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The Effects of Water Quality on Coastal Recreation Flounder Fishing

Matt Massey, Steve Newbold, and Brad Gentner

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Abstract

This paper describes a bioeconomic model of a coastal recreational fishery that combines standard models of fish population dynamics, recreational catch, and recreation site choice. The population model estimates the influence of water quality on overall fish abundance through the effects of dissolved oxygen (DO) on the survivorship of young juvenile fish. The catch model estimates the influence of fish abundance and water quality on anglers' average catch rates. The recreation demand model estimates welfare effects and changes in trip demand from changes in catch rates. The bioeconomic model also accounts for the feedback on the fish population through changes in the overall harvest pressure in the recreational fishery on the fish stock. The population model is specified using data on survival and reproduction from the fisheries science literature and government reports, and the model is calibrated using average historic recreational harvest levels in and out of the study area and historic commercial harvest levels for the entire fishery. The catch model is estimated using data on a sample of anglers who fished for summer flounder, data on water quality conditions from 23 water quality monitoring stations, and fishery-independent data on fish abundance collected in bottom trawl surveys, all in Maryland's coastal bays in 2002. The recreation demand model is estimated using data from a stated choice survey of anglers who fish for summer flounder on the Atlantic coast. The bioeconomic model is used to estimate the aggregate benefits to recreational anglers from several illustrative scenarios of changes in water quality. The results indicate that improving water quality throughout the range of the species could lead to substantial increases in the fish population and associated benefits to recreational anglers from increased catch rates. Water quality improvements confined to Maryland's coastal bays alone would have much smaller impacts.

Keywords: bioeconomic model, dissolved oxygen, recreation demand, recreational fishing, summer flounder, water quality

Subjects: water pollution, recreation, valuation methods

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1 Introduction

There is a substantial body of research on the effects of water quality on recreational angling. However, most studies focus on a single element in the chain of effects that connect water quality changes to the welfare of anglers, or they use a reduced form approach where some of the key variables are not accounted for explicitly. The result is a large number of studies that as a group indicate improvements in water quality conditions may lead to substantial benefits for recreational anglers, but individually are difficult to combine to evaluate specific water quality policies in a comprehensive manner.

Water quality may affect the site choices, trip demands, and ultimately the welfare of anglers through several pathways. First, water quality may directly affect the size of the stock of fish available for fishermen to catch through its impacts on the rates of reproduction and survival in the fish population. Second, water quality may directly affect the average catch rate of anglers through its impacts on fish movements and feeding activities. Third, water quality may directly affect the site choices and trip demands of anglers through either the potential health risks or the effects on the aesthetic characteristics of the water bodies used for recreational fishing.

In this paper we develop and apply a bioeconomic framework for valuing water quality changes in a coastal recreational fishery. The framework combines independently estimated models of each of the key processes by which water quality affects recreational anglers. We use a structural modeling approach rather than a reduced form approach because it allows us to take advantage of a variety of data sources, and it because it can provide more flexibility for evaluating a wider variety of water quality policies than most previous valuation models.

2 A stylized bioeconomic model of a recreational fishery

We begin by laying out a stylized bioeconomic model that describes how a recreational fishery evolves over time in response to water quality conditions and angler activities. The stylized model isolates the several pathways by which water quality may affect anglers, and it will provide a convenient organizational structure for discussing previous research on recreational fishing and the empirical application in this paper. First, water quality may affect fish abundance through its impact on the reproductive and survival rates of the fish population. Thus, the abundance of fish in time period *t*, A_t, will be a function of the abundance, the harvest, H_{t-1}, and water quality conditions, $\mathbf{W}_{t-1}$, in the previous time period:

$$A_t = f_1(A_{t-1}, H_{t-1}, \mathbf{W}_{t-1}) \qquad [1]$$

In addition to this long term "abundance effect," water quality also may influence anglers' catch rates through short term effects on fish movements and feeding activities. Thus, the average catch per trip for angler *i* in time period *t*, C_{it}, will be a function of the angler's skill and other personal characteristics, $\mathbf{Z}_i$, the current fish abundance, and water quality conditions:

