing year in the Gulf runs from July 1 to June 30 of the following year while in the Atlantic it runs from April 1 to March 31 of the following year. For notational purposes in this study, the fishing year 1981 refers to the fishing season from April 1, 1981 to March 31, 1982 for the Atlantic stock and the season from July 1, 1981 through June 30, 1982. In addition, it must be noted that the stock assessment (upon which this study is based) was carried out under the assumption that fifty percent of the catch in the mixing zone during the winter months (November 1-March 31) belonged to the Gulf stock and fifty percent to the Atlantic stock. The catch-at-age information used as an input into the ADAPT model was constructed under this assumption, and the resulting output therefore accounts for this assumption. Given that the mixing is mostly limited to southern Florida, it was not possible to explicitly model the migrations without assuming that the mixing could occur anywhere throughout the Gulf and Atlantic regions (SEDAR16 2009).

The remaining population dynamics parameters needed for the model were used as inputs in the VPA analysis and were taken from the Final Stock Assessment Report (SEDAR16 2009). Natural mortality at age for each stock ($M_{s,a}$) is given by a declining Lorenzen (1996) function of age. These natural mortality parameters, along with parameters for maturity at age ($Mat_{s,a}$) and fecundity at age ($Fec_{s,a}$), are given in Table 3.1. Given the lack of availability of more detailed information, it was assumed that 50% of the fish in each age class during each year are female for both stocks. The Beverton-Holt stock recruitment parameter α_s and β_s are given for each stock in Table 3.2. Weights-at-age were developed in five year blocks for the Atlantic and Gulf

¹⁹ This is only because a final model was not presented for the Atlantic stock in the latest stock assessment report. It is worth noting that the differences in output from the Gulf base and final models are small.

stocks and are presented for years 1981-2006 in Appendix B.

Table 3.1. Biological functions for the Atlantic and Gulf of Mexico king mackerel stocks

Source: SEDAR16 2009

Table 3.2. Beverton-Holt Stock Recruitment Parameters

Stock	α	β
Atlantic	3.46E+06	6453
Gulf	7.78E+06	11721

Source: SEDAR16 2009

While most of the required population dynamics parameters were easily obtained from the 2009 stock assessment report, annual commercial fishing mortality at age for each stock ($FComm_{s,a,t}$) was not readily available. The fishing mortality for a particular fleet²⁰ can be separated into an age effect (selectivity of the fishery) and a year effect (intensity of the fishing mortality) (Fournier and Archibald 1982; Deriso et al. 1985; Myers and Quinn 2002). Ideally, determining fishing mortality at age for the commercial king mackerel fleet requires information on selectivity at age and annual fishing mortality at maximum selectivity. While this information

²⁰ In fishing, a fleet is simply an aggregate of fishing vessels. It may refer to all vessels engaged in the harvesting of a particular species such as king mackerel, all vessels using a particular gear, or all vessels from a particular port, region, or country.

is available in stock assessments for some species, it is not for king mackerel. Consequently, an alternative method for determining commercial fishing mortality had to be devised.

The 2009 stock assessment report did not provide any information about the overall catch-at-age breakdown for the commercial king mackerel fishery, but partial catches at age were given for several of the tuning indices. For the commercial fisheries, partial catches at age were given for the Gulf of Mexico logbook index and the North Carolina Trip Ticket index²¹. Assuming that catches from the logbooks and trip tickets are accurate representations of the fishing activity throughout the Gulf and Atlantic, then that data can be used to determine the commercial catch proportion by age for each stock. Given that the total commercial catch for each stock is known, this information can be combined with weights at age for each stock to generate an estimate of the total number of king mackerel commercially caught from each stock:

$$(3.13) \quad CommCN_{s,t} = \frac{CommCW_t}{\sum_{a=0}^{11} \rho_{s,a,t} W_{s,a,t}}$$

where $CommCN_{s,t}$ is the total number of commercially caught fish in year t from stock s and $CommCW_{s,t}$ is the total weight of the commercial catch in year t from stock s , $\rho_{s,a,t}$ is the proportion of age a fish commercially caught from stock s during year t , and $W_{s,a,t}$ is the weight of an age a fish in year t from stock s . Commercial fishing mortality at age can then be calculated as:

$$(3.14) \quad FComm_{s,a,t} = \frac{Z_{s,a,t} \rho_{s,a,t} CommCN_{s,t}}{N_{s,a,t} (1 - e^{-Z_{s,a,t}})}$$

where $FComm_{s,a,t}$ is the instantaneous fishing mortality rate of age a fish resulting from commercial fishing activity during year t for stock s , $Z_{s,a,t}$ is the total instantaneous mortality rate

²¹ The North Carolina Trip Ticket index was chosen over the Atlantic logbook index for the Atlantic VPA model by the SEDAR assessment workshop.

of age a fish during year t for stock s , $\rho_{s,a,t}$ is the proportion of age a fish commercially caught from stock s during year t , $CommCN_{s,t}$ is the total number of commercially caught fish in year t from stock s , and $N_{s,a,t}$ is the number of age a fish at the beginning of year t in stock s . Although this may not completely reflect the true catch at age distribution of the stock, it should be reasonably close.²²

MODEL VALIDATION

Once constructed, a population dynamics model and its simulation output needs to be validated before it is used for policy research and analysis. In this particular case, it is important to see how well the parameterized model tracks the dynamics of the modeled system by comparing the simulated results for total landings and biomass with those reported in the SEDAR16 stock assessment (which, for the purposes of this study, are assumed to be the actual real-world values). To accomplish this, simulations were generated for the time period covered in the stock assessment, fishing years 1981-2006.

The results of population dynamics simulations for Atlantic king mackerel are shown in Figures 3.5 and 3.6. Although the simulated biomass generally matched the overall trend of the SEDAR16 data (Figure 3.5), the simulated values diverged early in the series due to an early turning point error in 1985 and a few large percent changes year-over-year (Table 3.3). These differences between the simulation and the SEDAR16 data were particularly acute in the period 1985-1987, a situation which permanently put the simulation on a lower track even though subsequent deviations between the simulations and the SEDAR16 data tended to cancel out over time. Overall, the biomass simulation experienced a 28 percent turning point error (TPE) rate (7 out of 26 observations) and an average percent movement error (APME) year-over-year of 8.27

²² Given that there is no way to know with certainty the true catch distribution, this process at a minimum allows a baseline (if not the true baseline) to be determined against which alternative harvesting patterns can be compared.

percent. Similar patterns were observed for the overall Atlantic landings (i.e., total removals from the stock including bycatch and dead discards), as the simulated results closely track the SEDAR16 data, but are always lower (Figure 3.6). In this latter case, a lower TPE rate (8 percent) and a modest APME year-over-year (4.2 percent) resulted in simulated landings being a better match to the SEDAR16 data (Table 3.3).

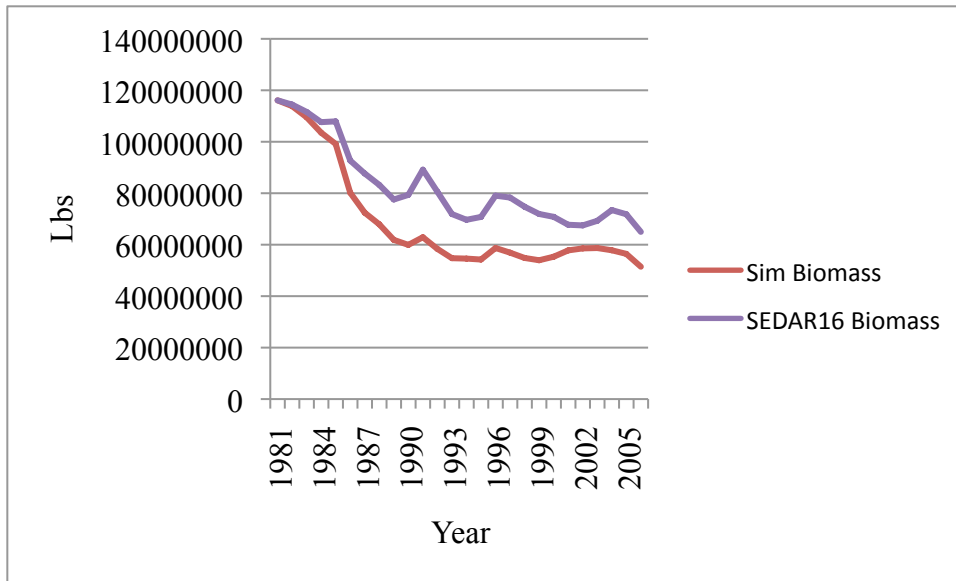


Figure 3.5: Actual and simulated biomass for the Atlantic stock (1981-2006)

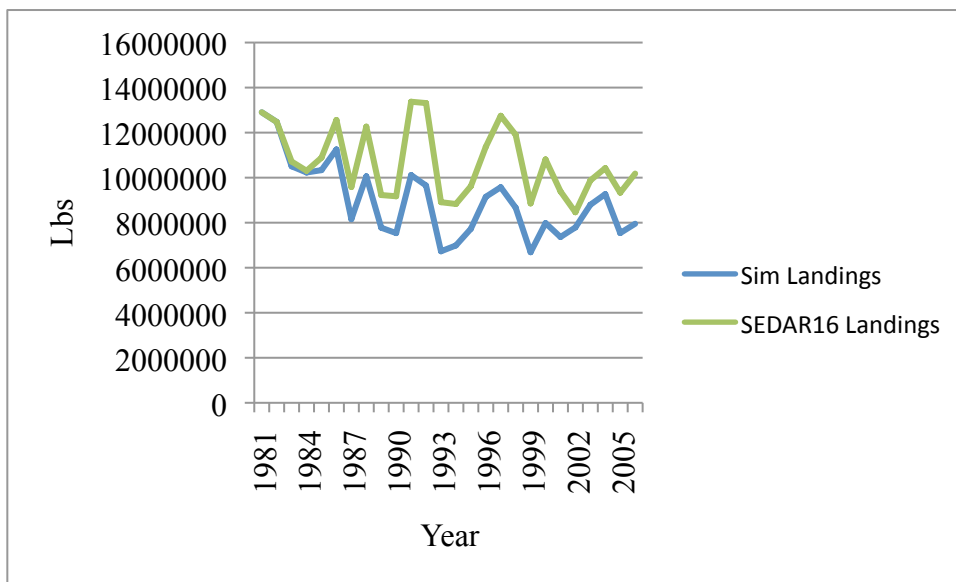


Figure 3.6: Actual and simulated landings for the Atlantic stock (1981-2006)

A comparison of the simulated population dynamics versus the SEDAR16 data for the Gulf of Mexico stock are shown in Figures 3.7 and 3.8. As compared to the Atlantic, the Gulf simulated biomass does not visually appear to follow the actual trend as well, and is higher than the actual biomass through most of the simulation time horizon. The calculated TPE rate and APME year-over-year, however, are both lower for the Gulf of Mexico simulations than for the Atlantic simulations (Table 3.4). The simulated Gulf landings pattern visually tracks the SEDAR16 landings data quite well and perhaps better than in the Atlantic simulations, especially in the latter part of the simulated time horizon (Figure 3.5), although the TPE rate and APME year-over-year were significantly higher for the Gulf simulations. This discrepancy highlights the fact that the raw TPE rate and APME values can, while giving an indication of the validity of the simulation, be somewhat misleading if the goal of the simulation is to track the general evolution of a system through time. At the same time, the generally high values for TPE rate and APME for both the Atlantic and the Gulf, especially early in the time horizon of the simulations, calls for an explanation.

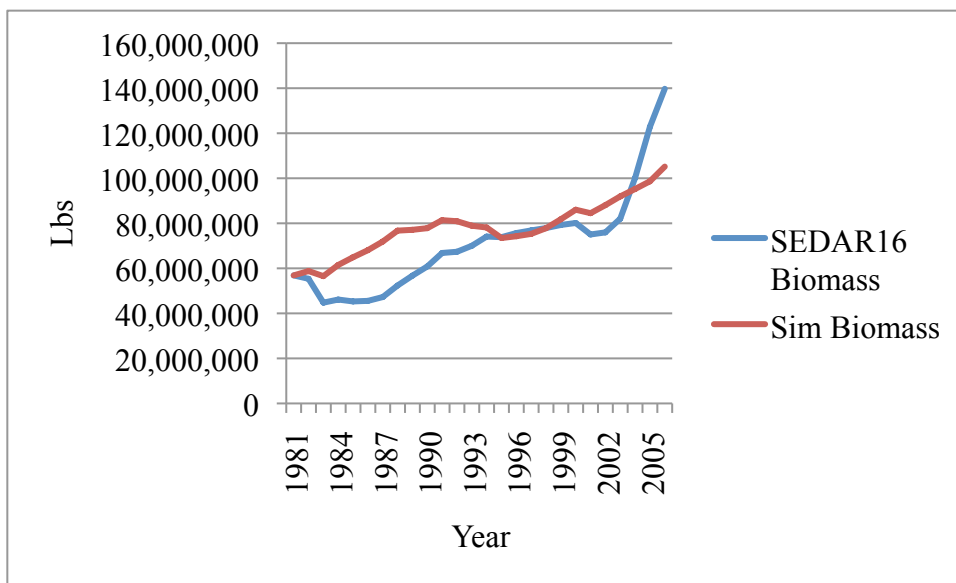


Figure 3.7: Actual and simulated biomass for the Gulf stock (1981-2006)

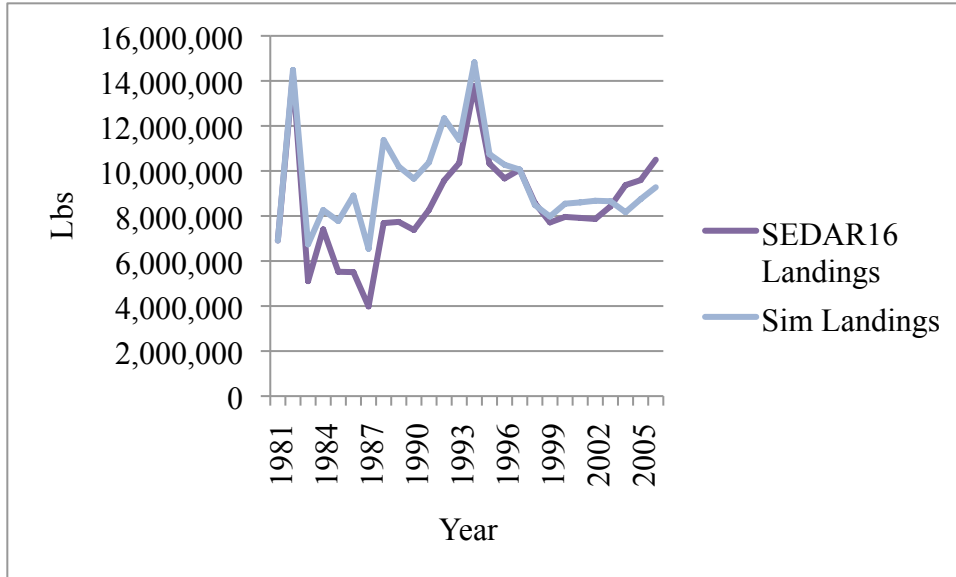


Figure 3.8: Actual and simulated landings for the Gulf stock (1981-2006)

Table 3.3. Turning point errors (TPE) and average percent movement errors (APME) for Atlantic stock simulations as compared to data contained in SEDAR16.

Year	Simulated Biomass TPE	Simulated Biomass APME (percent)	Simulated Landings TPE	Simulation Landings APME (percent)
1982		3.33		0.02
1983		6.30		1.72
1984		9.03		1.21
1985	X	4.52		4.60
1986		33.22		6.50
1987		15.13		3.80
1988		11.25		4.67
1989		15.72		2.07
1990	X	5.48		2.49
1991		7.39		11.57
1992		16.85		4.14
1992		16.93		2.80
1994		3.39	X	4.75
1995	X	2.26		1.63
1996		3.42		0.22
1997		4.02		7.49
1998		8.17		2.86
1999		5.39		2.78
2000	X	0.99		2.74
2001	X	0.10		5.38
2002	X	0.96	X	15.69
2003		2.40		2.81

2004	X	7.51		0.16
2005		4.65		8.19
2006		18.37		3.62
Average	28 percent	8.27	8 percent	4.20

Table 3.3 continued

Table 3.4. Turning point errors (TPE) and average percent movement errors (APME) for Gulf of Mexico stock simulations as compared to data contained in SEDAR16.

Year	Simulated Biomass TPE	Simulated Biomass APME (percent)	Simulated Landings TPE	Simulation Landings APME (percent)
1982	X	6.05		2.93
1983		15.28		10.78
1984		5.71		22.40
1985	X	7.38		19.51
1986		4.30	X	14.83
1987		1.90		1.05
1988		4.12		18.92
1989		7.89	X	11.06
1990		6.23		0.66
1991		5.29		4.63
1992	X	1.38		3.18
1993	X	6.41	X	15.87
1994	X	6.86		2.80
1995		5.61		2.46
1996		1.40		1.96
1997		0.14	X	6.11
1998		1.98		0.93
1999		3.35		4.05
2000		4.07		3.98
2001		4.58	X	1.22
2002		2.99	X	1.46
2003		3.59	X	7.34
2004		18.49	X	16.88
2005		19.10		4.76
2006		7.12		3.42
Average	20 percent	6.05	32 percent	7.33

First, the specification of a recruitment function is hampered by the paucity of data, as evidenced by the relationships used in the final SEDAR16 stock assessments (Figure 3.9). While the form of the relationship is conceptually attractive, it cannot be confirmed from the data.

Secondly, the spawning stock fecundity relationship in equation 3.14 assumes that 50% of the fish in each age class are female, and that this ratio is constant over time. If this assumption is incorrect, then the spawning stock fecundity will be incorrect, ultimately leading to recruitment estimations that diverge from those reported in SEDAR16. Another issue is that even if spawning stock fecundity is accurate, the maximum recruitment given by the estimated Beverton-Holt relationship turned out to be lower in some years than that reported and used by the SEDAR16 assessments. Unfortunately, the specification and estimation of recruitment functions is a problem that often plagues population dynamics models because data on stock and recruitment tend to be highly variable due to intrinsic variability in factors governing survival and measurement errors in estimates of recruitment and the spawning stock that generates it (Thunberg, Helser, and Mayo 1998).²³

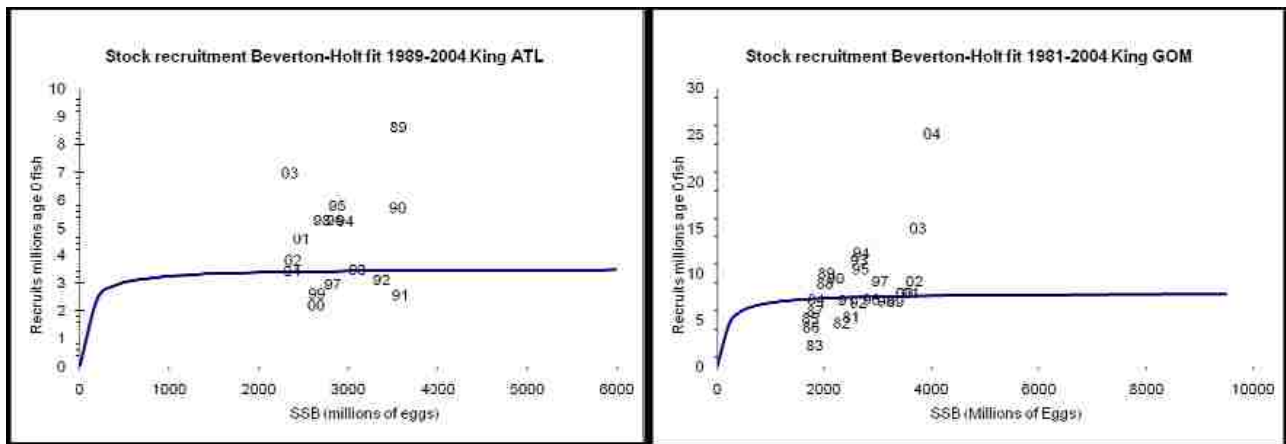


Figure 3.9. Beverton-Holt stock recruitment functions as used in the SEDAR 16 stock assessments (reproduced from SEDAR 16, March 2009).