$$C_{it} = f_2(\mathbf{Z}_i, A_t, \mathbf{W}_t) \qquad [2]$$

Independent of the "abundance effect" and the "catchability effect," water quality also may have a direct "site choice effect" either through potential health risks from exposure to the water or from eating contaminated fish, or through the aesthetic appeal of the sites from the affect of water quality on the appearance or odor of the water bodies. Thus, the number of fishing trips angler *i* takes in time period *t*, T_{it}, will depend on the angler's avidity and other personal characteristics, and the current (expected) catch and water quality conditions:

$$T_{it} = f_3(\mathbf{Z}_i, C_{it}, \mathbf{W}_t) \qquad [3]$$

The total harvest in period *t*, H_t, is simply the average catch multiplied by the number of trips summed over all anglers:

$$H_t = \sum_i C_{it} T_{it} \qquad [4]$$

Therefore, the average catch rate and the aggregate trip demand will determine the total recreational harvest level, which in turn will affect the abundance of fish that can spawn to produce fish in future years (H_t will appear in the expression for A_{t+1} as in Equation [1]), which will affect the catch rate in future years, which will affect trip demand… and so on.

3 Previous research

Previous economic studies of recreational angling can be organized in terms of the stylized model presented above. First, a number of researchers estimate some version of Equation [3] using random utility models of recreation demand to investigate the effects of water quality conditions on anglers' site choices. There are numerous studies that use measures of C_{it} as site attributes (see the reviews in Freeman [1993] and Van Houtven et al. [2001]). However, only a handful of studies have included measures of both C_{it} and $\mathbf{W}_t$ as site attributes. Examples of studies that include both catch rates and water quality include Jakus et al. (1997, 1998), who use the presence of fish consumption advisories and average catch rates as site characteristics, and Kaoru (1995), who uses estimated measures of nitrogen and phosphorus discharge, biochemical oxygen demand, and suspended solids, along with a measure of average catch.

To measure differences in catch rates across sites, most authors have used the average historic catch rates at different sites estimated either from the in-hand sample of recreators or from external sources of information such as the Marine Recreational Fishing Statistics Survey (MRFSS). In cases where no direct measures of water quality are included, catch rates themselves may be thought of as serving as a proxy for water quality conditions; higher catch rates are assumed to be at least partly the result of better water quality. Strand et al. (1991), Kaoru (1995), Jakus et al. (1997), Hicks et al. (1999), and McConnell and Strand (1999) are all employ variations of this strategy. However, there are at least two limitations to using average historic catch across all anglers as a proxy for actual or expected catch for individual anglers: it assumes catch rates at a site will be equal for all anglers at all times, and it provides no explicit linkage between catch rate and water quality.

To overcome some of the limitations of using the traditional measures of catch rates across sites, a number of researchers estimate some version of Equation [2] by modeling catch rate as a function of angler and site characteristics.[1] Most of these studies use one or more measures of $\mathbf{W}_{it}$ as explanatory variables, but we are aware of none that use a direct measure of A_t. Furthermore, most of these studies use temporally and spatially aggregated measures of water quality conditions, either single point-in-time measures or annual averages, which ignore the potential correlations between intra-annual variations in water quality conditions and catch

[1] A third approach to characterizing expected catch across sites is to use subjective ratings or indices of fishing quality, usually created by fisheries experts. This method is the least common in the literature, presumably due to the difficulties of constructing valid subjective measures.

rates . For example, Smith et al. (1993) use a household production framework to model expected catch. Landed fish are treated as a good produced by combined inputs of human capital (hours fished, fisherman skill, etc.) and environmental inputs (water quality). Kaoru et al. (1995), Englin et al. (1997), and Lipton and Hicks (2003) use similar approaches.[2] Both Smith et al. (1993) and Kaoru et al. (1995) include annual county level estimates of biochemical oxygen demand and nitrogen loadings as measures of water quality. Englin et al. (1997) relies on measurements of dissolved oxygen and turbidity taken at a single point in time for each site. Some recent studies have used more temporally detailed measures of water quality conditions. Lipton and Hicks (2003) use monthly and biweekly reported dissolved oxygen and water temperature readings from the closest date immediately proceeding the fishing trip. However, because information was not available on the anglers' actual fishing locations, measures of water quality from the monitoring station closest to the where each angler was intercepted and interviewed were assumed to represent the water quality conditions faced while fishing. As a result, if anglers did not fish nearby their intercept locations there remains a potential spatial mismatch between the water quality measures and the anglers' reported catch. All studies save Smith et al. (1993) use their predicted catch as site characteristics in random utility models of fishing trip demands, thereby linking Equation [2] and Equation [3].