²³ Some studies rely on a vector of assumed recruitment rather than specifying and estimating a particular functional form. This approach could have been used here, as it essentially was the approach taken in SEDAR16. However, given that the model constructed in this study is designed for forward simulation, future recruitment is not known previous to the evolution of the system over time. Another approach that could have been used is to assume constant recruitment, usually set to average recruitment over some time span. The advantage to using a recruitment function rather than constant recruitment is that if the stock were to become depleted to the point where the spawning stock is severely impacted the model would be able to account for the resulting loss in recruitment.

As indicated in the discussions above, the simulation model generally tracks the SEDAR16 data over time, but there are some instances where there are significant divergences, especially in the early years of the time horizon. Although the entire time period could be used, the remainder of this study will focus on the period 1999 through 2006 for a couple of reasons. First, 1999 was the year in which the minimum size limit for king mackerel was increased to 24 inches fork length. Secondly (and as described previously), by the late 1990s the king mackerel fishery was dominated by hook and line gear. In addition, the trip ticket data and logbook index data used in the extrapolation of commercial fishing mortality (as described earlier) appear to be more representative of the fishery in the later years of its collection.²⁴ Taken together, the relative stability in harvest requirements, gear use, and underlying data collection techniques suggest that simulations focusing on the 1999 to 2006 time period should be better representations of the actual dynamics in the real system.

Figure 3.10 and 3.11 show the simulated and actual commercial catch values for the Gulf and Atlantic stocks, respectively, over the 1999-2006 time span. Visually, the simulated values appear to closely track the pattern of the observed catches, although this is not surprising given the manner in which the commercial fishing mortality was constructed. Table 3.5 presents the turning point error analysis for this simulation. Overall, the commercial simulation for each stock experienced a 14 percent TPE rate (1 out of 7 observations) APME year-over-year of 4.74 percent for the Atlantic stock and 3.22 percent for the Gulf stock. Given these results, over the shortened time frame of 1999-2006, the simulated results adequately capture the relevant features of the system.

²⁴ The approach outlined for determining commercial fishing mortality occasionally yielded commercial fishing mortality rates at a given age higher than the total fishing mortality rates for that same age. For the Atlantic, this did not occur after 1986, and for the Gulf not after 1992.

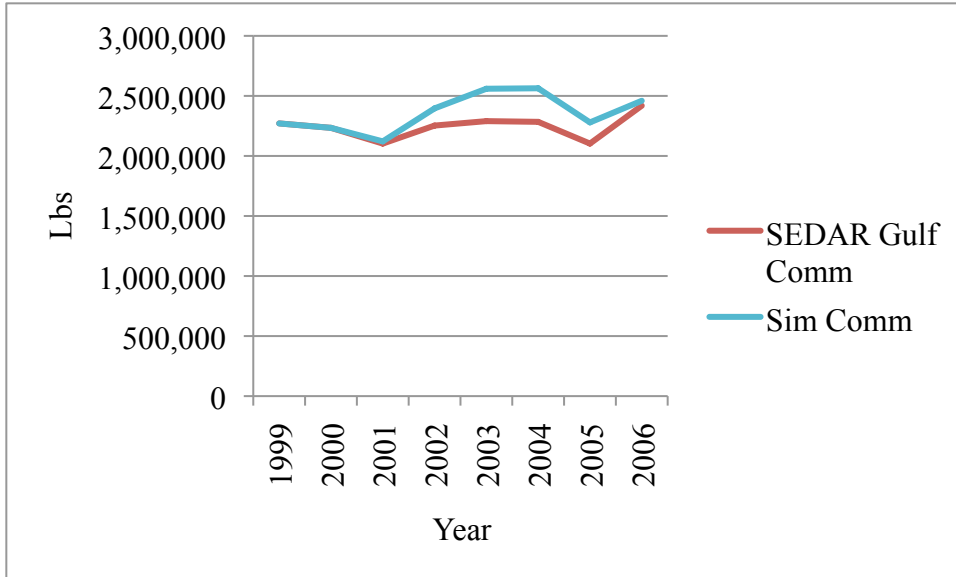


Figure 3.10: Actual and simulated commercial catch for the Gulf stock (1999-2006)



Figure 3.11: Actual and simulated commercial catch for the Atlantic stock (1999-2006)

Table 3.5. Turning point errors (TPE) and average percent movement errors (APME) for Atlantic and Gulf of Mexico commercial harvesting simulations as compared to data contained in SEDAR16.

Year	Simulated Atlantic Commercial Harvest TPE	Simulated Atlantic Commercial Harvest APME (percent)	Simulated Gulf Commercial Harvest TPE	Simulated Gulf Commercial Harvest APME (percent)
2000		0.01		0.02
2001		0.50		0.82
2002	X	6.47		5.82
2003		6.14		5.19
2004		7.01	X	0.41
2005		9.22		3.17
2006		3.71		7.09
Average	14.3 percent	4.72	14.3 percent	3.22

CONCLUSION

This chapter presented the development and implementation of an age-structured population dynamics model for king mackerel in the Atlantic and Gulf of Mexico. The model relies heavily on both inputs used in the stock assessment VPA process, and the VPA output estimates of abundance and fishing mortality at age. The model is important to understanding how the king mackerel stocks have changed over time in response to changes in fishing pressure. Simulation runs and subsequent validation calculations for total landings and biomass indicated a reasonable fit over the historical time period of 1981-2006. In order to examine commercial landings, the simulation time frame was shortened to 1999-2006, a time period that more accurately reflects the current fishery in terms of regulations and gear structure. Model tracking of simulated versus actual commercial landings was quite good over that time period, and is more than adequate for use in future applied research of the king mackerel stocks. Improvements to the model could be made by incorporating a stochastic error term to the recruitment function. Alternately, a new recruitment function could be estimated that relies on spawning stock biomass

rather than fecundity, eliminating the need to make assumptions concerning the sex ratio of the stock. The model could also be combined with an economic model to form a bioeconomic model for investigating both the biological and economic impacts of fishery regulations and policy. The population dynamics model, or the subsequent bioeconomic model, could also be linked with age-structured mercury concentration information to create a model that could be used to investigate alternative management scenarios aimed at reducing consumer exposure to mercury.

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CHAPTER 4: A BIOECONOMIC MODEL INCORPORATING MERCURY CONTAMINATION

INTRODUCTION

As an exercise in data synthesis and interpretation, the development of population dynamics models has been important to the understanding of how fish stocks change over time, particularly in response to fishing pressure. Population dynamics models, however, have little or no economic content, and thus cannot by themselves be used to guide management when the policy making process requires that the state of the fishing industry be considered. To include the needed human dimension, this chapter begins by adding an economic framework to the previously developed age structured population dynamics model, thereby forming a bioeconomic model for king mackerel. The output of the bioeconomic model is then linked to a mercury concentration model through a fish-size/mercury relationship. To the author's knowledge, this is the first time that mercury has been incorporated into a bioeconomic fishery model. Collins, Pascoe, and Whitmarsh (2003) incorporated pollution externalities into a bioeconomic framework to examine management response to pollution in a fishery, but focused primarily on the economic damages from an acute pollution event affecting the industry via shellfish harvesting closures. In contrast, the current research incorporates a pollution externality that focuses on chronic contamination of the fish themselves, rather than the waters the fish are caught in. Even if the levels of mercury in a given waterbody are below closure levels, concentrations of methylmercury in large fish can exceed that of the surrounding water by a million-fold (U.S. EPA 2004a), highlighting the problems faced by the bioaccumulative nature of mercury. The chapter then concludes with an examination of various potential management scenarios.

ECONOMIC MODEL

The economic submodel accounts for the revenues and costs of harvesting king mackerel and is defined in terms of commercial catch. A standard revenue function for the commercial fishery can be represented as:

$$(4.1) \quad \text{Rev}_{s,t} = P \cdot \text{CommCW}_{s,t}$$

where $\text{Rev}_{s,t}$ is the revenue generated in year t by catches from stock s , P is the average unit ex-vessel price for king mackerel, and $\text{CommCW}_{s,t}$ is the total weight of the commercial catch in year t from stock s .²⁵ The National Marine Fisheries Service (NMFS) maintains the Accumulated Landings System (ALS) database of monthly landing and the value of these landings for a variety of species. The ALS database²⁶ was used to calculate an average ex-vessel price of \$1.49 per pound for king mackerel over the years 1999 to 2006. A single price was used for both stocks because of the difficulty brought about by the mixing zone and the way the biological model was defined and parameterized.²⁷ While some authors have included price-quantity relationships in their bioeconomic models (e.g., Thunberg, Helser, and Mayo 1998; Kennedy 1999), many studies assume constant prices, either because the fishery studied is a small fraction of the overall market (Bjorndal, Ussif and Sumaila 2004; Yew and Heaps 1996; Amundsen, Bjorndal, and Conrad 1995) or due to the lack of adequate data (Pintassilgo and Duarte 2002; Kulmala, Laukkanen, and Michielsens 2008). An unpublished analysis of the demand for king mackerel (Vondruska 1999) suggests that the constant price assumption is reasonable in this case, as

²⁵ This formulation of the revenue function implicitly assumes that prices are not a function of the distribution of size or quality of the fish caught. There was no available data for king mackerel that distinguishes price by size class.

²⁶ Source: <http://www.st.nmfs.noaa.gov/st1/commercial/>

²⁷ Recall that the population dynamics model makes use of parameters generated under the assumption that 50% of the winter mixing zone catches are from the Atlantic and not under the FMP assumption that attributes them all to the Gulf stock. Therefore, a fish caught in the mixing zone in the winter could be from either stock and trying to assign prices based on stocks is impossible given the lack of catch-specificity in the data.

demand was found to be highly elastic with respect to price.²⁸ Additionally, an examination of the relationship between king mackerel price and quantity landed from 1977-2007 revealed no significant relationship.²⁹

The link between the population dynamics and economic model is a Cobb Douglas harvest function relating catch to fishing effort and biomass. Fishing effort is simply a measure of the amount of fishing and is expressed in a variety of terms in the fishery economics literature. Commonly used measures include fishing days, gear days, days at sea, or number of trips. The harvest function for this study is given by

$$(4.2) \quad CommCW_{s,t} = q_s E_{s,t}^{\phi_s} B_{s,t}^{\gamma_s}$$

where $CommCW_{s,t}$ is the total weight of all fish caught in year t from stock s , q_s is the catchability coefficient for stock s , $E_{s,t}$ is the fishing effort exerted on stock s in year t , $B_{s,t}$ is the biomass of stock s in year t , ϕ_s is the catch-effort elasticity for stock s , and γ_s is the catch-stock elasticity for stock s . Harvest functions of this type are often used for schooling species like mackerel (Kennedy 1992; Bjorndal 1988; Pintassilgo and Duarte 2002). Additionally, this form of the harvest function relates commercial catch to a measure of effort that can be evaluated in economic terms (Pintassilgo and Duarte 2002). Thus, given an estimate of the commercial catch weight, equation 4.2 allows for the calculation of an estimate of fishing effort that can ultimately be used in a cost equation.

Data used to estimate the harvest function was obtained through the Coastal Logbook database maintained by NMFS. The database includes a unique trip identifier, landing date,

²⁸ The referenced study estimated that if landings of king mackerel were reduced by 1 million pounds, ex-vessel price would only increase by 2 cents per pound. It should be noted that Vondruska acknowledged uncorrected problems of serially correlated residuals in his models. His findings, however, were consistent with those in an earlier unpublished work by Easley et al. (1993) who used an autoregressive procedure to address the problem.

²⁹ An early study of the king mackerel pricing system by Prochaska (1979) found that a change in landings of 1 million pounds resulted in a 7 cent change in price. However, this study was conducted when the industry was much larger than it is today, and the results are not directly meaningful to this research. Additionally, the scenarios presented in this research sought to minimize overall changes in commercial catch.

fishing gear deployed, areas fished, number of days at sea, number of crew, species caught, whole weight of the landings, and gear specific fishing effort. In the case of hook and line fisheries, these effort measures include number of lines fished, number of hooks per line and estimated total fishing time. Collection of effort data on the logbook form began in 1998 for king mackerel and, for the purposes of this study, extended through the year 2006. Biomass estimates were calculated from the numbers-at-age and weights-at-age for each migratory group given in the SEDAR 16 Final Stock Assessment Report (2009). The data available for estimation therefore consisted of 9 years of observations (1998-2006).

Only trips that reported one area and one gear fished were included in the analysis.³⁰ Additionally, data were limited to catch and effort measures reported from vessels that had king mackerel as its primary harvested species (i.e. king mackerel accounted for the greatest percentage of catch on that trip) and that utilized hook and line gear. Clear outliers in the data were also excluded from the analyses, including trips reporting more than seven lines fished, 20 hooks per line fished, more than 10 days at sea, or more than 3,120 pounds of king mackerel landed.³¹ Because the logbook only contains information from fishing trips taken by fishermen holding a federal fishing permit, it therefore does not contain all the king mackerel landings reported in the ALS data. For the purposes of this study, however, it was assumed that the information found in the logbook data could be extended to adequately represent non-federal permit holders who commercially fished for king mackerel.

The presence of a stock mixing zone off of the south coast of Florida presented additional problems for analyzing the commercial catch of king mackerel, as catches reported in the mixing

³⁰ A single fishing trip may report multiple gears and multiple areas fished. In that case, it is difficult to assign catch and effort to specific gears or locations. Eliminating trips with more than one area or gear fished accounted for the removal of less than one percent of the available observations.

³¹ These outlier values were used by McCarty (2008) in constructing a king mackerel tuning index using the coastal logbook data and were adopted here to allow for consistency with previous studies.

zone during winter could belong to either the Atlantic or Gulf stock. Thus, the approach used in this study was to separate the logbook catch and effort data into three regions: the Atlantic, Gulf, and Mixing (defined in SEDAR16 2009). The mixing zone was further broken down into summer catches attributed entirely to the Atlantic stock, and winter catches that (for management purposes) are counted as Gulf catches. The data were then aggregated by year for each region. The ALS catch data for the same years was then broken down using the same process. The proportion of total catch accounted for by the logbook data was calculated for each group during each year, and used to scale effort to correspond with the ALS catch. In other words, if the 1999 logbook catches attributed to the Gulf represented 81% of the Gulf catch reported in the ALS, the corresponding logbook effort was divided by that proportion to obtain a scaled version of effort to use in the estimation.³² The data were then aggregated by year for the Gulf and Atlantic stocks using the 50% winter mixing zone assumption employed by SEDAR 16. Gulf catch was calculated by summing Gulf catches and half the winter mixing zone catches from the ALS data set, while the Atlantic catch was determined by summing Atlantic catches, summer mixing zone catches, and half of the winter mixing zone catches for a given year. A similar approach was employed for the rescaled effort measures.

Given that the available data was limited, it was not feasible to estimate a separate harvest function for each migratory group (stock). Under the assumption that the catchability coefficient, catch-stock elasticity, and catch-effort elasticity were the same for both stocks, a single production function was estimated from the constructed data (2 stocks for each of 9 years, or 18 total observations). This approach was considered reasonable given that hook and line was

³² This rescaling was necessary because of the catch-effort elasticity parameter on effort in equation 4.2. In order to accurately estimate this relationship, it is important to have a measure of all of the effort.

the primary gear used throughout the king mackerel fishery for the years examined, thereby avoiding the specification problems that may have occurred with changing gears by stocks.

Hours fished was chosen as the measure of effort for the production (harvest) estimation after some experimentation with various effort metrics. Estimation then proceeded using the Gulf catch and effort data described above with the calculated Gulf biomass, and the Atlantic catch and effort data with the Atlantic biomass. Equation 4.2 was linearized by taking the natural log of both sides, and then estimated using OLS regression. The parameter estimates are given in Table 4.1. While the overall model fit is rather low (implying the potential for better specifications, especially in terms of explanatory variables, if the data were available), the parameter estimates appear reasonable given previously reported values in the literature.

Table 4.1: Production function estimation results

N=18	F value	Pr>F	R-squared	0.4046
	5.1	0.0205	Adj R-Sq	0.3252
Parameter	Coefficient	Std. Error	t Value	Pr > t
ln q	3.5934	3.5314	1.02	0.3250
ϕ	0.5256	0.1860	2.83	0.0128
γ	0.2948	0.1165	2.53	0.0230

The catch-stock elasticity estimate of 0.2948 is in line with prior applied studies of schooling species that used constant elasticity production functions, most of which found very low catch-stock elasticities (Amunsden, Bjorndal, and Conrad 1995; Bjorndal 1988). Although Pintassilgo and Duarte (2002) note that catch-effort elasticities for schooling species are generally very close to one (Pintassilgo and Duarte 2002), the estimated result of 0.5256 does not seem unreasonable given that king mackerel are primarily harvested with hook and line gear and tend to strongly school only during migration. Under these conditions, an increase in effort, holding stock size constant, would be expected to lead to a less than proportional increase in

catch. In the final analysis, the catch-effort and catch-stock relationships for any given species are empirical questions. Despite the limited data and low degree of fit, it was felt that the estimated parameters were preferred to the alternative used by Pintassilgo and Duarte (2002), where the catch-effort elasticity was assumed one, the catch-stock elasticity was assumed to be either 0.20 or 0.80 depending on the gear utilized, and then the catchability coefficient was calculated for the base year and assumed to hold for all remaining years.

Given the assumptions and estimates above, fishing costs can be modeled as a function of fishing effort, where the cost of fishing for king mackerel is represented as:

$$(4.3) \quad Cost_{s,t} = cE_{s,t}$$

where $Cost_{s,t}$ is the variable cost of fishing from stock s in year t , c is the constant cost per unit of effort, and $E_{s,t}$ is the fishing effort exerted on stock s in year t . Fixed costs were not considered because modeling was not done at the vessel level and because most fleets pursue other species in addition to king mackerel, thus making the assignment of fixed cost to mackerel fishing problematic (Pintasilgo and Duarte 2002; Thunberg, Helser, and Mayo 1998). The assumption of constant cost per unit of effort is commonly used in the fishery economics literature (Kulmala, Laukken, and Michielsens 2008; Garza-Gil and Varela-Lafuente 2007; Bjorndal and Brasao 2006; Garza-Gil, Varela-Lafuente, and Suris-Regueiro 2003; Pintassilgo and Duarte 2002; Thunberg, Helser, and Mayo 1998)

Cost information was obtained from NMFS through the coastal logbook database. The logbook form was modified in 2002 to collect data on the variable expenditures associated with each fishing trip. Available data for years 2002 through 2007 included the amount and cost of fuel, ice, bait and groceries, along with the wages or shares for the crew and captain. As before, this study focused on catch and effort measures reported from vessels that had king mackerel as its primary harvested species and that utilized hook and line gear. Clear outliers in the data were

again excluded from the analyses. Trip cost was calculated by summing labor cost, fuel cost, ice cost, bait cost, and groceries. This was divided by hours fished to obtain a cost per hour fished for each trip. The average cost per hour fished over the time period 2002-2007 was then calculated for the model. As in the case of prices, the same cost is used for both migratory groups, a reasonable assumption given that most catches occur in the mixing zone and the gear used to target king mackerel is primarily hook and line for both stocks.