The only studies of which we are aware that use an objective measure of fish abundance is the series by Cameron (1989, 1990, 1992) on the effects of water quality on the non-market value of the marine recreational fishery along the Texas Gulf Coast. One must look outside of the economics literature for research on the direct effect of water quality on fish, and most

[2] McConnell and Strand (1994), McConnell et al. (1995), and Jakus et al. (1998) also model expected catch rates but do not include any measures of water quality as explanatory variables.

bioeconomic models of either commercial or recreational fisheries have been used to evaluate harvest policies, not water quality changes. We are aware of no studies that estimate multiple models to operationalize the entire system of Equations [1] through [4].

The goal of this paper is to construct a bioeconomic model that integrates the effects of water quality on fish populations, expected catch, and trip demand. We estimate empirical versions of Equations [1] through [4] using relatively detailed data on fish abundance, water quality, and angler preferences and activities over the course of a season and across several sites. We then use the model to value water quality changes in an important recreational fishery in a local area on the U.S. Atlantic coast.

4 Application: summer flounder in Maryland's coastal bays

Summer flounder on the east coast of the U.S. spend their first one or two years as juveniles in coastal bays and estuaries mostly between Cape Hatteras, NC and Long Island, NY (Able and Kaiser 1994). The adults spend the winter months and breed in the open ocean along the continental shelf, during which time they are subject to a large commercial fishery (average commercial landings between 1982 and 2001 was 8,500 metric tons [NFSC 2002b]). A large portion of the adult stock spends the summer months along with the juveniles feeding in the bays and estuaries (Rogers and Van Den Avyle 1983; Kraus and Musick 2001), when they are subject to an active recreational fishery (average recreational landings between 1982 and 2001 was 5,400 metric tons [NFSC 2002b]).[3]

[3] The mean weight of all summer flounder landed in the commercial fishery between Main and Virginia increased consistently from 0.55 kg in 1982 to 1.01 kg in 2001 (NSFSC 2002a). We use 0.75 kg per fish to convert weight of landings to numbers of fish throughout this paper.

Maryland's four coastal bays, shown in Figure 1, are centrally located in the range of east coast summer flounder and support a significant portion of Maryland's summer flounder fishery. According to NMFS, in between 1994 and 2002 anglers took roughly 77,000 flounder fishing trips on average in the coastal bays per year, and approximately three-quarters of the recreational harvest in Maryland during that time was from the coast (Doctor 2002). The coastal bays also serve as an important nursery for young juvenile summer flounder, who depend on the bays' shelter and food sources for survival in the most vulnerable portion of their lives. Public concerns over water quality conditions in Maryland's coastal bays have been growing over the last several decades. Although most have stabilized in recent years, populations of many native fish (including summer flounder), shellfish, and underwater sea grasses have decreased significantly from their historic levels. These decreases have prompted a number of measures to protect or restore habitat and water quality conditions in the coastal bays ecosystem. In 1995 the Maryland coastal bays became one of only 29 estuaries nationwide accepted into the National Estuary Program. In 1996 the Maryland Coastal Bays Program (MCBP) was founded as a cooperative effort between local, state, and national stakeholder and legislative organizations. The MCBP's initial task was the development of the coastal bays Comprehensive Conservation and Management Plan. In other efforts, the Maryland Department of Natural Resources, the National Park Service, and the Maryland Coastal Bays Program have sponsored a number of water quality monitoring programs since 1987. Because the eastern shore of Maryland is home to a large amount of agriculture, the Maryland Department of Agriculture has also established TMDL's to limit nutrient input to the bays. Land trusts and the 2002 Atlantic Coastal Bays Critical Areas Legislation have also been used control development along the bays and decrease agricultural, commercial, and residential runoff into the bays (Wazniak et al. 2004).