With the revenue and cost functions defined, the profit function can be described as:

$$(4.4) \quad \pi_{s,t} = Rev_{s,t} - Cost_{s,t}$$

where $\pi_{s,t}$ is the profit from commercial king mackerel fishing in stock s during year t . For all forward-looking simulations of the system, the profit was discounted over a study period of 25 years to obtain the net present value:

$$(4.5) \quad NPV_s = \sum_{t=1}^T \left(\frac{1}{1+r} \right)^t \pi_{s,t}$$

where NPV_s is the net present value of the fishery for stock s , $\pi_{s,t}$ is the profit from commercial king mackerel fishing in stock s during year t , and r is the discount rate. The discount rate chosen for this study was 5 percent, a value that is similar to those recently used by Bjorndal and Lindroos (2004), Bjorndal et al. (2004), and Kulmala, Laukken, and Michielsens (2008).³³

MERCURY CONCENTRATION MODEL

One of the unique contributions of this research is the linking of species-specific mercury concentration information with a bioeconomic model of the commercial mackerel fishery. In order to accomplish this linkage, functional relationships need to be identified between

³³ Given that this study focuses on how NPV might change given various regulatory changes, the exact discount rate used is not critical as long as the time dynamics of the regulatory impacts are similar across scenarios. To the extent that they are not, however, sensitivity analysis could be used to determine the impact of changing discount rates on implications of model results.

biological stages of the fish and the degree to which mercury (in this case) has bioaccumulated over time. One approach for developing these linkages is to relate fish size with mercury concentration information. To do this, growth curves are presented for king mackerel that relate fish length to age, thus providing the backward linkage into the population dynamics model. Next the equations relating fish size to mercury concentration are presented, and then the relationship is extended to show mercury concentration by age class. Finally, the average mercury concentration for commercially caught king mackerel is determined.

King mackerel are assumed to grow according to a standard Von Bertalanffy growth function (as in SEDAR 16 Final Report 2009) such that

$$(4.6) \quad FL_{s,a} = L_{\infty,s} [1 - e^{-K_s(a-a_{0,s})}]$$

where $FL_{s,a}$ is the fork length (measured in centimeters) of an age a king mackerel from stock s , $L_{\infty,s}$ is the asymptotic length for stock s , K_s is a positive parameter for stock s , and $a_{0,s}$ is the arbitrary origin of the growth curve for stock s (Beverton and Holt 1957). The estimation of the parameters in this model is discussed in Ortiz and Palmer (2008), and their parameter estimates for the Gulf and Atlantic groups are given in the Table 4.2. As indicated in the table, there are slight differences in the growth patterns between the two king mackerel stocks, with the Gulf group growing slightly larger. This is illustrated in Figure 4.1, along with the observation that king mackerel are fast growing fish, reaching the current minimum legal size limit of 24 inches at approximately 2 years of age.

Table 4.2: Von Bertalanffy growth parameters for the Atlantic and Gulf king mackerel stocks

Stock	L_{∞}	K	t_0
Atlantic	114.1	0.245	-1.689
Gulf	122.4	0.177	-2.651

Source: Ortiz and Palmer (2008)

Given the prevalence of mercury bioaccumulation in aquatic species, larger king mackerel would be expected to have greater concentrations of mercury. This has led many states to issue king mackerel consumption advisories to recreational fisherman based on the fork length of the fish caught.³⁴ In a recent study, Adams and McMichael (2007) examined mercury levels for king mackerel off the Gulf and Atlantic coasts of Florida and found a significant positive relationship between fish size and mercury concentration for king mackerel. They sampled 143 fish from the near and offshore waters of Florida’s Atlantic coast and 136 from near and offshore waters of the Gulf coast of Florida. The Gulf king mackerel were found to contain significantly higher amounts of mercury than those in the Atlantic, with mean mercury levels in the sample of 0.94 parts per million (ppm) for the Atlantic waters and 1.51 ppm for the Gulf waters. All but a few of the fish sampled were above the minimum legal size limit. Linear and non-linear regressions were used to describe the relationships between king mackerel size and total mercury concentration. The estimations from that study, which will be used to quantify the relationship between king mackerel size and mercury concentration, are given below:³⁵

$$(4.7) \quad Hg_s = \begin{cases} 1.11 \cdot 10^{-7} FL_s^{3.51} & f\ or\ s = Atlantic \\ e^{-3.09 + .032 FL_s} & f\ or\ s = Gulf \end{cases}$$

where Hg_s is the mercury concentration in ppm for a fish from stock s and FL_s is the fork length in centimeters for a fish from stock s . While it would have been preferable to obtain size/mercury samples from throughout the Gulf and Atlantic waters to estimate the relationship, it is not unreasonable to use the Adams and McMichael (2007) estimations given that most king mackerel are caught off the Florida coast (and, in particular, in the mixing zone). It should be

³⁴ See <http://www.epa.gov/waterscience/fish/states.htm> for detailed information on each state.

³⁵ The equations given in 4.7 have been converted to use fork length in centimeters. Adams and McMichael (2007) use fork length in millimeters for their estimations. Additionally, the Gulf equation was presented and estimated in logged form in the original work.

noted that mercury data were available for states bordering the Northern Gulf of Mexico through a database developed for the Gulf of Mexico Mercury Project (Ache, Boyle, and Morse 2000). This information, however, was simply a compilation of state monitoring databases that were inconsistent in their sampling procedures and reporting, with the bulk of the observations from Texas where little king mackerel is commercially caught. Thus, for the purposes of this study, it was assumed that the Adams and McMichael information was more directly applicable.

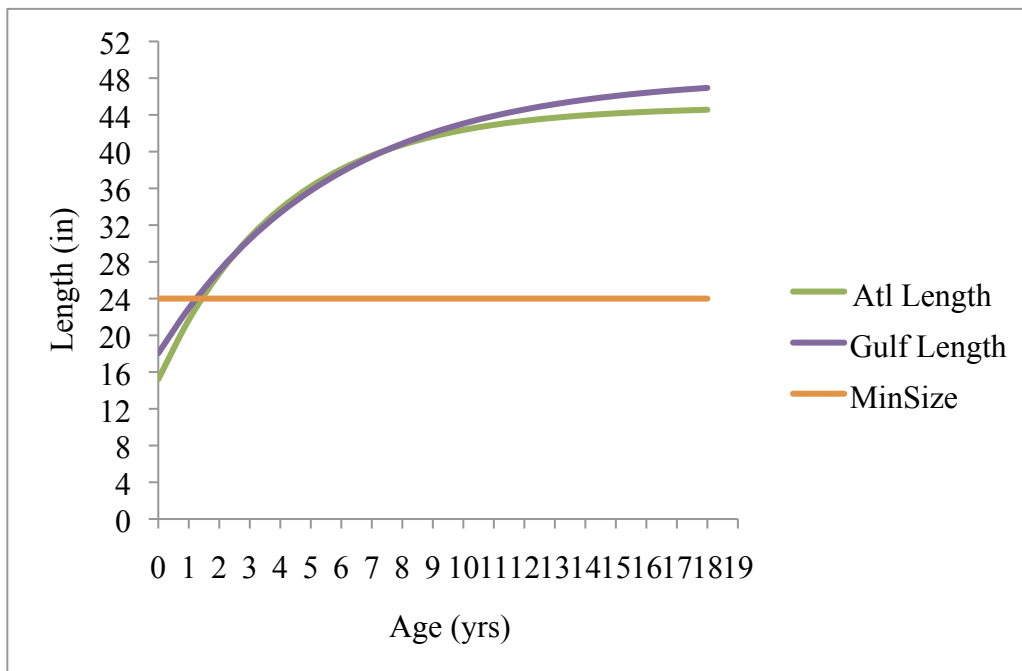


Figure 4.1 Von Bertalanffy growth curves and the minimum size limit for the Atlantic and Gulf of Mexico king mackerel migratory groups³⁶

Given that the king mackerel population dynamics model was constructed using age classes, it is useful to convert the size-mercury relationship reported in Adams and McMichael into age-mercury relationship by using equation 4.6 to calculate the average length of a fish for each age class. Subsequently, equation 4.7 can be used to determine average mercury

³⁶ While the growth relationships define length in terms of centimeters, they are graphed here in terms of inches for clarity with respect to the 24 inch catch size limit.

concentration at age by substituting the average length for each age class into the equation as follows:

$$(4.8) \quad Hg_{s,a} = \begin{cases} 1.11 \cdot 10^{-7} FL_{s,a}^{3.51} & f\ ors = Atlantic \\ e^{-3.09+.032FL_{s,a}} & f\ ors = Gulf \end{cases}$$

where $Hg_{s,a}$ is the mercury concentration in ppm for a fish of age a from stock s and FL_s is the average fork length in centimeters for a fish of age a from stock s . The resulting relationship is presented graphically in Figure 4.2 with the current FDA limit of 1 ppm highlighted.³⁷ For both stocks, the average king mackerel is at or exceeds the FDA limit by the time it reaches 6 years of age.

With a relationship between age and mercury concentration established, it would be ideal to use surveyed population consumption information to link this back through the bioeconomic model and ultimately to human exposure. Unfortunately, there is little information available regarding the consumption of king mackerel in the United States. It is known that king mackerel are not widely consumed in the U.S., with a recent mercury assessment study estimating the market share to be around .05% based on 2001 reported landings (Carrington, Montwill, and Bolger 2004). Further compounding the issue is the fact that king mackerel are often lumped together in consumption surveys with other mackerel species such as Spanish or Atlantic mackerels. In the absence of specific consumption information for king mackerel, this study calculated the average mercury concentration for all commercially caught king mackerel. This is

³⁷ While the graph includes up through age 15, recall that the terminal age class in the population dynamics model is the age 11+ group which contains all fish age 11 or older. For determining the appropriate parameters to use for the 11+ age class, the preferred method is to construct a weighted average of the parameter values over the remaining ages that make up the plus group. There was no information available about the age breakdown within the plus group. Rather than equally weight the mean mercury concentration over an arbitrary number of age classes, this study uses the age 11 values for the age 11+ group.

done by linking the relationship between age and mercury concentration with the output of the bioeconomic model as follows:

$$(4.23) \quad \overline{Hg}_{s,t} = \frac{\sum_{a=0}^{11} Hg_{s,a} \cdot CommCN_{s,a,t}}{CommCN_{s,t}}$$

where $\overline{Hg}_{s,t}$ is the mean mercury concentration for all commercially caught king mackerel from stock s in year t , $Hg_{s,a}$ is the mercury concentration in ppm for a fish of age a from stock s , $CommCN_{s,a,t}$ is the number of age a fish commercially caught in year t from stock s , and $CommCN_{s,t}$ is the total number of commercially caught fish from stock s in year t . This metric will be used as a benchmark to measure the impacts of simulated changes in how king mackerel are harvested or targeted. If the total annual amount of mercury in all commercially caught fish cannot be reduced, it seems unlikely that any health benefits would come from any alternative harvesting scenarios.

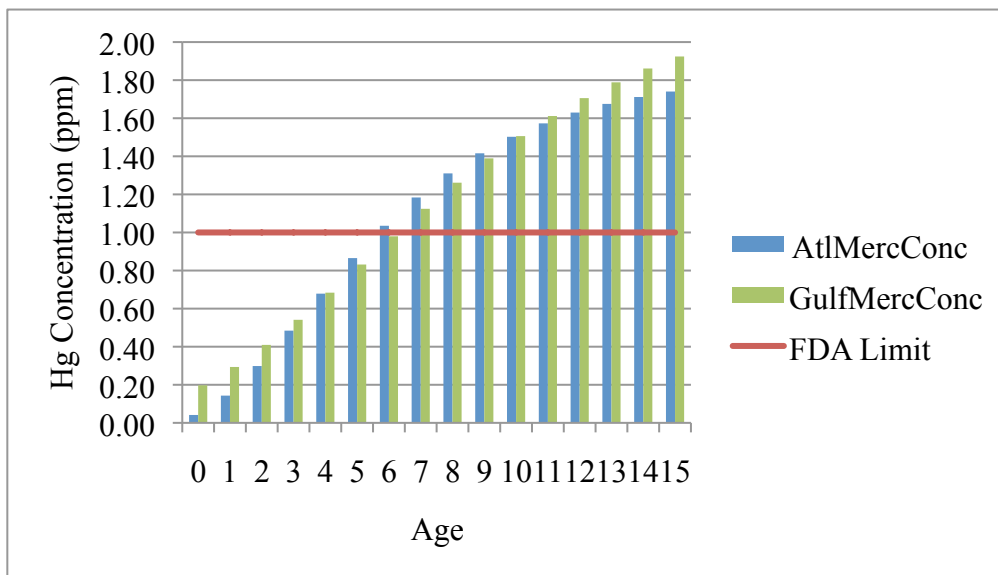


Figure 4.2 Average mercury concentration by age for the Gulf and Atlantic king mackerel stocks

SIMULATION RESULTS

The integrated population dynamics, economic, and contamination model was designed to investigate the impact of alternative fisheries management schemes on the movement of mercury from its environmental stock to the human population. Specifically, the study sought to discover if alternative harvesting patterns could reduce the amount of mercury reaching king mackerel consumers without severely affecting the economic viability of the harvesting industry or damaging the biological viability of the king mackerel stocks. Given that mercury contamination of fish is primarily an age/size phenomenon, it was anticipated that the primary policy objective would be to alter the age (and therefore size) composition of the commercial catch. From a modeling perspective, this can be accomplished by changing commercial fishing mortality at age and comparing the results across scenarios. For the purposes of the discussion below, these scenarios were developed to compare the results of shifting fishing pressure to progressively smaller and younger (and thus, less contaminated) fish.³⁸ Specifically, the simulated scenarios examine the (1) status quo, (2) elimination from the catch of fish age 6 and older; (3) the establishment of a less than 33" fork length maximum size limit (with no increased catch of smaller fish), (4) scenario 3 with an increase in catch of smaller fish, (5) a reduction in the catch of age 4 fish accompanied by an increased catch of younger fish, and (6) scenario 5 with consideration for incidental catch. The model was implemented in Matlab and code for the status quo scenario is contained in Appendix D.

³⁸ The bioeconomic literature typically approaches these types of investigation in two distinct ways; direct and indirect optimization. Direct optimization is generally relegated to those models that are analytically tractable, which is not the case in this study. Indirect optimization involves a wide range of approaches that usually incorporate some form of a grid search (either formal or informal) over the potential solution space. This study takes the informal approach, examining potential solutions via a set of pre-specified scenarios. Although these scenarios will not result in the identification of an optimal solution, it does provide an opportunity to determine if a solution might exist within the defined space and helps to narrow the space for use in potential future multi-objective optimization studies.

The results from the selected scenarios are presented next, with the accompanying discussions focusing on the following key variables; annual mean mercury concentration in the harvest, annual commercial catch in pounds, annual stock biomass, annual profits in the fishing industry, and NPV of the fishery. Figures 4.3-4.6 graphically depict simulated mercury concentration, catch, biomass, and profit for all Gulf scenarios, thereby allowing for easy comparison among the potential management actions. Figures 4.7-4.10 present the same for the Atlantic scenarios. Table 4.4 presents the NPV for each Gulf scenario along with minimum, maximum, and mean mercury concentrations over the 25 year simulation time frame, while Table 4.5 presents the same for the Atlantic scenarios.

Gulf Scenario (1): Status Quo

The status quo scenario establishes a baseline model that describes the biological, economic, and contamination status of the Gulf king mackerel stock for use in evaluating the effect of the other alternatives. The time horizon of the simulation is 25 years, spanning 1999-2023. This time span was chosen because it allows the complete tracking of a number of cohorts through time and, thus, allows the full implications of any new management regime to be examined. The economic parameters outlined above are used throughout the simulation time span. In terms of the population dynamics model, all time invariant parameters previously described are used. Initial numbers at age for 1999 are taken from SEDAR16, as are weights at age for 1999-2006. The commercial fishing mortality and remaining fishing mortality are derived (as discussed in the population dynamics model) for the years 1999-2006 and used in the simulation, but assumptions concerning these parameters must be used for the latter part of the simulation time horizon. One approach would be to use the mean values from 1999-2006 (or a subset of those years) for all remaining years in the simulation horizon (2007-2023), as done with recruitment values in Bjorndal, Ussif, and Sumaila (2004). Using this approach, simulated

biomass and catches (along with the corresponding profit and average mercury concentration) quickly level off as the system reaches a steady state. It is unrealistic, however, to think that catches will remain the same from year to year. Even when regulations remain largely unchanged, there is always some variability in the year-to-year catches due to both economic and environmental conditions. Thus, for the purposes of this research, it was decided to simply repeat the 1999-2006 time series of fishing mortality values throughout the simulation time span. Assuming that regulations remain the same, this approach captures the inherent variability in the catches while projecting the current system characteristics into the future.³⁹ A table presenting all baseline parameters is given in Appendix C.

Results for the status quo scenario indicate that under the current catch composition, the average mercury concentration of commercially caught king mackerel ranges from a low of 0.64 ppm to a high of 0.88 ppm with a mean of 0.76 ppm over the simulation time span. Stock biomass increases throughout the simulation years, albeit at a decreasing rate over the later years. This is not surprising given that commercial catches follow a similar pattern and the fishery is currently managed under a TAC that was designed to rebuild the stock from an overfished level. Annual profits to the fishery average \$2.5 million over the simulation period.

One aspect of the status quo scenario that warrants more discussion is the mean mercury concentration of 0.76 ppm over the simulation time frame. Given the U.S. FDA's action limit of 1 ppm, it is tempting to conclude that, since the simulated mean is lower, no action is warranted. The U.S. FDA (2001), however, reported a mean mercury value of 0.73 for all king mackerel, a value that was high enough to prompt consumption advisories and a study to reevaluate the original 1.0 ppm limit. The U.S. EPA already has put in place a more stringent threshold

³⁹ An alternative would be to develop the model using stochastic functions for the parameters, but given the limited data, it was not obvious that the additional complexity of this approach would yield any improvements in the model's ability to represent future outcomes.

regarding exposure to mercury. Defined as a reference dose (RfD), or the estimated daily amount of a substance that can be consumed safely over a lifetime, this new threshold calls for a maximum mercury exposure of 0.1 micrograms per kilogram of body weight. Unlike the U.S. FDA's limit, the U.S. EPA RfD depends not just on the concentration of mercury in the fish consumed, but also on the amount of consumption, the frequency of consumption, and the bodyweight of the consumer. Table 4.3 presents an analysis of the maximum mercury amount in ppm that could be present in a consumed fish while keeping the consumer at the U.S. EPA RfD given a consumption rate of one 6 ounce (170 g) meal per week. Based on the mean mercury level of 0.76 found in the status quo simulation, even a consumer weighing 250 pounds would greatly exceed the weekly RfD if they ate even one meal each week. Keeping this result in mind, the remainder of the simulations will be discussed.

Table 4.3: Average mercury concentration for persons of varying bodyweight needed to stay at or below the U.S. EPA reference dose of 0.1 micrograms given a consumption rate of one 6 ounce meal per week.⁴⁰

Weight (lbs)	Weight (kg)	EPA Daily RfD (micrograms/kg)	EPA Weekly RfD (micrograms/kg)	Ave. conc (ppm) to meet EPA RfD (1- 6oz meal/week)
45	20.41	2.04	14.29	0.08
50	22.68	2.27	15.88	0.09
60	27.22	2.72	19.05	0.11
70	31.75	3.18	22.23	0.13
80	36.29	3.63	25.4	0.15
90	40.82	4.08	28.58	0.17
100	45.36	4.54	31.75	0.19
110	49.9	4.99	34.93	0.21
120	54.43	5.44	38.1	0.22
130	58.97	5.9	41.28	0.24
140	63.5	6.35	44.45	0.26

⁴⁰ The idea for the calculations contained in this table resulted from calculations presented in PBS Now. Science and Health: The Mercury Story. January 21, 2005. See <http://www.pbs.org/now/science/mercuryinfish.html> for more details.

150	68.04	6.8	47.63	0.28
160	72.57	7.26	50.8	0.3
170	77.11	7.71	53.98	0.32
180	81.65	8.16	57.15	0.34
190	86.18	8.62	60.33	0.35
200	90.72	9.07	63.5	0.37
210	95.25	9.53	66.68	0.39
220	99.79	9.98	69.85	0.41
230	104.33	10.43	73.03	0.43
240	108.86	10.89	76.2	0.45
250	113.4	11.34	79.38	0.47

Table 4.3 continued

Gulf Scenario (2): Eliminate Harvesting of Fish Age 6 and Older

The next scenario investigated the effects on the fishery if management regulations prohibited catching king mackerel over age 6, or the age when the average king mackerel from the Gulf stock exceeds the U.S. EPA limit of 1 ppm.⁴¹ Commercial fishing mortalities were set to zero for ages 6-11, while commercial fishing mortalities for ages 0-5 were left at their baseline levels. As is the case in all scenarios investigated, the remaining fishing mortality is assumed unchanged from the baseline scenario.⁴² Simulation results for this scenario indicate that average mercury concentration of the commercially caught fish would be reduced to 0.57 ppm, but at a substantial cost to the harvesting industry. While biomass increases in this scenario relative to the status quo (as would be expected given that fishing mortality – and thus targeted effort – is assumed unchanged for the allowable age classes), commercial catches and profits dropped dramatically compared to the baseline model, with the NPV of the fishery decreasing by 29%.

⁴¹ Of course, in practice this age restriction would be implemented using a fork-length size restriction.

⁴² This research is concerned only with the commercial fishery, and does not aim to change the behavior of the recreational fisherman. Given that many recreational fisherman fish for fun or pleasure rather than food, it does not make sense to limit the size of their catch.

Gulf Scenario (3): Establish a Less than 33” FL Maximum Size Limit

Given the reduction of mercury found in Scenario 2 from eliminating the catches of age 6 and older fish, scenario 3 investigated an even more restrictive model. Many states that issue consumption advice for king mackerel consider those with a fork length of 33 inches or less safe for unrestricted consumption. A fork length of 33 inches corresponds to age 4 in the Gulf stock, so this scenario eliminated all catches of age 5 and older fish. Commercial fishing mortalities were set to zero for ages 5-11, while commercial fishing mortalities for ages 0-4 were left at their baseline levels. As in scenario 2, average mercury levels were significantly reduced from the baseline – in this case to an average of 0.52 ppm – but at the cost of a 44% reduction in the NPV of the fishing industry. Similarly to what occurred in scenario 2, biomass increases in this scenario relative to the status quo. Again this was expected given that fishing mortality –is assumed unchanged for the allowable age classes even as the number of harvestable age classes declines.

Gulf Scenario (4): Scenario 3 With an Increase in the Catch of Younger Fish

Scenario 4 builds on scenario 3 by adding some realism to the allocation of harvest (and, implicitly, the allocation of effort) across the age classes. While eliminating the catch of older fish can significantly decrease the average mercury level that will reach consumers, it is unrealistic to think that fishing effort will not be reallocated (in the absence of restrictive TACs) from larger to smaller fish. Scenario 4 assumes that commercial fishing mortality on ages 0 and 1 are unchanged (given the continuation of the current 24” minimum size limit) and that for ages 5-11 commercial fishing mortalities are again set to zero. For ages 2 and 3, it is assumed that commercial fishing mortalities will double from their baseline levels and age 4 commercial fishing mortalities remain at their baseline levels. This assumption about increasing fishing mortality for ages 2 and 3 was made in order to examine the effect of increased fishing on the

younger age classes, with the specific magnitude of the change being arbitrary but large enough to expect some response from the system simulation. Age 4 was left at baseline in an attempt to further alter the age composition of the catch and reduce average mercury concentration. Under these simulation assumptions, average mercury levels were reduced from the baseline levels to 0.50 ppm, or just slightly lower than what occurred without effort reallocation. Commercial catches and profits fell from baseline levels, but increased from scenarios 2 and 3. King mackerel stocks remained higher than baseline levels over time, suggesting that a switch to harvesting smaller fish does not necessarily have a negative impact on the stock health when larger, highly fecund fish are allowed to remain in the reproducing population and when catches remain below the baseline levels. Overall, fishing industry NPV was 25% lower compared to the baseline scenario 1.

Gulf Scenario (5): Reduction in Age 4 Catch Plus Increased Catch of Younger Fish

Given that the increased fishing pressure on younger fish in Scenario 4 does not negatively impact stock health, scenario 5 increases the fishing effort to an even larger degree. As in the previous scenario, scenario 5 assumes that commercial fishing mortality on ages 0 and 1 are unchanged and that for ages 5-11 commercial fishing mortalities are zero. For ages 2 and 3, it is assumed that commercial fishing mortalities will quadruple from their baseline levels. Age 4 commercial fishing mortalities are assumed to be half of their baseline levels in an attempt to reduce average mercury concentration even further. While these changes are to an extent arbitrary, they were chosen to keep the average commercial catch and effort levels relatively close to the average commercial catch from the baseline scenario. Average mercury concentration of the harvest under this scenario was reduced to 0.48 ppm, while the NPV of the fishing industry only fell 7% from the baseline scenario. Stock biomass is slightly below baseline in this scenario, but still exhibiting a pattern of increases over time.

Gulf Scenario (6): Scenario 5 Plus Incidental Catch

The final scenario explored is an extension of scenario 5. Given that fishing pressure on ages 2 and 3 are already quadrupled from baseline levels and it is most likely impossible to increase it without bound, it does not seem realistic to simulate the effects of further increases. However, it also seems unrealistic to simply eliminate all catches of age 6 and older king mackerel. While a maximum size limit can be implemented, thus rendering the sale of oversized fish illegal, the regulation will not actually stop these catches altogether – it merely prevents (for the most part) the marketing of the catch. Bycatch, or non-targeted or incidental catch of non-target age classes, is going to occur. To capture this phenomenon, this scenario builds on scenario 5 by including the bycatch of the larger age classes.

The inclusion of bycatch in the simulations is important for two reasons. First, if enough larger fish are caught incidentally this could negatively impact biomass, depending on release mortality. Second, even though a fisherman may not be able to legally sell oversized fish, they certainly incur a cost in terms of effort from landing the incidental catch. Unfortunately, under current management regimes larger fish are targeted (in king mackerel and most other species) and there is little or no information concerning potential bycatch of the larger age classes if they were made illegal. Given that it is in the fishermen's best interest to limit bycatch from an effort/cost perspective, this scenario assumed that commercial fishing mortalities for ages 5-11 fell to only 10% of their baseline values and not to zero as would occur under perfect adherence to size limits. All of the resulting catches from those age classes were then considered incidental and incurred a cost (both monetary and biological) even though they did not contribute to revenue. Further, it was assumed that the release mortality of the incidental catch was 100%, or that all fish caught and released later die. This is a somewhat extreme assumption, as the actual release mortality may be relatively low for hook and line fisheries in general and for king

mackerel in particular (SEDAR16 RD09 2009). This assumption was made because it can be viewed as a worst case scenario for the stock. The remaining fishing mortalities by age class are unchanged from scenario 5.

Scenario 6 simulated commercial catches and average mercury concentrations remained identical to scenario 5 because it was assumed that the incidental catch was not marketed. The difference with scenario 6 results lies in biomass and profits. With an additional cost incurred in harvesting unmarketable fish, average profits were predictably lower. These catches also resulted in a lower average biomass compared to scenario 5 and the baseline scenario 1, although the stock health still appears to be high. The NPV of the fishery was 14% less than baseline, with the average mercury reduction remaining at 0.48 ppm, a 37% change from baseline.

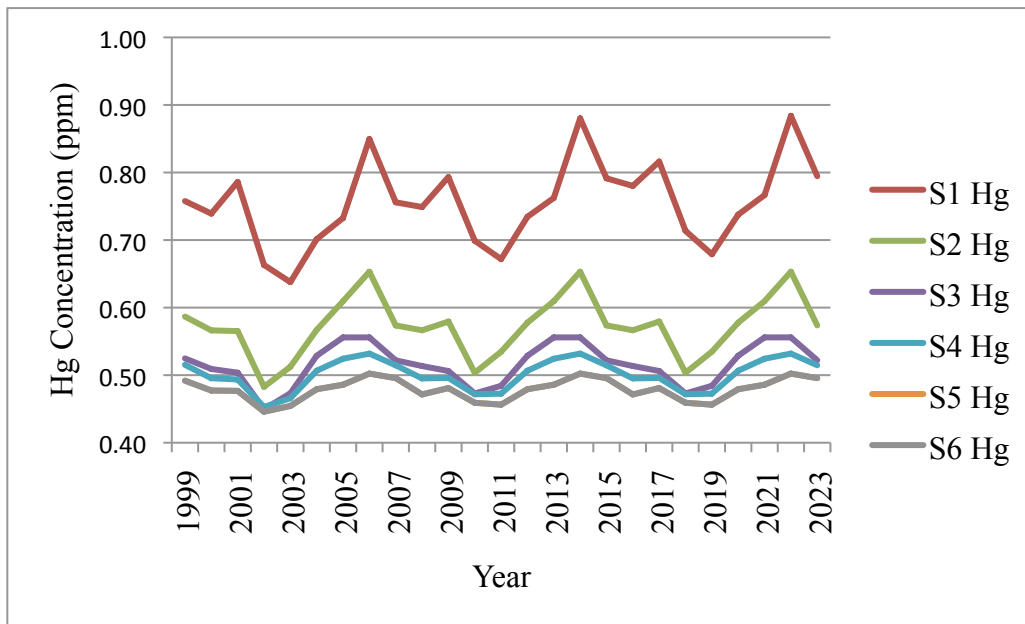


Figure 4.3: Simulated mercury concentrations (Hg) for given scenarios, Gulf stock

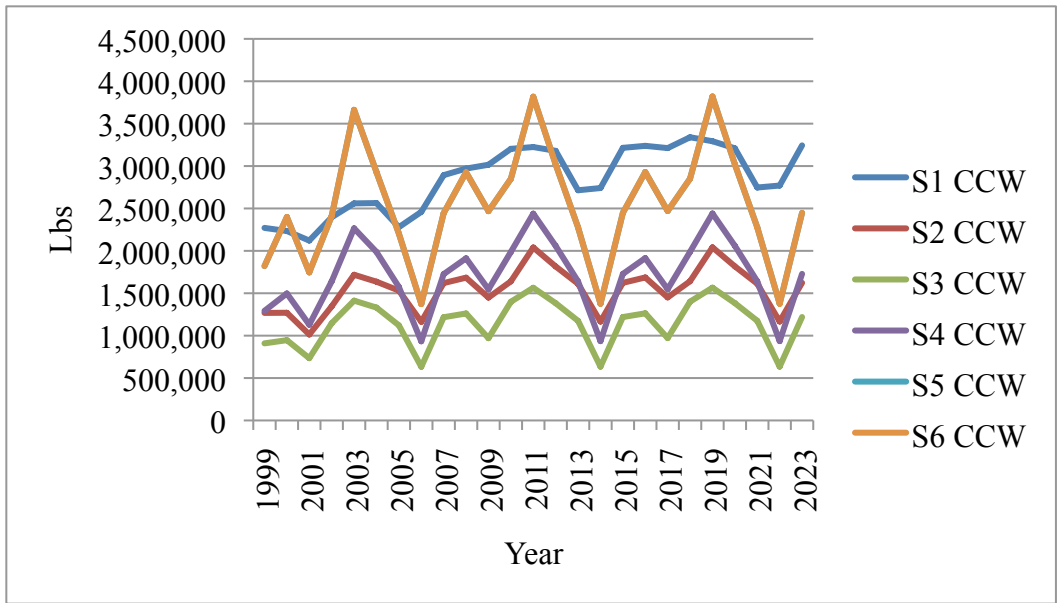


Figure 4.4: Simulated commercial catch in lbs for given scenarios, Gulf stock

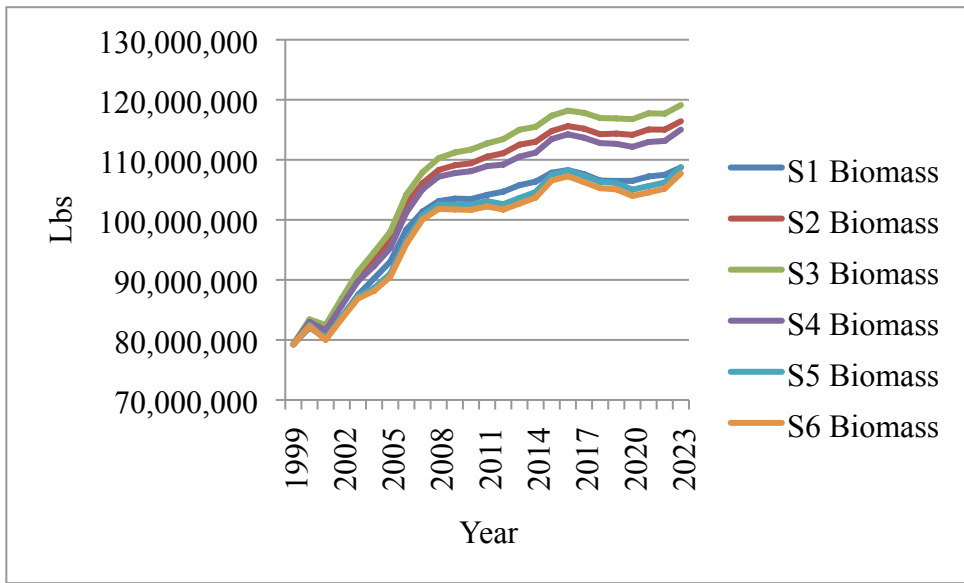


Figure 4.5: Simulated biomass for given scenarios, Gulf Stock

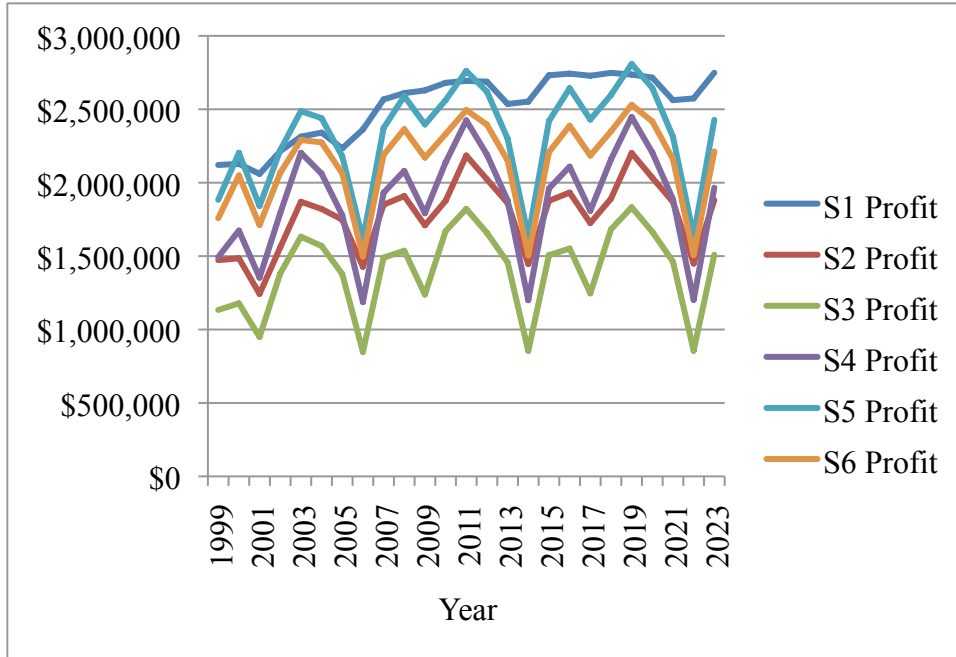


Figure 4.6: Simulated profit for given scenarios, Gulf stock

Table 4.4: Comparison of simulated NPV, percentage change from status quo, and mercury concentrations (Hg) for given scenarios, Gulf stock

Scenario	NPV	% Change	Min Hg	Max Hg	Mean Hg
1	\$34,561,343	-	0.64	0.88	0.76
2	\$24,403,130	-29.39%	0.48	0.65	0.57
3	\$19,406,615	-43.85%	0.45	0.56	0.52
4	\$25,985,333	-24.81%	0.45	0.53	0.50
5	\$32,192,745	-6.85%	0.45	0.50	0.48
6	\$29,635,679	-14.25%	0.45	0.50	0.48

Atlantic Scenario (1): Status Quo

The status quo scenario for the Atlantic stock was constructed in the same manner as described for the Gulf status quo. Results for the status quo scenario indicate that under the current catch composition, the yearly average mercury concentration of all commercially caught king mackerel ranges from a low of 0.56 ppm to a high of 0.86 ppm with a mean of 0.67 ppm over the simulation time span. Stock biomass decreases throughout the early simulation years before mostly leveling off over the latter simulation years. Commercial catches over the

simulation period averaged 2.5 million pounds, while annual profits to the fishery averaged \$1.9 million over the simulation period.

Atlantic Scenario (2): Eliminate Harvesting of Fish Age 6 and Older

The next scenario investigated the effects on the fishery if management regulations prohibited catching king mackerel over age 6, or the age when the average king mackerel from the Atlantic stock exceeds the U.S. EPA limit of 1 ppm. Commercial fishing mortalities were set to zero for ages 6-11, while commercial fishing mortalities for ages 0-5 were left at their baseline levels. As is the case in all scenarios investigated, the remaining fishing mortality is assumed unchanged from the baseline scenario. Simulation results for this scenario indicate that average mercury concentration would be reduced to 0.51 ppm, but at a cost of a 15% decrease in the NPV of the fishery. As expected (given the decreased harvesting of highly fecund older fish), stock biomass increases in this scenario relative to the status quo, while commercial catches and profits dropped dramatically compared to the baseline model.

Atlantic Scenario (3): Establish a Less than 33" FL Maximum Size Limit

Given the reduction of mercury found in Scenario 2 from eliminating the catches of age 6 and older fish, scenario 3 investigated an even more restrictive model for the Atlantic stock. A fork length of 33 inches corresponds to age 4 in the Atlantic stock, so this scenario eliminated all catches of age 5 and older fish. Commercial fishing mortalities were set to zero for ages 5-11, while commercial fishing mortalities for ages 0-4 were left at their baseline levels. As in scenario 2, average mercury levels were significantly reduced from the baseline – in this case to an average of 0.44 ppm – but at the cost of a 27% reduction in the NPV of the fishing industry. Biomass is again significantly higher than baseline, an expected result given the assumptions.

Atlantic Scenario (4): Scenario 3 With an Increase in the Catch of Younger Fish.

Scenario 4 builds on scenario 3 by adding some realism to the allocation of harvest across the age classes. While eliminating the catch of older fish can significantly decrease the average mercury level that will reach consumers, it is unrealistic to think that fishing effort will not be reallocated to target smaller fish. Scenario 4 assumes that commercial fishing mortality on ages 0 and 1 are unchanged and that for ages 5-11 commercial fishing mortalities are again set to zero. For ages 2 and 3, it is assumed that commercial fishing mortalities will double from their baseline levels and age 4 commercial fishing mortalities remain at their baseline levels. Under these simulation assumptions, average mercury levels were reduced from the baseline levels to 0.41 ppm, or just slightly lower than what occurred without effort reallocation. On average, commercial catches and profits fell from baseline levels, but increased from scenarios 2 and 3. King mackerel stocks remained slightly higher than baseline levels over time, once again highlighting that a switch to harvesting smaller fish does not necessarily have a negative impact on the stock health when larger, highly fecund fish are allowed to remain in the reproducing population and when catches remain below the baseline levels. Overall, fishing industry NPV was more than 10% lower when compared to the baseline scenario.

Atlantic Scenario (5): Reduction in Age 4 Catch plus Increased Catch of Younger Fish

Given that the increased fishing pressure on younger fish in Scenario 4 does not negatively impact stock health, scenario 5 increases the fishing effort to an even larger degree. As in the previous scenario, scenario 5 assumes that commercial fishing mortality on ages 0 and 1 are unchanged and that for ages 5-11 commercial fishing mortalities are zero. For ages 2 and 3, it is assumed that commercial fishing mortalities will triple from their baseline levels. Age 4 commercial fishing mortalities are assumed to be half of their baseline levels in an attempt to reduce average mercury concentration even further. As in the case of the Gulf, these changes

were chosen to keep the average commercial catch and effort levels relatively close to the average commercial catch and effort levels from the baseline scenario. Average mercury concentration of the harvest under this scenario was again reduced to 0.38 ppm, while the NPV of the fishing industry decreased only 8% from the baseline scenario. Commercial catches were lower than baseline on average, as were profits. Biomass values were higher than baseline for this scenario, an expected result given the decreased catch.

Atlantic Scenario (6): Scenario 5 Plus Incidental Catch

As in the case of the Gulf stock, the final scenario explored is an extension of scenario 5 that includes incidental catch of the larger age classes. This scenario assumed that commercial fishing mortalities for ages 5-11 fell to only 10% of their baseline values and not to zero as modeled in the other scenarios. All of the resulting catches from those age classes were then considered incidental and incurred a cost (both monetary and biological) even though they did not contribute to revenue. Further, it was assumed that the release mortality of the incidental catch was 100%, or that all oversized fish caught and released later die. The remaining fishing mortalities by age class are unchanged from scenario 5. Scenario 6 simulated commercial catches and average mercury concentrations remained identical to scenario 5 because it was assumed that the incidental catch was not marketed. The difference with scenario 6 results lies in biomass and profits. With an additional cost incurred in harvesting unmarketable fish, average profits were lower than in Scenario 5. These catches also resulted in a slightly lower average biomass compared to scenario 5 and the baseline scenario 1. After the initial decline, the biomass levels are generally fairly stable (suggesting good stock health), but possibly exhibit a slight downward trend over the last few years of the simulation horizon. The NPV of the fishery was 19% less than baseline, while the average mercury is reduced by 44%.

Table 4.5: Comparison of simulated NPV, percentage change from status quo, and mercury concentrations (Hg) for given scenarios, Atlantic stock

Scenario	NPV	% Change	Min Hg	Max Hg	Mean Hg
1	\$26,920,041	-	0.56	0.86	0.67
2	\$22,923,928	-14.84%	0.45	0.59	0.51
3	\$19,540,856	-27.41%	0.37	0.50	0.44
4	\$24,170,976	-10.21%	0.36	0.47	0.41
5	\$24,669,159	-8.36%	0.34	0.43	0.38
6	\$21,873,321	-18.75%	0.34	0.43	0.38

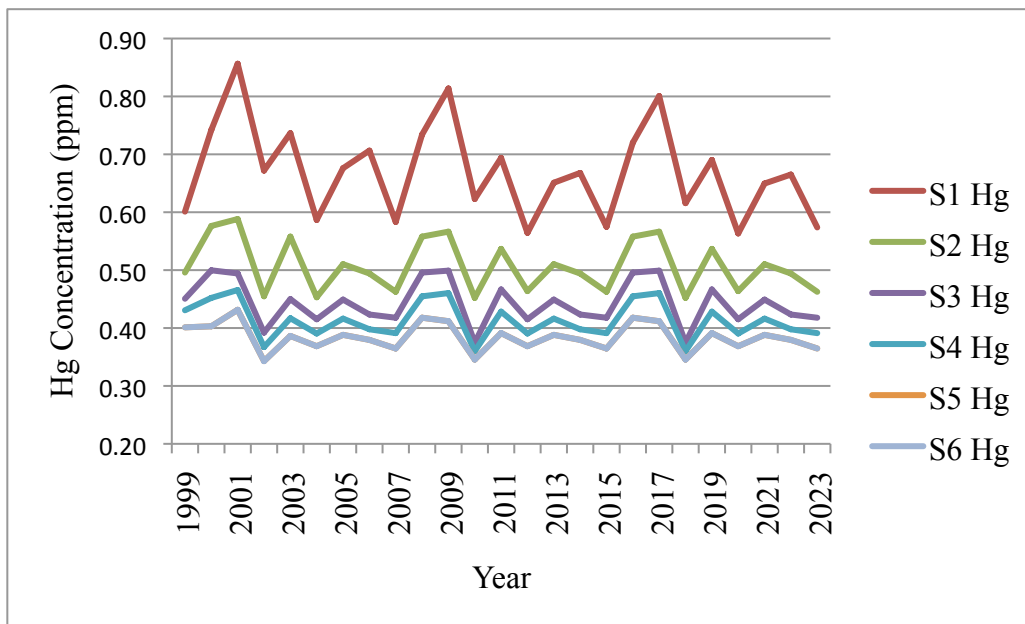


Figure 4.7: Simulated mercury concentrations (Hg) for given scenarios, Atlantic stock

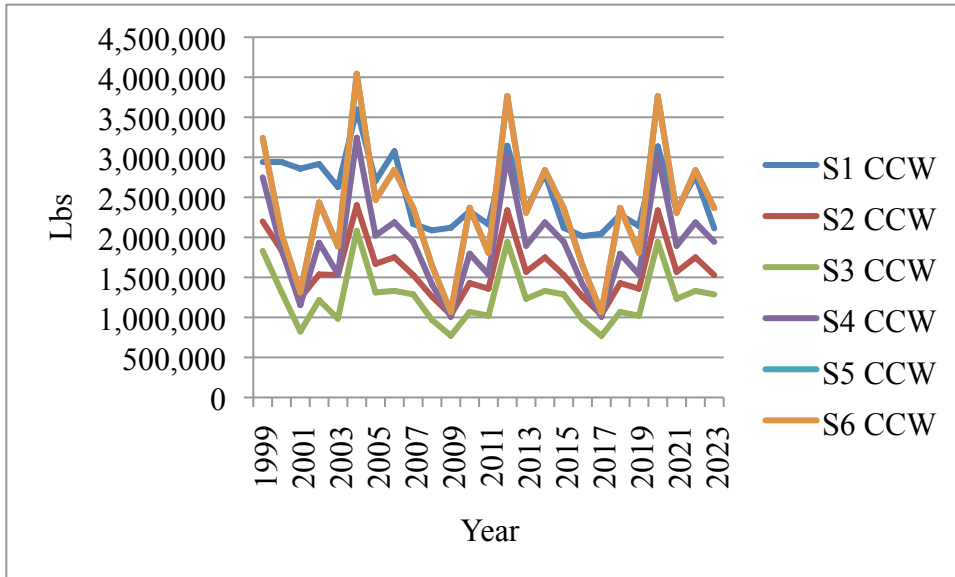


Figure 4.8: Simulated commercial catch in lbs for given scenarios, Atlantic stock

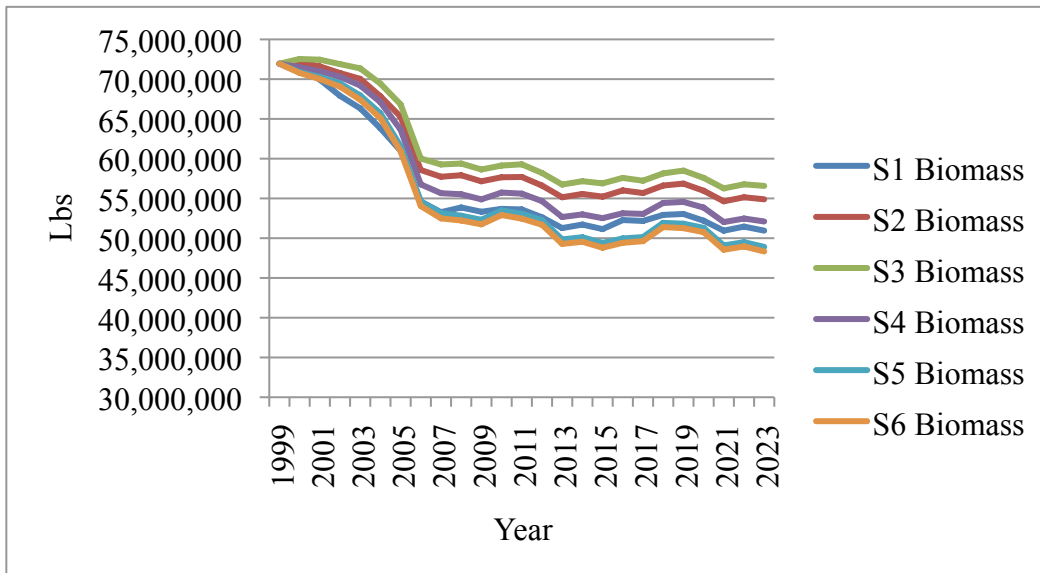


Figure 4.9: Simulated stock biomass in lbs for given scenarios, Atlantic stock

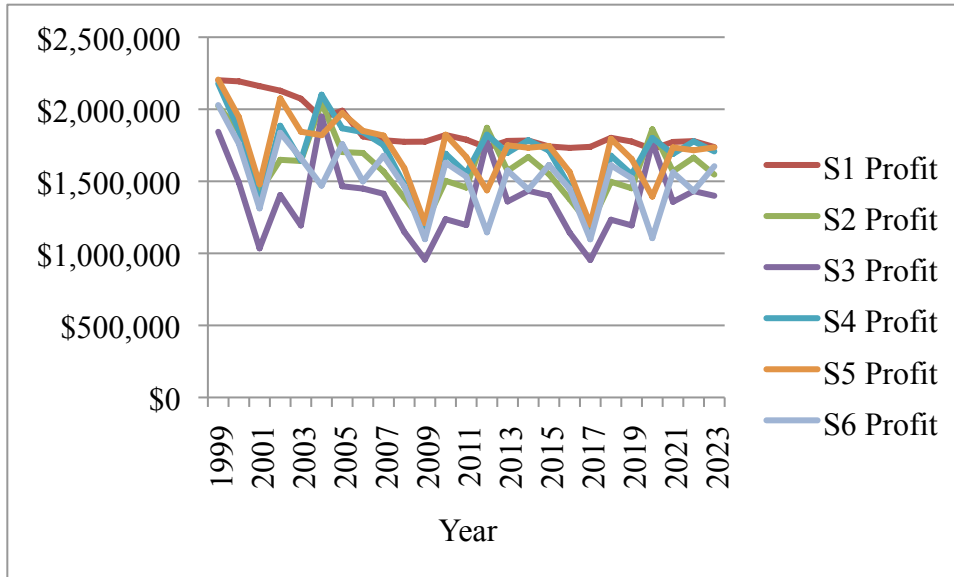


Figure 4.10: Simulated profit for given scenarios, Atlantic stock

DISCUSSION AND CONCLUSIONS

Although there are infinitely many scenarios that could have been examined, the chosen simulations demonstrate the possibility for reducing the amount of mercury that reaches consumers by altering the age composition of the commercially marketed catch. The Gulf and Atlantic simulations illustrate that it is even possible for this to occur without seriously impacting either commercial catch or the long-run stability of the biomass stock. Both the Atlantic and Gulf stock reductions in mercury came at the price of reduced fishery profits and losses in NPV, highlighting that some tradeoffs are necessary. While mercury levels under Scenario 6 in the Gulf were reduced by over 36%, to under half of the U.S. FDA limit (based on Table 4.3), those levels would still put virtually all consumers over the U.S. EPA RfD. Average mercury concentrations from Atlantic harvesting were reduced substantially from baseline levels under Scenario 6, down to a level of 0.38 ppm. This level of exposure would allow consumers over 210 lbs to safely eat one 6 ounce meal of king mackerel per week without exceeding the U.S. EPA's RfD. However, it should be noted that the U.S. EPA RfD is one of the most stringent

recommendations concerning mercury. In 2003, the World Health Organization revised its recommendation for safe intake levels to 1.6 micrograms per kg bodyweight per week, or approximately .23 micrograms per day (WHO 2003). The U.S. Agency for Toxic Substances and Disease Registry maintains that daily intake of methylmercury at a level of 0.3 micrograms per kilogram of body weight per day for a lifetime presents no risk of adverse health outcomes in even the most sensitive human populations (such as pregnant women, developing fetuses, and young children) -- ATSDR 1999). Under these guidelines, the simulated reductions can be viewed as substantial improvements.

Another issue worthy of exploration is how to transfer the model findings into real world management rules and regulations (Thunberg, Helser, and Mayo 1998). In the simulation scenarios, commercial fishing mortalities were changed, but the drivers behind those changes were not defined. Recall from the population dynamics model that the commercial fishing mortality can be separated into an age effect representing the selectivity of the fishery and a year effect representing intensity of the fishing mortality. Altering either of these effects will change fishing mortality at age. The intensity of fishing mortality can be altered through changes in TACs or by incorporating effort limitations. This will not generally reduce mercury exposure, however, because simply changing the overall fishing mortality without changing the age composition of the catch will not lead to an overall reduction in the contamination level of the marketed fish. Any policy or regulation must alter the selectivity patterns by age class of the fishery. This could be achieved in a number of ways, ranging from gear modifications to restrictions on times and areas fished (Thunberg, Helser, and Mayo 1998). Of course, area and seasonal restrictions will only be effective if the stock exhibits a distinct spatial or temporal distribution (Anderson 1977). Although king mackerel are known to form schools of similar sized individuals, further research will be needed to examine the spatial and seasonal distribution

of smaller-sized king mackerel to determine if area or seasonal restrictions can be used to shift fishing pressure towards younger age classes.

As illustrated by Scenario 6 for both the Atlantic and Gulf stocks, it seems that a harvesting slot limit, where all fish below the current minimum size limit and all fish above a maximum size limit are off-limits, could effectively reduce the mercury concentration that reaches consumers. When implemented to preserve stocks, however, size limits will only be effective if fish can be returned to the water unharmed or if size can be determined before capture (Anderson 1977). In this case, the slot limits would be implemented to reduce the amount of mercury reaching consumers, but would still require some ability to minimize incidental catch of larger fish in order to prevent depletion of the stock. The simulated scenarios show that slot limits are effective in reducing the average mercury in marketed fish, and when catches remain around historical levels, can also preserve the stock if bycatch is low. If bycatch of oversized fish was high enough, there could be a negative impact on biomass, jeopardizing the status and stability of the stock. Scenarios 5 and 6 show that minimizing bycatch is also necessary to limit losses to the commercial fisherman. For both stocks, losses in NPV were smallest under Scenario 5 which assumed perfect adherence to the slot limit with no incidental catch. Losses were considerably greater in Scenario 6 highlighting the importance of minimizing bycatch of larger fish and the cost associated with it.

Another potential issue, beyond the scope of the current study, is that since most king mackerel fishermen also target other species, policies implemented to alter the catch composition of king mackerel could alter the catch composition of other species as well. The king mackerel stocks are stable and recovered from overfishing. Additionally, all simulations resulted in commercial catches that were no larger, on average, than those currently occurring. Since both stocks are currently managed under a TAC that was designed to rebuild the stock, this is perhaps

the most important reason why the targeting patterns described in the simulations do not negatively impact stock status. If the stocks of other species that are targeted along with king mackerel are currently overfished or not yet rebuilt, inadvertently changing the catch composition of those species could negatively affect their stocks.

Finally, it is important to understand some of the limitations of this study and directions for future research. While the assumption of constant price does not seem unreasonable when looking at overall price levels for the catch, it would be preferable to incorporate price by age class to account for any differences in quality by size. Unfortunately, no data is available distinguishing king mackerel price by size or age class. More research is needed to determine if there are substantial price differences by age class. Additionally, if there are no current differences in price by age class, it is reasonable to think that in the future there could be based on the reduced mercury from the harvesting patterns proposed in this research. More work is needed to determine the amount (if any) of a price premium for lower mercury levels in king mackerel. If the price premium for smaller, less contaminated fish were substantial enough, the losses to the commercial fisherman's profit and the NPV of the fishery could be offset (to some degree) by the increase in revenue. This raises the possibility of a win-win situation and certainly warrants further investigation.

In addition to the assumption of constant price, this study also made use of a constant cost per unit of effort, in this case hours fished. The incorporation of cost into bioeconomic models of fisheries is usually problematic due to inadequate data. The cost data for this study came from self-reported logbook observations and accounted only for variable costs, including labor, fuel, bait, ice and miscellaneous costs. While most fisheries operate under a share system, the logbook data do not provide strong evidence of any relationship between reported labor costs and revenue. Many boats were also owner operated, with the captain as the sole crew member

on board, making it difficult to discern whether the fishery operated under a share system or a some sort of wage rate. This research also relied on the assumption that the cost structure of the fishery would not change as effort is reapportioned to younger age classes. This may not be realistic depending on how much effort is shifted. Future work is needed to develop a more comprehensive cost analysis of the king mackerel fishery, possibly involving personal interviews with fisherman in order to get a stronger understanding of the cost structures of their harvesting activities. Finally, if any gear or technology improvements are needed for the fleet to harvest smaller fish, those costs are not accounted for in the model. If those costs are large enough, the results of this study may be misleading, as the impact on the fishery in Scenario 6 would be greater than presented.

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CHAPTER 5: CONCLUSIONS

The presence of mercury and other contaminants in the U.S. fish supply is a growing public health concern. At high levels, these substances can be harmful to humans and ecosystems, and thus represent a growing threat not only to public health, but also to the economic and ecological viability of many fisheries. This research examined the economic issues surrounding fish contamination, developed a population dynamics and bioeconomic model to investigate the problem, and compared a variety of current and proposed management actions to reduce consumer exposure to contaminants.

Chapter 2 highlighted the public health and economic implications of contaminated U.S. fisheries, with a focus on harmful algal blooms and several primary persistent, bioaccumulative, and toxic contaminants present in the fishing supply, including mercury. An overview of contamination issues in U.S. fisheries was presented through a brief history of contamination problems, including a review of the public health impacts and the related economic costs experienced in the past. Management strategies for dealing with fish and shellfish contamination were investigated, along with an examination of the apparent efficacy of these actions and their implied economic cost to the fishing industry. This chapter served as the background and motivation for examining other alternatives to reduce the amount of contaminants reaching fish consumers.

The research continued with a focus on mercury contamination in pelagic fisheries. Given that mercury concentration is shown to increase with fish length, this study sought to examine the implications of harvesting smaller (and therefore less contaminated fish) through the development and application of an age-structured bioeconomic model. King mackerel, a particularly mercury-plagued species, was chosen as the specific fishery to be examined,

although the approach used could be applied to any pelagic (and even non-pelagic) fishery. The development of the bioeconomic model first required a realistic population dynamics model accounting for recruitment, mortality and growth of the fish stock. This model development was presented in Chapter 3, beginning with an overview of the king mackerel fishery and then constructing a multiple cohort population dynamics model for the U.S. king mackerel fishery, validating its use against information collected and analyzed during a recent king mackerel stock assessment. Model tracking of simulated versus actual values suggested that the model was adequate for use in future applied research involving king mackerel stocks.

Using the population dynamics model as a base, Chapter 4 incorporated the economic characteristics of the fishery, mercury contamination relationships, and exposure assumptions to create a comprehensive bioeconomic model. Forward simulations were then used to examine the plausibility of different management alternatives for the Gulf and Atlantic king mackerel stocks in the presence of mercury contamination. The simulations demonstrated the possibility of reducing the amount of mercury that reaches consumers by altering the age composition of the commercially marketed catch. Furthermore, the simulations illustrated that it may be possible for this to occur without seriously impacting the long-run stability of the biomass stock. However, there are tradeoffs in terms of the economic position of the fishery. In the case of both the Atlantic and Gulf stocks, reductions in mercury came at the price of reduced fishery profits and losses in NPV. The chapter concluded with a discussion of research limitations, how to transfer the model findings into regulatory actions through the implementation of a slot limit and some of the challenges that may be faced.

The underlying population dynamics model can be improved through the estimation of an alternative recruitment function or through the incorporation of a stochastic error term on the recruitment function. As alluded to in Chapter 4, more work is needed to determine the

possibility of a price premium for lower mercury levels in king mackerel, and a more thorough investigation into the king mackerel fishery cost structure could also improve the model. The manner in which the bioeconomic model is used can also be changed. In the current research, the model was used to simulate changes in average mercury concentration of harvested fish, stock biomass, commercial catch, profit, and the NPV of the fishery in response to changes in fishing mortality for different age classes. An additional use would be to dynamically optimize and analyze the model under various objectives (i.e., pure profit maximization, minimization of average mercury, profit maximization constrained by mercury limits and biomass limits) and the use of potential policy instruments. A comparison of the different optimization scenarios could then be used to generate policy relevant management suggestions under varying management objectives. Another area of future work should include a closer look into the simulation scenarios, and how changes in fishing mortality directly translates into changes in effort and catch in terms of numbers. It also may be interesting to apply a similar model to a mercury contaminated fishery that is both more widely consumed and for which there is more extensive demand data available, such as one of the tuna species. Given the depleted nature of many of the tuna stocks, it would be interesting to examine what economic and biological tradeoffs would be necessary to reduce the average mercury reaching consumers.

The bioaccumulative nature of mercury, and its multiple anthropogenic and natural sources, ensures that it will be present in our fish stocks for many years to come. Mercury exposure through food supplies will continue to remain a public health concern among consumers and potential consumers of seafood products. Currently, the amount of mercury reaching consumers is not considered in the harvesting decision, even when it is known that larger fish contain significantly higher amounts of mercury than smaller fish. This research demonstrated what might happen if attempts were made to reduce the mercury that reaches

consumers through the harvest of smaller fish, and it can be used as a base for further research that seeks to examine contaminant concentration and health concerns associated with fishery harvesting decisions.

APPENDIX A: LETTER OF PERMISSION

From: "Tina Willson" <twills2@lsu.edu>
To: oyb@dal.ca
Subject: Copyright question
Date: Thursday, April 03, 2008 2:59:02 PM

Dear Mr. Coffen-Smout,

My manuscript entitled "The Public Health and Economic Impacts of Contaminated U.S. Fisheries" was published in Ocean Yearbook Volume 21. This work was part of my dissertation research, and I am writing to find out what steps I must take in order to receive permission to include the article in my dissertation. My university, Louisiana State University (LSU), requires written permission from the copyright holder. If requested, access to the dissertation can be restricted to LSU campus viewing for 365 days from the date of its approval by the Graduate School or it can be completely withheld from all public access for a period of 1-2 years from its approval date. Thank you for your assistance.

Sincerely,

Tina Willson

From: "Gaby van Rietschoten" <rietschoten@brill.nl>
To: <twills2@lsu.edu>
Subject: FW: Copyright question
Date: Tuesday, April 08, 2008 4:16:56 AM

Dear Ms Willson,

Your permissions request has been forwarded to Brill's rights & permission department.

Koninklijke Brill NV gladly grants you permission to include your article Tina M. Willson, Richard F. Kazmierczak "The Public Health and Economic Impacts of Contaminated U.S. Fisheries" in 'Ocean Yearbook 21' Edited by Aldo Chircop, Scott Coffen-Smout, Moira McConnell, ISBN 9789004157552 (2007), pp. 307-337, in your dissertation, on condition of due acknowledgement of the original source of publication, its year and publisher.

I hope the above is sufficient for your purposes. Should you have any questions, please do not hesitate to contact me.

Yours sincerely,

BRILL
Gaby van Rietschoten (Ms)
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www.brill.nl

APPENDIX B: POPULATION DYNAMICS MODEL INPUTS

Table B.1 List of symbols used in population dynamics

Symbol	Description	Values
<i>a</i>	King Mackerel age in years	0,1,2,...11+
<i>t</i>	Time in years	Fishing years 1981-2006 or 1999-2006
<i>s</i>	Stock	Gulf, Atl
<i>M</i>	Instantaneous Natural Mortality	Table 3.1
<i>Mat</i>	Maturity Rate	Table 3.1
a,b	Beverton-Holt Recruitment parameters	Table 3.2
<i>W</i>	Weight in lbs	Atl: Table B.6 Gulf: Table B.7
<i>Fec</i>	Millions of Female Eggs	Table 3.1
<i>Fem</i>	Proportion Female	Assumed 0.50
<i>N</i>	Number of fish	Atl: Table B.2 Gulf: Table B.3
<i>F</i>	Instantaneous Fishing Mortality	Atl: Table B.4 Gulf: Table B.5
<i>Z</i>	Instantaneous Total Mortality	Calculated by Equation 3.2
<i>B</i>	Biomass	Calculated by Equation 3.5
<i>SSF</i>	Spawning Stock Fecindity	Calculated by Equation 3.4
<i>CN</i>	Catch in numbers	Calculated by Equation 3.6
<i>C</i>	Catch in lbs	Calculated by Equation 3.7
<i>FComm</i>	Instantaneous Commercial Fishing Mortality	Calculated by Equation 3.14
ρ	Commercial catch proportion	Atl: Table B.8 Gulf: Table B.9
<i>Frem</i>	Instantaneous Remaining Fishing Mortality	Calculated by Equation 3.8
<i>CommCN</i>	Commercial catch in numbers	Calculated by Equation 3.13
<i>CommCW</i>	Commercial catch in lbs	Calculated by Equation 3.12

Table B.2 King mackerel numbers at age 1981-2006, Atlantic stock

Source: SEDAR16 2009

Table B.3 King mackerel numbers at age 1981-2006, Gulf stock

	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9	Age 10	Age 11
1981	4487847	1326649	861939	1012106	737795	189395	120981	101667	75034	27830	151474	484493
1982	4004816	1718063	994313	647790	617960	426782	111854	75357	64959	54490	19795	522568
1983	1962239	1702742	1258719	620999	382417	317248	216456	33816	46564	38475	27253	294514
1984	6391395	604344	1214959	820621	403992	287193	220550	151849	22342	30735	28046	257191
1985	4472095	1994897	441621	901928	461274	214185	203392	138359	103319	14390	25013	224921
1986	3719424	1601811	1495910	297061	638561	310112	100747	139991	103663	69518	7592	194849
1987	5267802	1200690	1185966	989007	166502	487667	205611	47866	105442	84143	52216	157947
1988	7694286	1497823	826800	861890	763392	112368	373735	154996	32627	84149	67604	170180
1989	8987466	2478087	1098133	562895	611683	562929	64123	247268	102206	21825	58396	158840
1990	8467824	2417308	1777787	717915	378114	433635	415171	42078	185246	75701	14301	163097
1991	6431419	2583534	1747014	1250171	481547	241798	324481	309537	28381	137992	55916	132164
1992	5865852	1695455	1806143	1158514	887671	328121	162793	243875	246317	19700	98265	143690
1993	9795834	1779149	1232179	1239851	765914	629089	220600	92386	184200	175371	8127	166899
1994	10506494	2744740	1300452	837926	858086	501636	460885	149889	53888	132056	135624	115843
1995	9132397	2868587	1977142	883712	560265	552282	306086	319348	87556	22861	92933	145662
1996	6115982	2484441	2139385	1397203	563137	362978	395252	192793	237764	57312	12086	177187
1997	7689198	1951572	1813284	1464707	979570	379788	250364	294865	128906	173926	36465	120607
1998	6440792	2696715	1436676	1287173	991976	703203	247881	166470	204077	81343	126673	99371
1999	5795505	2218106	1999622	1022737	894578	655086	513488	169173	116538	155105	50018	176486
2000	5632798	1908428	1646700	1455704	735391	622509	458163	399235	116150	79884	109962	179312
2001	5702963	1917670	1395362	1173375	1009375	509835	453904	353422	302290	89258	54409	219408
2002	6593401	2146941	1415859	966131	809066	714954	356554	332788	255560	229681	67978	200082
2003	10790938	2645720	1570975	930713	657890	557157	521369	259325	249815	186526	179119	200684
2004	20369976	4009460	1987859	1098422	604167	447836	396512	378613	193091	186322	140688	291068
2005	16600606	7584351	3019205	1356615	764213	396355	319209	289619	281633	151806	135999	347110
2006	13425502	6469070	5745916	2223468	930270	511124	256400	216922	203424	208785	117227	375148

Source: SEDAR16 2009

Table B.4 Estimated instantaneous fishing mortality 1981-2006, Atlantic stock

	0	1	2	3	4	5	6	7	8	9	10	11
1981	0	0.017	0.026	0.285	0.442	0.262	0.413	0.018	0.054	0.006	0.103	0.103
1982	0.01	0.036	0.031	0.103	0.386	0.38	0.163	0.346	0.016	0.117	0.087	0.087
1983	0.024	0.127	0.101	0.175	0.113	0.189	0.381	0.055	0.064	0.002	0.077	0.077
1984	0.015	0.037	0.024	0.201	0.287	0.096	0.223	0.126	0.009	0.063	0.057	0.057
1985	0.045	0.029	0.113	0.16	0.441	0.393	0.054	0.061	0.069	0.029	0.022	0.022
1986	0	0.048	0.165	0.128	0.273	0.169	0.178	0.189	0.143	0.288	0.164	0.164
1987	0	0.11	0.094	0.106	0.145	0.159	0.208	0.103	0.128	0.149	0.147	0.147
1988	0	0.017	0.129	0.172	0.188	0.105	0.199	0.287	0.135	0.181	0.267	0.267
1989	0.004	0.044	0.104	0.127	0.115	0.158	0.196	0.154	0.219	0.142	0.203	0.203
1990	0.015	0.044	0.128	0.142	0.156	0.118	0.176	0.147	0.143	0.331	0.186	0.186
1991	0.014	0.032	0.132	0.207	0.194	0.213	0.19	0.248	0.311	0.216	0.283	0.283
1992	0.013	0.038	0.139	0.202	0.226	0.259	0.201	0.178	0.199	0.254	0.345	0.345
1993	0.008	0.039	0.092	0.105	0.111	0.133	0.173	0.212	0.201	0.222	0.318	0.318
1994	0.006	0.039	0.158	0.14	0.132	0.131	0.239	0.197	0.182	0.189	0.252	0.252
1995	0.01	0.043	0.17	0.179	0.208	0.121	0.177	0.188	0.289	0.361	0.241	0.241
1996	0.016	0.026	0.18	0.218	0.261	0.243	0.121	0.193	0.306	0.366	0.198	0.198
1997	0.007	0.046	0.151	0.195	0.265	0.247	0.27	0.204	0.235	0.39	0.296	0.296
1998	0.013	0.024	0.119	0.19	0.263	0.315	0.256	0.255	0.207	0.253	0.236	0.236
1999	0.017	0.031	0.139	0.123	0.196	0.191	0.233	0.181	0.22	0.146	0.152	0.152
2000	0.011	0.016	0.175	0.227	0.229	0.19	0.156	0.17	0.113	0.263	0.226	0.226
2001	0.002	0.015	0.095	0.137	0.285	0.214	0.222	0.163	0.124	0.133	0.238	0.238
2002	0.003	0.029	0.236	0.169	0.191	0.269	0.175	0.13	0.143	0.14	0.144	0.144
2003	0.003	0.012	0.146	0.23	0.264	0.268	0.358	0.27	0.18	0.113	0.138	0.138
2004	0.003	0.016	0.165	0.203	0.298	0.332	0.346	0.377	0.278	0.146	0.116	0.116
2005	0.003	0.009	0.141	0.183	0.208	0.284	0.257	0.253	0.344	0.243	0.089	0.089
2006	0.004	0.01	0.154	0.194	0.231	0.359	0.252	0.247	0.358	0.19	0.156	0.156

Source: SEDAR16 2009.

Table B.5 Estimated instantaneous fishing mortality 1981-2006, Gulf stock

	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9	Age 10	Age 11
1981	0.195	0.014	0.043	0.271	0.34	0.33	0.285	0.266	0.143	0.168	0.033	0.033
1982	0.09	0.037	0.228	0.305	0.46	0.483	1.008	0.3	0.347	0.52	0.448	0.448
1983	0.413	0.063	0.185	0.208	0.079	0.167	0.166	0.233	0.239	0.143	0.061	0.061
1984	0.4	0.04	0.055	0.354	0.427	0.149	0.278	0.203	0.263	0.033	0.075	0.075
1985	0.262	0.014	0.154	0.123	0.19	0.558	0.185	0.107	0.219	0.466	0.086	0.086
1986	0.366	0.027	0.171	0.357	0.062	0.215	0.556	0.102	0.032	0.113	0.086	0.086
1987	0.493	0.099	0.076	0.037	0.186	0.07	0.094	0.201	0.049	0.046	0.047	0.047
1988	0.368	0.036	0.142	0.121	0.097	0.365	0.225	0.235	0.225	0.192	0.239	0.239
1989	0.548	0.058	0.182	0.176	0.137	0.108	0.233	0.107	0.123	0.25	0.123	0.123
1990	0.422	0.051	0.109	0.177	0.24	0.094	0.105	0.212	0.118	0.13	0.132	0.132
1991	0.568	0.084	0.168	0.121	0.176	0.199	0.097	0.047	0.188	0.167	0.105	0.105
1992	0.428	0.045	0.133	0.192	0.137	0.201	0.378	0.099	0.163	0.713	0.206	0.206
1993	0.507	0.039	0.143	0.146	0.216	0.115	0.198	0.357	0.156	0.084	0.25	0.25
1994	0.533	0.054	0.144	0.181	0.233	0.298	0.179	0.356	0.681	0.178	0.38	0.38
1995	0.537	0.019	0.104	0.229	0.227	0.138	0.274	0.113	0.247	0.464	0.133	0.133
1996	0.377	0.041	0.136	0.133	0.187	0.175	0.105	0.221	0.136	0.279	0.288	0.288
1997	0.283	0.032	0.1	0.168	0.124	0.23	0.22	0.186	0.284	0.144	0.294	0.294
1998	0.301	0.025	0.097	0.142	0.208	0.118	0.194	0.175	0.098	0.313	0.081	0.081
1999	0.346	0.024	0.075	0.108	0.155	0.161	0.064	0.194	0.201	0.171	0.07	0.07
2000	0.313	0.039	0.096	0.144	0.159	0.12	0.071	0.096	0.086	0.211	0.111	0.111
2001	0.212	0.029	0.125	0.15	0.138	0.161	0.122	0.142	0.098	0.099	0.15	0.15
2002	0.148	0.038	0.177	0.162	0.166	0.119	0.13	0.105	0.138	0.076	0.126	0.126
2003	0.225	0.012	0.115	0.21	0.177	0.144	0.132	0.113	0.116	0.109	0.1	0.1
2004	0.223	0.01	0.139	0.141	0.214	0.142	0.126	0.114	0.064	0.142	0.054	0.054
2005	0.178	0.004	0.063	0.155	0.195	0.239	0.198	0.171	0.122	0.086	0.089	0.089
2006	0.103	0.004	0.036	0.107	0.208	0.263	0.287	0.243	0.164	0.091	0.087	0.08

Source: SEDAR16 2009

Table B.6 King mackerel weights (in lbs) at age 1981-2006, Atlantic stock

	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9	Age 10	Age 11
1981	0.528	3.3176	6.2986	8.5184	10.6392	12.771	15.1976	17.072	18.8144	20.4996	21.3818	25.08
1982	0.528	3.3176	6.2986	8.5184	10.6392	12.771	15.1976	17.072	18.8144	20.4996	21.3818	25.08
1983	0.528	3.3176	6.2986	8.5184	10.6392	12.771	15.1976	17.072	18.8144	20.4996	21.3818	25.08
1984	0.528	3.3176	6.2986	8.5184	10.6392	12.771	15.1976	17.072	18.8144	20.4996	21.3818	25.08
1985	0.528	3.3176	6.2986	8.5184	10.6392	12.771	15.1976	17.072	18.8144	20.4996	21.3818	25.08
1986	0.528	2.629	5.4802	7.7924	9.273	11.0242	12.7798	14.9336	16.2954	17.908	17.292	22.4334
1987	0.528	2.629	5.4802	7.7924	9.273	11.0242	12.7798	14.9336	16.2954	17.908	17.292	22.4334
1988	0.528	2.629	5.4802	7.7924	9.273	11.0242	12.7798	14.9336	16.2954	17.908	17.292	22.4334
1989	0.528	2.629	5.4802	7.7924	9.273	11.0242	12.7798	14.9336	16.2954	17.908	17.292	22.4334
1990	0.528	2.629	5.4802	7.7924	9.273	11.0242	12.7798	14.9336	16.2954	17.908	17.292	22.4334
1991	0.528	3.8302	6.2524	7.9376	9.8692	11.4378	13.6378	15.2526	16.588	18.5218	20.0816	24.2638
1992	0.528	3.8302	6.2524	7.9376	9.8692	11.4378	13.6378	15.2526	16.588	18.5218	20.0816	24.2638
1993	0.528	3.8302	6.2524	7.9376	9.8692	11.4378	13.6378	15.2526	16.588	18.5218	20.0816	24.2638
1994	0.528	3.8302	6.2524	7.9376	9.8692	11.4378	13.6378	15.2526	16.588	18.5218	20.0816	24.2638
1995	0.528	3.8302	6.2524	7.9376	9.8692	11.4378	13.6378	15.2526	16.588	18.5218	20.0816	24.2638
1996	0.528	3.399	6.578	9.1498	11.6446	13.882	16.3856	17.1182	19.3556	19.9474	22.5346	27.2272
1997	0.528	3.399	6.578	9.1498	11.6446	13.882	16.3856	17.1182	19.3556	19.9474	22.5346	27.2272
1998	0.528	3.399	6.578	9.1498	11.6446	13.882	16.3856	17.1182	19.3556	19.9474	22.5346	27.2272
1999	0.528	3.399	6.578	9.1498	11.6446	13.882	16.3856	17.1182	19.3556	19.9474	22.5346	27.2272
2000	0.528	3.399	6.578	9.1498	11.6446	13.882	16.3856	17.1182	19.3556	19.9474	22.5346	27.2272
2001	0.528	4.4946	6.7606	9.0706	11.1232	13.4926	16.2602	18.6604	20.823	24.1736	25.9072	27.3504
2002	0.528	4.4946	6.7606	9.0706	11.1232	13.4926	16.2602	18.6604	20.823	24.1736	25.9072	27.3504
2003	0.528	4.4946	6.7606	9.0706	11.1232	13.4926	16.2602	18.6604	20.823	24.1736	25.9072	27.3504
2004	0.528	4.4946	6.7606	9.0706	11.1232	13.4926	16.2602	18.6604	20.823	24.1736	25.9072	27.3504
2005	0.528	4.4946	6.7606	9.0706	11.1232	13.4926	16.2602	18.6604	20.823	24.1736	25.9072	27.3504
2006	0.528	3.3176	6.2986	8.5184	10.6392	12.771	15.1976	17.072	18.8144	20.4996	21.3818	25.08

Source: SEDAR16 2009

Table B.7 King mackerel weights (in lbs) at age 1981-2006, Gulf stock

	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9	Age 10	Age 11
1981	0.9328	4.0854	6.1974	8.415	10.615	13.211	15.5364	17.875	19.6724	22.0506	23.7292	28.237
1982	0.9328	4.0854	6.1974	8.415	10.615	13.211	15.5364	17.875	19.6724	22.0506	23.7292	28.237
1983	0.9328	4.0854	6.1974	8.415	10.615	13.211	15.5364	17.875	19.6724	22.0506	23.7292	28.237
1984	0.9328	4.0854	6.1974	8.415	10.615	13.211	15.5364	17.875	19.6724	22.0506	23.7292	28.237
1985	0.9328	4.0854	6.1974	8.415	10.615	13.211	15.5364	17.875	19.6724	22.0506	23.7292	28.237
1986	0.9328	3.1438	5.786	8.1334	10.8966	14.531	16.335	18.6186	20.6536	23.3222	23.7402	32.3994
1987	0.9328	3.1438	5.786	8.1334	10.8966	14.531	16.335	18.6186	20.6536	23.3222	23.7402	32.3994
1988	0.9328	3.1438	5.786	8.1334	10.8966	14.531	16.335	18.6186	20.6536	23.3222	23.7402	32.3994
1989	0.9328	3.1438	5.786	8.1334	10.8966	14.531	16.335	18.6186	20.6536	23.3222	23.7402	32.3994
1990	0.9328	3.1438	5.786	8.1334	10.8966	14.531	16.335	18.6186	20.6536	23.3222	23.7402	32.3994
1991	0.9328	3.9314	6.3096	8.5844	11.5126	14.1372	17.0698	18.9816	19.9738	22.187	24.585	26.741
1992	0.9328	3.9314	6.3096	8.5844	11.5126	14.1372	17.0698	18.9816	19.9738	22.187	24.585	26.741
1993	0.9328	3.9314	6.3096	8.5844	11.5126	14.1372	17.0698	18.9816	19.9738	22.187	24.585	26.741
1994	0.9328	3.9314	6.3096	8.5844	11.5126	14.1372	17.0698	18.9816	19.9738	22.187	24.585	26.741
1995	0.9328	3.9314	6.3096	8.5844	11.5126	14.1372	17.0698	18.9816	19.9738	22.187	24.585	26.741
1996	0.9328	4.3758	6.9652	8.6064	10.6524	12.9294	14.9644	18.3524	22.033	23.7226	25.9424	28.8266
1997	0.9328	4.3758	6.9652	8.6064	10.6524	12.9294	14.9644	18.3524	22.033	23.7226	25.9424	28.8266
1998	0.9328	4.3758	6.9652	8.6064	10.6524	12.9294	14.9644	18.3524	22.033	23.7226	25.9424	28.8266
1999	0.9328	4.3758	6.9652	8.6064	10.6524	12.9294	14.9644	18.3524	22.033	23.7226	25.9424	28.8266
2000	0.9328	4.3758	6.9652	8.6064	10.6524	12.9294	14.9644	18.3524	22.033	23.7226	25.9424	28.8266
2001	0.9328	4.851	5.94	8.2544	9.933	12.4168	14.0426	16.423	18.2842	19.6988	21.637	24.8072
2002	0.9328	4.851	5.94	8.2544	9.933	12.4168	14.0426	16.423	18.2842	19.6988	21.637	24.8072
2003	0.9328	4.851	5.94	8.2544	9.933	12.4168	14.0426	16.423	18.2842	19.6988	21.637	24.8072
2004	0.9328	4.851	5.94	8.2544	9.933	12.4168	14.0426	16.423	18.2842	19.6988	21.637	24.8072
2005	0.9328	4.851	5.94	8.2544	9.933	12.4168	14.0426	16.423	18.2842	19.6988	21.637	24.8072
2006	0.9328	4.0854	6.1974	8.415	10.615	13.211	15.5364	17.875	19.6724	22.0506	23.7292	28.237

Source: SEDAR16 2009

Table B.8 Assumed king mackerel commercial catch proportion by age 1999-2006, Atlantic stock

	1999	2000	2001	2002	2003	2004	2005	2006
Age 0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Age 1	0.09	0.00	0.03	0.07	0.01	0.10	0.01	0.01
Age 2	0.21	0.20	0.07	0.26	0.30	0.34	0.41	0.23
Age 3	0.22	0.15	0.17	0.10	0.11	0.24	0.21	0.36
Age 4	0.24	0.25	0.18	0.15	0.14	0.09	0.15	0.09
Age 5	0.09	0.16	0.16	0.11	0.19	0.06	0.06	0.13
Age 6	0.05	0.08	0.14	0.11	0.06	0.09	0.03	0.02
Age 7	0.02	0.03	0.06	0.07	0.09	0.02	0.05	0.02
Age 8	0.02	0.01	0.03	0.05	0.04	0.03	0.02	0.04
Age 9	0.02	0.02	0.02	0.02	0.02	0.01	0.03	0.01
Age 10	0.02	0.03	0.03	0.01	0.01	0.01	0.01	0.03
Age 11	0.02	0.05	0.11	0.06	0.03	0.01	0.02	0.07

Table B.9 Assumed king mackerel commercial catch proportion by age 1999-2006, Gulf stock

	1999	2000	2001	2002	2003	2004	2005	2006
Age 0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Age 1	0.11	0.07	0.06	0.20	0.11	0.07	0.07	0.04
Age 2	0.09	0.20	0.15	0.15	0.27	0.20	0.25	0.20
Age 3	0.17	0.21	0.18	0.16	0.17	0.19	0.16	0.20
Age 4	0.21	0.16	0.13	0.15	0.14	0.17	0.21	0.12
Age 5	0.15	0.13	0.13	0.07	0.11	0.11	0.12	0.15
Age 6	0.07	0.04	0.11	0.07	0.06	0.10	0.06	0.11
Age 7	0.03	0.08	0.09	0.07	0.03	0.05	0.05	0.07
Age 8	0.06	0.02	0.06	0.06	0.05	0.02	0.02	0.05
Age 9	0.06	0.02	0.01	0.03	0.02	0.04	0.02	0.02
Age 10	0.01	0.03	0.02	0.01	0.02	0.02	0.02	0.01
Age 11	0.03	0.05	0.06	0.04	0.03	0.02	0.04	0.05

Table B.10 King mackerel commercial catch (in thousands of lbs) calculated from ALS data based on 50% winter mixing zone assumption

Fyear	Atlantic	Gulf
1981	5886	3361
1982	3592	3567
1983	2885	2290
1984	2864	2519
1985	3693	2281
1986	2907	1835
1987	2872	1538
1988	2591	1499
1989	2674	1670
1990	2898	1309
1991	2864	2119
1992	3186	2402
1993	2681	2202
1994	2437	2026
1995	2662	2389
1996	3360	2586
1997	3030	2708
1998	3317	3236
1999	2645	2559
2000	2466	2503
2001	2315	2382
2002	2446	2684
2003	2653	2892
2004	2725	2554
2005	3203	3086
2006	2835	2939

APPENDIX C: BIOECONOMIC MODEL INPUTS

Table C.1 List of symbols used in the bioeconomic model

Symbol	Description	Value
a	King Mackerel age in years	0,1,2,...11+
t	Time in years	Fishing years 1999-2006
s	Stock	Gulf, Atl
$CommCW$	Commercial catch in lbs	Calculated by Equation 3.12
Rev	Revenue	Caalculated by Equation 4.1
P	Ex-vessel Price	\$1.49, assumed constant
c	Cost per unit of effort	\$25.60, assumed constant
E	Effort, defined by hours fished	Calculated from Equation 4.2
$Cost$	Variable cost of fishing	Calculated in Equation 4.3
q	catchability coefficient	36.36
ϕ	Catch- effort elasticity	0.5256
γ	Catch-stock elasticity	0.2948
π	Profit	Calculated by Equation 4.4
NPV	Net present value	Calculated by Equation 4.5
r	Discount rate	5%
L, K, t_0	Von Bertalannfy growth parameters	Table 4.2
Hg	Mercury concentration in ppm	Equations 4.7 and 4.8

APPENDIX D: MATLAB CODE FOR STATUS QUO SCENARIOS

```
%Program name: AtlBaseS1.m
%Description: Simulates Atlantic Scenario (1): Status Quo

%Set T=25 years(1999-2023)
T=25;
%Set age classes (0-11+)
A=12;

%Atlantic Natural Mortality at age
M=[0.672; 0.256; 0.220; 0.199; 0.186; 0.176; 0.170; 0.165; 0.161; 0.158; 0.156; 0.152];

%Calculated Atlantic Commercial Fishing Mortality
CF=[0.0000 0.0000 0.0000 0.0001 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000
0.0000 0.0001 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0001 0.0000
0.0000 0.0000 0.0000 0.0000;
0.0114 0.0007 0.0061 0.0080 0.0013 0.0119 0.0019 0.0016 0.0114 0.0007 0.0061
0.0080 0.0013 0.0119 0.0019 0.0016 0.0114 0.0007 0.0061 0.0080 0.0013 0.0119
0.0019 0.0016 0.0114;
0.0631 0.0285 0.0165 0.0834 0.0431 0.1052 0.0535 0.0719 0.0631 0.0285 0.0165
0.0834 0.0431 0.1052 0.0535 0.0719 0.0631 0.0285 0.0165 0.0834 0.0431 0.1052
0.0535 0.0719 0.0631;
0.0524 0.0558 0.0290 0.0349 0.0536 0.0911 0.0735 0.0791 0.0524 0.0558 0.0290
0.0349 0.0536 0.0911 0.0735 0.0791 0.0524 0.0558 0.0290 0.0349 0.0536 0.0911
0.0735 0.0791 0.0524;
0.0770 0.0696 0.0839 0.0409 0.0706 0.1212 0.0688 0.0545 0.0770 0.0696 0.0839
0.0409 0.0706 0.1212 0.0688 0.0545 0.0770 0.0696 0.0839 0.0409 0.0706 0.1212
0.0688 0.0545 0.0770;
0.0551 0.0634 0.0557 0.0914 0.0755 0.0932 0.0978 0.1026 0.0551 0.0634 0.0557
0.0914 0.0755 0.0932 0.0978 0.1026 0.0551 0.0634 0.0557 0.0914 0.0755 0.0932
0.0978 0.1026 0.0551;
0.0699 0.0582 0.0650 0.0639 0.0853 0.0994 0.0635 0.0477 0.0699 0.0582 0.0650
0.0639 0.0853 0.0994 0.0635 0.0477 0.0699 0.0582 0.0650 0.0639 0.0853 0.0994
0.0635 0.0477 0.0699;
```

0.0374	0.0556	0.0508	0.0512	0.0694	0.0897	0.0785	0.0544	0.0374	0.0556	0.0508
0.0512	0.0694	0.0897	0.0785	0.0544	0.0374	0.0556	0.0508	0.0512	0.0694	0.0897
0.0785	0.0544	0.0374;								
0.0608	0.0363	0.0675	0.0559	0.0455	0.0613	0.0858	0.1156	0.0608	0.0363	0.0675
0.0559	0.0455	0.0613	0.0858	0.1156	0.0608	0.0363	0.0675	0.0559	0.0455	0.0613
0.0858	0.1156	0.0608;								
0.0289	0.0719	0.0455	0.0543	0.0269	0.0340	0.0798	0.0595	0.0289	0.0719	0.0455
0.0543	0.0269	0.0340	0.0798	0.0595	0.0289	0.0719	0.0455	0.0543	0.0269	0.0340
0.0798	0.0595	0.0289;								
0.0323	0.0581	0.1580	0.0389	0.0388	0.0192	0.0288	0.1236	0.0323	0.0581	0.1580
0.0389	0.0388	0.0192	0.0288	0.1236	0.0323	0.0581	0.1580	0.0389	0.0388	0.0192
0.0288	0.1236	0.0323;								
0.0332	0.0584	0.0930	0.0671	0.0402	0.0231	0.0290	0.0905	0.0332	0.0584	0.0930
0.0671	0.0402	0.0231	0.0290	0.0905	0.0332	0.0584	0.0930	0.0671	0.0402	0.0231
0.0290	0.0905	0.0332];								

%Calculated Atlantic Recreational/Other Fishing Mortality

RF=[0.017	0.011	0.002	0.003	0.003	0.003	0.003	0.004	0.017	0.011	0.002
0.003	0.003	0.003	0.003	0.004	0.017	0.011	0.002	0.003	0.003	0.003
0.003	0.004	0.017;								
0.020	0.015	0.009	0.021	0.011	0.004	0.007	0.008	0.020	0.015	0.009
0.021	0.011	0.004	0.007	0.008	0.020	0.015	0.009	0.021	0.011	0.004
0.007	0.008	0.020;								
0.076	0.147	0.078	0.153	0.103	0.060	0.087	0.082	0.076	0.147	0.078
0.153	0.103	0.060	0.087	0.082	0.076	0.147	0.078	0.153	0.103	0.060
0.087	0.082	0.076;								
0.071	0.171	0.108	0.134	0.176	0.112	0.110	0.115	0.071	0.171	0.108
0.134	0.176	0.112	0.110	0.115	0.071	0.171	0.108	0.134	0.176	0.112
0.110	0.115	0.071;								
0.119	0.159	0.201	0.150	0.193	0.177	0.139	0.176	0.119	0.159	0.201
0.150	0.193	0.177	0.139	0.176	0.119	0.159	0.201	0.150	0.193	0.177
0.139	0.176	0.119;								
0.136	0.127	0.158	0.178	0.192	0.239	0.186	0.256	0.136	0.127	0.158
0.178	0.192	0.239	0.186	0.256	0.136	0.127	0.158	0.178	0.192	0.239
0.186	0.256	0.136;								

```

0.163  0.098  0.157  0.111  0.273  0.247  0.194  0.204  0.163  0.098  0.157
0.111  0.273  0.247  0.194  0.204  0.163  0.098  0.157  0.111  0.273  0.247
0.194  0.204  0.163;
0.144  0.114  0.112  0.079  0.201  0.287  0.174  0.193  0.144  0.114  0.112
0.079  0.201  0.287  0.174  0.193  0.144  0.114  0.112  0.079  0.201  0.287
0.174  0.193  0.144;
0.159  0.077  0.057  0.087  0.135  0.217  0.258  0.242  0.159  0.077  0.057
0.087  0.135  0.217  0.258  0.242  0.159  0.077  0.057  0.087  0.135  0.217
0.258  0.242  0.159;
0.117  0.191  0.087  0.086  0.086  0.112  0.163  0.130  0.117  0.191  0.087
0.086  0.086  0.112  0.163  0.130  0.117  0.191  0.087  0.086  0.086  0.112
0.163  0.130  0.117;
0.120  0.168  0.080  0.105  0.099  0.097  0.060  0.032  0.120  0.168  0.080
0.105  0.099  0.097  0.060  0.032  0.120  0.168  0.080  0.105  0.099  0.097
0.060  0.032  0.120;
0.119  0.168  0.145  0.077  0.098  0.093  0.060  0.065  0.119  0.168  0.145
0.077  0.098  0.093  0.060  0.065  0.119  0.168  0.145  0.077  0.098  0.093
0.060  0.065  0.119];

```

%Atlantic Maturity at age

```
Mat=[0.000; 0.548; 0.861; 0.924; 0.948; 0.970; 0.989; 1.000; 1.000; 1.000; 1.000; 1.000];
```

%Atlantic Fecundity (eggs) at age

```
Eggs=[0.000; 0.130; 0.250; 0.388; 0.528; 0.662; 0.783; 0.890; 0.981; 1.058; 1.123;
1.288];
```

%Atlantic initial numbers for Fishing Year 1999/2000

```
N(:,1)=[2622388;2674857;1131931;1402722;1036567;572242;227080;150576;93855;215569;184322;
168971];
```

%Atlantic Weights at age 1999-2003 (lbs)

```
W=[0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528
0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528  0.528
0.528  0.528  0.528;
3.399  3.399  4.4946  4.4946  4.4946  4.4946  4.4946  3.3176  3.3176  3.3176  3.3176
```

```

3.3176 3.3176 3.3176 3.3176 3.3176 3.3176 3.3176 3.3176 3.3176 3.3176 3.3176
3.3176 3.3176 3.3176;
6.578 6.578 6.7606 6.7606 6.7606 6.7606 6.7606 6.2986 6.2986 6.2986 6.2986
6.2986 6.2986 6.2986 6.2986 6.2986 6.2986 6.2986 6.2986 6.2986 6.2986 6.2986
6.2986 6.2986 6.2986;
9.1498 9.1498 9.0706 9.0706 9.0706 9.0706 9.0706 8.5184 8.5184 8.5184 8.5184
8.5184 8.5184 8.5184 8.5184 8.5184 8.5184 8.5184 8.5184 8.5184 8.5184 8.5184
8.5184 8.5184 8.5184;
11.6446 11.6446 11.1232 11.1232 11.1232 11.1232 11.1232 10.6392 10.6392 10.6392 10.6392
10.6392 10.6392 10.6392 10.6392 10.6392 10.6392 10.6392 10.6392 10.6392 10.6392 10.6392
10.6392 10.6392 10.6392;
13.882 13.882 13.4926 13.4926 13.4926 13.4926 13.4926 12.771 12.771 12.771 12.771
12.771 12.771 12.771 12.771 12.771 12.771 12.771 12.771 12.771 12.771 12.771
12.771 12.771 12.771;
16.3856 16.3856 16.2602 16.2602 16.2602 16.2602 16.2602 15.1976 15.1976 15.1976 15.1976
15.1976 15.1976 15.1976 15.1976 15.1976 15.1976 15.1976 15.1976 15.1976 15.1976 15.1976
15.1976 15.1976 15.1976;
17.1182 17.1182 18.6604 18.6604 18.6604 18.6604 18.6604 17.072 17.072 17.072 17.072
17.072 17.072 17.072 17.072 17.072 17.072 17.072 17.072 17.072 17.072 17.072
17.072 17.072 17.072;
19.3556 19.3556 20.823 20.823 20.823 20.823 20.823 18.8144 18.8144 18.8144 18.8144
18.8144 18.8144 18.8144 18.8144 18.8144 18.8144 18.8144 18.8144 18.8144 18.8144 18.8144
18.8144 18.8144 18.8144;
19.9474 19.9474 24.1736 24.1736 24.1736 24.1736 24.1736 20.4996 20.4996 20.4996 20.4996
20.4996 20.4996 20.4996 20.4996 20.4996 20.4996 20.4996 20.4996 20.4996 20.4996 20.4996
20.4996 20.4996 20.4996;
22.5346 22.5346 25.9072 25.9072 25.9072 25.9072 25.9072 21.3818 21.3818 21.3818 21.3818
21.3818 21.3818 21.3818 21.3818 21.3818 21.3818 21.3818 21.3818 21.3818 21.3818 21.3818
21.3818 21.3818 21.3818;
27.2272 27.2272 27.3504 27.3504 27.3504 27.3504 27.3504 25.08 25.08 25.08 25.08
25.08 25.08 25.08 25.08 25.08 25.08 25.08 25.08 25.08 25.08 25.08
25.08 25.08 25.08];

```

```
%Atlantic Recruitment parameters
```

```
Alpha=3.46*10^6;
```



```
Beta=6453;
```

```
%Population Dynamics
```

```
for a=1:A;
```

```
    F(a,1)=CF(a,1)+RF(a,1);
```

```
    Z(a,1)=F(a,1)+M(a);
```

```
    CN(a,1)=F(a,1)/Z(a,1)*N(a,1)*(1-exp(-Z(a,1)));
```

```
    CW(a,1)=CN(a,1)*W(a,1);
```

```
    CCN(a,1)=CF(a,1)/Z(a,1)*N(a,1)*(1-exp(-Z(a,1)));
```

```
    CCW(a,1)=CCN(a,1)*W(a,1);
```

```
    SSB(a,1)=N(a,1)*W(a,1)*Mat(a);
```

```
    B(a,1)=N(a,1)*W(a,1);
```

```
    SSF(a,1)=N(a,1)*Mat(a)*Eggs(a)*.5;
```

```
end
```

```
SF=sum(SSF,1);
```

```
for t=2:T;
```

```
    %Atlantic recruitment-Age 0
```

```
    N(1,t)=(Alpha*SF(t-1))/(Beta + SF(t-1));
```

```
    F(1,t)=CF(1,t)+RF(1,t);
```

```
    Z(1,t)=F(1,t)+M(1);
```

```
    CN(1,t)=F(1,t)/Z(1,t)*N(1,t)*(1-exp(-Z(1,t)));
```

```
    CW(1,t)=CN(1,t)*W(1,t);
```

```
    CCN(1,t)=CF(1,t)/Z(1,t)*N(1,t)*(1-exp(-Z(1,t)));
```

```
    CCW(1,t)=CCN(1,t)*W(1,t);
```

```
    B(1,t)=N(1,t)*W(1,t);
```

```
    SSB(1,t)=N(1,t)*W(1,t)*Mat(1);
```

```
    SSF(1,t)=N(1,t)*Eggs(1)*Mat(1)*.5;
```

```
%ages 1 to 10
```

```
    for a=2:A-1
```

```

F(a,t)=CF(a,t)+RF(a,t);
Z(a,t)=F(a,t)+M(a);
N(a,t)=N(a-1,t-1)*exp(-Z(a-1,t-1));
CN(a,t)=F(a,t)/(Z(a,t))*N(a,t)*(1-exp(-Z(a,t)));
CW(a,t)=CN(a,t)*W(a,t);
CCN(a,t)=CF(a,t)/(Z(a,t))*N(a,t)*(1-exp(-Z(a,t)));
CCW(a,t)=CCN(a,t)*W(a,t);
SSB(a,t)=N(a,t)*Mat(a)*W(a,t);
B(a,t)=N(a,t)*W(a,t);
SSB(a,t)=N(a,t)*W(a,t)*Mat(a);
SSF(a,t)=N(a,t)*Eggs(a)*Mat(a)*.5;
end

```

```

%Atlantic age 11

```

```

F(A,t)=CF(A,t)+RF(A,t);
Z(A,t)=F(A,t)+M(A);
N(A,t)=N(11,t-1)*exp(-Z(11,t-1))+N(A,t-1)*exp(-Z(A,t-1));
CN(A,t)=F(A,t)/(Z(A,t))*N(A,t)*(1-exp(-Z(A,t)));
CW(A,t)=CN(A,t)*W(A,t);
CCN(A,t)=CF(A,t)/(Z(A,t))*N(A,t)*(1-exp(-Z(A,t)));
CCW(A,t)=CCN(A,t)*W(A,t);
SSB(A,t)=N(A,t)*W(A,t)*Mat(A);
B(A,t)=N(A,t)*W(A,t);
SSF(A,t)=N(A,t)*Mat(A)*Eggs(A)*.5;

```

```

SB=sum(SSB,1);
Bt=sum(B,1);
SF=sum(SSF,1);
N_total=sum(N,1);
CN_total=sum(CN,1);
CW_total=sum(CW,1);
CCN_total=sum(CCN,1);
CCW_total=sum(CCW,1);

```

```

end

```

```

%Growth Parameters%
%VonBertalanffy Growth-Atlantic No Mix
Linf=114.1;
K=0.245;
t0=-1.689;

%Mercury Paramters
b1=3.43*10^-11;
b2=3.51;

%Mercury at Age
for a=1:A;
    %Generate length and mercury at age;
    FL_cm(a,1)=Linf*(1-exp(-K*((a-1)-t0)));
    FL_mm=FL_cm*10;
    HgLength(a,1)=b1*FL_mm(a,1)^b2;

end

%Mean Mercury
for t=1:T;
    for a=1:A;
        HgAge(a,t)=CCN(a,t)*HgLength(a);
        Hg_tot=sum(HgAge,1);
        MeanHg(t)=Hg_tot(t)/CCN_total(t);
    end
end

%Economic Model

%Constant Price
Price=1.49;

```

```

%Constant cost per hour fished
EffCost=25.60;

%Parameters to Calculate Effort(hours fished)
q=exp(3.59337);
smalla=.52561;
smallb=.29483;

%Discount rate
delta=.05;

for t=1:T;
    %Effort and Cost;
    Eff(t)=(CCW_total(t)/(q*(Bt(t))^smallb))^(1/smalla);
    Cost(t)=EffCost*Eff(t);

    %Revenue;
    TRev(t)=CCW_total(t)*Price;

    %NPV
    Profit(t)=TRev(t)-Cost(t);

    rho=1/(1+delta);

    NPV(t)=rho^t*Profit(t);
end

TNPV=sum(NPV);
%Program name: GulfBaseS1.m
%Description: Simulates Gulf Scenario (1): Status Quo

%Set T=25 years(1999-2023)
T=25;
%Set age classes (0-11+)
A=12;

```

%Gulf Natural Mortality at age

M=[0.765;0.274;0.243;0.222;0.207;0.196;0.188;0.182;0.177;0.173;0.17;0.162];

%Calculated Commercial Fishing Mortality Gulf

CF=[0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000
0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000 0.0000
0.0000 0.0000 0.0000 0.0000;
0.0109 0.0075 0.0062 0.0233 0.0108 0.0046 0.0021 0.0013 0.0109 0.0075 0.0062
0.0233 0.0108 0.0046 0.0021 0.0013 0.0109 0.0075 0.0062 0.0233 0.0108 0.0046
0.0021 0.0013 0.0109;
0.0102 0.0262 0.0232 0.0272 0.0475 0.0260 0.0196 0.0078 0.0102 0.0262 0.0232
0.0272 0.0475 0.0260 0.0196 0.0078 0.0102 0.0262 0.0232 0.0272 0.0475 0.0260
0.0196 0.0078 0.0102;
0.0363 0.0314 0.0333 0.0433 0.0510 0.0452 0.0288 0.0209 0.0363 0.0314 0.0333
0.0433 0.0510 0.0452 0.0288 0.0209 0.0363 0.0314 0.0333 0.0433 0.0510 0.0452
0.0288 0.0209 0.0363;
0.0530 0.0471 0.0278 0.0481 0.0612 0.0763 0.0668 0.0305 0.0530 0.0471 0.0278
0.0481 0.0612 0.0763 0.0668 0.0305 0.0530 0.0471 0.0278 0.0481 0.0612 0.0763
0.0668 0.0305 0.0530;
0.0504 0.0468 0.0525 0.0261 0.0521 0.0647 0.0750 0.0724 0.0504 0.0468 0.0525
0.0261 0.0521 0.0647 0.0750 0.0724 0.0504 0.0468 0.0525 0.0261 0.0521 0.0647
0.0750 0.0724 0.0504;
0.0272 0.0164 0.0500 0.0462 0.0302 0.0667 0.0468 0.1041 0.0272 0.0164 0.0500
0.0462 0.0302 0.0667 0.0468 0.1041 0.0272 0.0164 0.0500 0.0462 0.0302 0.0667
0.0468 0.1041 0.0272;
0.0435 0.0421 0.0548 0.0487 0.0292 0.0305 0.0419 0.0768 0.0435 0.0421 0.0548
0.0487 0.0292 0.0305 0.0419 0.0768 0.0435 0.0421 0.0548 0.0487 0.0292 0.0305
0.0419 0.0768 0.0435;
0.1059 0.0307 0.0435 0.0584 0.0484 0.0291 0.0150 0.0589 0.1059 0.0307 0.0435
0.0584 0.0484 0.0291 0.0150 0.0589 0.1059 0.0307 0.0435 0.0584 0.0484 0.0291
0.0150 0.0589 0.1059;
0.0855 0.0687 0.0153 0.0305 0.0339 0.0501 0.0267 0.0188 0.0855 0.0687 0.0153
0.0305 0.0339 0.0501 0.0267 0.0188 0.0855 0.0687 0.0153 0.0305 0.0339 0.0501
0.0267 0.0188 0.0855;

0.0422 0.0630 0.0791 0.0330 0.0287 0.0294 0.0305 0.0222 0.0422 0.0630 0.0791
 0.0330 0.0287 0.0294 0.0305 0.0222 0.0422 0.0630 0.0791 0.0330 0.0287 0.0294
 0.0305 0.0222 0.0422;
 0.0361 0.0579 0.0531 0.0515 0.0335 0.0177 0.0249 0.0289 0.0361 0.0579 0.0531
 0.0515 0.0335 0.0177 0.0249 0.0289 0.0361 0.0579 0.0531 0.0515 0.0335 0.0177
 0.0249 0.0289 0.0361];

%Calculated Recreational/Other Fishing Mortality Gulf

RF=[0.3460 0.3130 0.2120 0.1480 0.2250 0.2230 0.1780 0.1030 0.3460 0.3130
 0.2120 0.1480 0.2250 0.2230 0.1780 0.1030 0.3460 0.3130 0.2120 0.1480 0.2250
 0.2230 0.1780 0.1030 0.3460;
 0.0131 0.0315 0.0228 0.0147 0.0012 0.0054 0.0019 0.0027 0.0131 0.0315 0.0228
 0.0147 0.0012 0.0054 0.0019 0.0027 0.0131 0.0315 0.0228 0.0147 0.0012 0.0054
 0.0019 0.0027 0.0131;
 0.0648 0.0698 0.1018 0.1498 0.0675 0.1130 0.0434 0.0282 0.0648 0.0698 0.1018
 0.1498 0.0675 0.1130 0.0434 0.0282 0.0648 0.0698 0.1018 0.1498 0.0675 0.1130
 0.0434 0.0282 0.0648;
 0.0717 0.1126 0.1167 0.1187 0.1590 0.0958 0.1262 0.0861 0.0717 0.1126 0.1167
 0.1187 0.1590 0.0958 0.1262 0.0861 0.0717 0.1126 0.1167 0.1187 0.1590 0.0958
 0.1262 0.0861 0.0717;
 0.1020 0.1119 0.1102 0.1179 0.1158 0.1377 0.1282 0.1775 0.1020 0.1119 0.1102
 0.1179 0.1158 0.1377 0.1282 0.1775 0.1020 0.1119 0.1102 0.1179 0.1158 0.1377
 0.1282 0.1775 0.1020;
 0.1106 0.0732 0.1085 0.0929 0.0919 0.0773 0.1640 0.1906 0.1106 0.0732 0.1085
 0.0929 0.0919 0.0773 0.1640 0.1906 0.1106 0.0732 0.1085 0.0929 0.0919 0.0773
 0.1640 0.1906 0.1106;
 0.0368 0.0546 0.0720 0.0838 0.1018 0.0593 0.1512 0.1829 0.0368 0.0546 0.0720
 0.0838 0.1018 0.0593 0.1512 0.1829 0.0368 0.0546 0.0720 0.0838 0.1018 0.0593
 0.1512 0.1829 0.0368;
 0.1505 0.0539 0.0872 0.0563 0.0838 0.0835 0.1291 0.1662 0.1505 0.0539 0.0872
 0.0563 0.0838 0.0835 0.1291 0.1662 0.1505 0.0539 0.0872 0.0563 0.0838 0.0835
 0.1291 0.1662 0.1505;
 0.0951 0.0553 0.0545 0.0796 0.0676 0.0349 0.1070 0.1051 0.0951 0.0553 0.0545
 0.0796 0.0676 0.0349 0.1070 0.1051 0.0951 0.0553 0.0545 0.0796 0.0676 0.0349
 0.1070 0.1051 0.0951];

```

0.0855 0.1423 0.0837 0.0455 0.0751 0.0919 0.0593 0.0722 0.0855 0.1423 0.0837
0.0455 0.0751 0.0919 0.0593 0.0722 0.0855 0.1423 0.0837 0.0455 0.0751 0.0919
0.0593 0.0722 0.0855;
0.0278 0.0480 0.0709 0.0930 0.0713 0.0246 0.0585 0.0648 0.0278 0.0480 0.0709
0.0930 0.0713 0.0246 0.0585 0.0648 0.0278 0.0480 0.0709 0.0930 0.0713 0.0246
0.0585 0.0648 0.0278;
0.0339 0.0531 0.0969 0.0745 0.0665 0.0363 0.0641 0.0581 0.0339 0.0531 0.0969
0.0745 0.0665 0.0363 0.0641 0.0581 0.0339 0.0531 0.0969 0.0745 0.0665 0.0363
0.0641 0.0581 0.0339];

```

%Gulf Maturity at Age

```
Mat=[0; .157; .529;.704; .856; .989; 1; 1; 1; 1; 1;1];
```

%Gulf Fecundity (eggs) at Age

```
Eggs=[0; 0.155; 0.267; 0.395; 0.531; .669; .801; .926; 1.041; 1.145; 1.238; 1.524];
```

%Gulf initial numbers Fishing Year 1999/2000

```
N(:,1)=[5795505;2218106;1999622;1022737;894578;655086;513488;169173;116538;155105;50018;176486];
```

%Gulf Weights at age 1999-2023 (lbs)

```
W=[0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328
0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328 0.9328
0.9328 0.9328 0.9328 0.9328;
4.3758 4.3758 4.851 4.851 4.851 4.851 4.851 4.0854 4.0854 4.0854 4.0854
4.0854 4.0854 4.0854 4.0854 4.0854 4.0854 4.0854 4.0854 4.0854 4.0854
4.0854 4.0854 4.0854;
6.9652 6.9652 5.94 5.94 5.94 5.94 5.94 6.1974 6.1974 6.1974 6.1974
6.1974 6.1974 6.1974 6.1974 6.1974 6.1974 6.1974 6.1974 6.1974 6.1974
6.1974 6.1974 6.1974;
8.6064 8.6064 8.2544 8.2544 8.2544 8.2544 8.2544 8.415 8.415 8.415 8.415
8.415 8.415 8.415 8.415 8.415 8.415 8.415 8.415 8.415 8.415
8.415 8.415 8.415;
10.6524 10.6524 9.933 9.933 9.933 9.933 9.933 10.615 10.615 10.615 10.615
10.615 10.615 10.615 10.615 10.615 10.615 10.615 10.615 10.615 10.615
```

```

10.615 10.615 10.615;
12.9294 12.9294 12.4168 12.4168 12.4168 12.4168 12.4168 13.211 13.211 13.211 13.211
13.211 13.211 13.211 13.211 13.211 13.211 13.211 13.211 13.211 13.211 13.211
13.211 13.211 13.211;
14.9644 14.9644 14.0426 14.0426 14.0426 14.0426 14.0426 15.5364 15.5364 15.5364 15.5364
15.5364 15.5364 15.5364 15.5364 15.5364 15.5364 15.5364 15.5364 15.5364 15.5364 15.5364
15.5364 15.5364 15.5364;
18.3524 18.3524 16.423 16.423 16.423 16.423 16.423 17.875 17.875 17.875 17.875
17.875 17.875 17.875 17.875 17.875 17.875 17.875 17.875 17.875 17.875 17.875
17.875 17.875 17.875;
22.033 22.033 18.2842 18.2842 18.2842 18.2842 18.2842 19.6724 19.6724 19.6724 19.6724
19.6724 19.6724 19.6724 19.6724 19.6724 19.6724 19.6724 19.6724 19.6724 19.6724 19.6724
19.6724 19.6724 19.6724;
23.7226 23.7226 19.6988 19.6988 19.6988 19.6988 19.6988 22.0506 22.0506 22.0506 22.0506
22.0506 22.0506 22.0506 22.0506 22.0506 22.0506 22.0506 22.0506 22.0506 22.0506 22.0506
22.0506 22.0506 22.0506;
25.9424 25.9424 21.637 21.637 21.637 21.637 21.637 23.7292 23.7292 23.7292 23.7292
23.7292 23.7292 23.7292 23.7292 23.7292 23.7292 23.7292 23.7292 23.7292 23.7292 23.7292
23.7292 23.7292 23.7292;
28.8266 28.8266 24.8072 24.8072 24.8072 24.8072 24.8072 28.237 28.237 28.237 28.237
28.237 28.237 28.237 28.237 28.237 28.237 28.237 28.237 28.237 28.237 28.237
28.237 28.237 28.237];

```

```

%Gulf Recruitment parameters

```

```

Alpha=7.78*10^6;

```

```

Beta=11721;

```

```

%Population Dynamics

```

```

for a=1:A;

```

```

    F(a,1)=CF(a,1)+RF(a,1);

```

```

    Z(a,1)=F(a,1)+M(a);

```

```

    CN(a,1)=F(a,1)/Z(a,1)*N(a,1)*(1-exp(-Z(a,1)));

```

```

    CW(a,1)=CN(a,1)*W(a,1);

```

```

    CCN(a,1)=CF(a,1)/Z(a,1)*N(a,1)*(1-exp(-Z(a,1)));

```

```

    CCW(a,1)=CCN(a,1)*W(a,1);

```



```

SSB(a,1)=N(a,1)*W(a,1)*Mat(a);
B(a,1)=N(a,1)*W(a,1);
SSF(a,1)=N(a,1)*Mat(a)*Eggs(a)*.5;

end

SF=sum(SSF,1);

for t=2:T;
    %Gulf recruitment-Age 0
    N(1,t)=(Alpha*SF(t-1))/(Beta + SF(t-1));

    F(1,t)=CF(1,t)+RF(1,t);
    Z(1,t)=F(1,t)+M(1);
    CN(1,t)=F(1,t)/Z(1,t)*N(1,t)*(1-exp(-Z(1,t)));
    CW(1,t)=CN(1,t)*W(1,t);
    CCN(1,t)=CF(1,t)/Z(1,t)*N(1,t)*(1-exp(-Z(1,t)));
    CCW(1,t)=CCN(1,t)*W(1,t);
    B(1,t)=N(1,t)*W(1,t);
    SSB(1,t)=N(1,t)*W(1,t)*Mat(1);
    SSF(1,t)=N(1,t)*Eggs(1)*Mat(1)*.5;

    %ages 1 to 10
    for a=2:A-1
        F(a,t)=CF(a,t)+RF(a,t);
        Z(a,t)=F(a,t)+M(a);
        N(a,t)=N(a-1,t-1)*exp(-Z(a-1,t-1));
        CN(a,t)=F(a,t)/(Z(a,t))*N(a,t)*(1-exp(-Z(a,t)));
        CW(a,t)=CN(a,t)*W(a,t);
        CCN(a,t)=CF(a,t)/(Z(a,t))*N(a,t)*(1-exp(-Z(a,t)));
        CCW(a,t)=CCN(a,t)*W(a,t);
        SSB(a,t)=N(a,t)*Mat(a)*W(a,t);
        B(a,t)=N(a,t)*W(a,t);
        SSB(a,t)=N(a,t)*W(a,t)*Mat(a);
        SSF(a,t)=N(a,t)*Eggs(a)*Mat(a)*.5;
    end
end

```

end

%Gulf age 11

```
F(A,t)=CF(A,t)+RF(A,t);
Z(A,t)=F(A,t)+M(A);
N(A,t)=N(11,t-1)*exp(-Z(11,t-1))+N(A,t-1)*exp(-Z(A,t-1));
CN(A,t)=F(A,t)/(Z(A,t))*N(A,t)*(1-exp(-Z(A,t)));
CW(A,t)=CN(A,t)*W(A,t);
CCN(A,t)=CF(A,t)/(Z(A,t))*N(A,t)*(1-exp(-Z(A,t)));
CCW(A,t)=CCN(A,t)*W(A,t);
SSB(A,t)=N(A,t)*W(A,t)*Mat(A);
B(A,t)=N(A,t)*W(A,t);
SSF(A,t)=N(A,t)*Mat(A)*Eggs(A)*.5;
```

```
SB=sum(SSB,1);
Bt=sum(B,1);
SF=sum(SSF,1);
N_total=sum(N,1);
CN_total=sum(CN,1);
CW_total=sum(CW,1);
CCN_total=sum(CCN,1);
CCW_total=sum(CCW,1);
```

end

%Growth Parameters%

%VonBertalanffy Growth-Gulf No Mix

Linf=122.4;

K=0.177;

t0=-2.651;

%Mercury Paramters

b1=-3.09;

```

b2=.0032;

%Mercury at Age

    for a=1:A;
        %Generate length and mercury at age;
        FL_cm(a,1)=Linf*(1-exp(-K*((a-1)-t0)));
        FL_mm=FL_cm*10;
        HgLength(a,1)=exp(b1+b2*FL_mm(a,1));
    end

%Mean Mercury

for t=1:T;
    for a=1:A;
        HgAge(a,t)=CCN(a,t)*HgLength(a);
        Hg_tot=sum(HgAge,1);
        MeanHg(t)=Hg_tot(t)/CCN_total(t);
    end
end

%Economic Model

%Constant Price
Price=1.49;

%Constant cost per hour fished
EffCost=25.60;

%Parameters to Calculate Effort (hours fished)
q=exp(3.59337);
smalla=.52561;
smallb=.29483;

```

```

delta=.05;
for t=1:T;

    %Effort and Cost;
    Eff(t)=(CCW_total(t)/(q*(Bt(t))^smallb))^(1/smalla);
    Cost(t)=EffCost*Eff(t);

    %Revenue;
    TRev(t)=CCW_total(t)*Price;

    %NPV
    Profit(t)=TRev(t)-Cost(t);

    rho=1/(1+delta);

    NPV(t)=rho^t*Profit(t);
end

TNPV=sum(NPV);

```

VITA

Tina Marie Willson was born in Erie, Pennsylvania. Tina graduated with honors from Seneca High School in 1997. She subsequently attended the Pennsylvania State University in Erie, where she earned a bachelor's degree in mathematics and graduated with distinction in 2001. Upon graduation, Tina married her husband, Nate, and moved to Albany, New York, to begin graduate studies at the University of Albany School of Public Health. She graduated in December 2002 with a master of science in biostatistics. In January 2003, she entered the doctoral program in the Department of Agricultural Economics and Agribusiness at Louisiana State University, and is currently a candidate for the degree of Doctor of Philosophy. Upon completion of her doctoral studies, Tina will begin a post-doctoral fellowship in the Department of Agricultural and Applied Economics at the University of Wyoming.