4.1 The population model

We begin by specifying a model of the population dynamics of summer flounder that incorporates functional relationships between water quality conditions, fish abundance, and recreational and commercial harvest levels. The model is designed to allow simulation of changes in water quality conditions in the study area while holding conditions elsewhere constant, or simulation of uniform changes in water quality both in and out of the study area. This will be important for evaluating policies that might affect water quality in only part of the species' range, and it provides the flexibility for investigating the effects of the geographic scope of improved water quality conditions in addition to the magnitude of the changes. The model is also designed to be calibrated using readily available information on recreational harvest levels in and out of the study area and commercial harvest levels for the fishery as a whole.

First, reproduction is modeled as a function of the size of the entire breeding stock.[4] We use J_t to represent the number of young juvenile summer flounder (less than one year old) that were laid as eggs in the latter months of the previous year (November is the peak of the breeding season) and survive through the egg and larval stages on their journey from the open ocean to the coastal bays and estuaries where they are found in high concentrations by the early

[4] There is some evidence based on fish tagging studies that there may be two or three breeding stocks in the Atlantic instead of a single mixed stock (Able and Kaiser 1994, Kraus and Musick 2001). We ignore this complication here, though in principle it could be addressed if the recreational and commercial harvest data used to calibrate the model could be disaggregated appropriately.

summer months. Reproduction and survival through the egg and larval stages is assumed to be density dependent and described by a Beverton-Holt function:

$$J_t = S_{t-1}\left[\frac{a}{1+S_{t-1}/b}\right] \quad [5]$$

where S_{t-1} is the number of spawning adults in the previous year, a is the number of juveniles surviving to the early summer per adult at very low stock size, and b is the spawning stock size at which the survival rate to the juvenile stage is one half its maximum value.

Next, because our study area encompasses only a small part of the range of the Atlantic stock of summer flounder, [5] we must account for the fact that some portion of the population will move in and out of the study area from year to year. Thus, we model the stock as consisting of two connected sub-populations, which are defined by the boundary of the study area and the entire range of the stock. A fixed proportion, g, of the juveniles are assumed to be carried into the coastal bays in the study area with the prevailing tides and currents every year (Rogers and Van Den Avyle 1983). (All state variables and parameters specific to conditions inside and outside of the study area will be distinguished by superscripts *in* and *out*.) Thus, the numbers of juveniles in and out of the study area at the beginning of the summer in year *t* are:

$$J_t^{in} = J_t g \quad [6a]$$

$$J_t^{out} = J_t(1-g) \quad [6b]$$

The number of mature juveniles (between one and two years old) in the study area is the number of young juveniles in the study area the year before that survive and return, plus the

[5] Recreational landings of summer flounder in Maryland comprised about 4% of the total east coast landings between 1981 and 2002 (ASMFC 2003).

young juveniles outside of the study area the year before that survive and migrate in. The number out of the study area is the number of surviving young juveniles in the study area that do not return, plus the surviving young juveniles out of the study area that do not migrate into it:

$$M_t^{in} = J_{t-1}^{in} s_j SF + J_{t-1}^{out} s_j (1-SF)g \qquad [7a]$$

$$M_t^{out} = J_t^{in} s_j (1-SF) + J_t^{out} s_j (1-(1-SF)g) \qquad [7b]$$

where *SF* is the "site fidelity" for the species, the probability that a fish will return to the same bay or estuary it inhabited the year before,[6] and s_j is the survivorship of young juveniles. Note that the proportion $(1-SF)g$ of the young juveniles outside of the study area in one year is assumed to migrate into the study area the next. This is based on the assumption that a fraction *SF* of the now mature juveniles return to the bay or estuary they inhabited the year before, and the remainder re-enter the large mixed pool of juveniles while in the open ocean during the winter months and are subject to the same influences from the tides and currents that determine the fraction of young juveniles that are "deposited" in and out of the study area every year.

The number of adults in the study area at the beginning of the summer (i.e., available to be caught by anglers) is the number of adults in the study area in the previous year that were not harvested in the recreational fishery, survived over the winter, and returned to the study area, plus the number of adults out of the study area in the previous year that were not

[6] Summer flounder appear to exhibit an intermediate degree of site fidelity (Rogers and Van Den Avyle 1983, Able and Kaiser 1994, Grimes et al. 1989, Kraus and Musick 2001), but a precise estimate of *SF* does not seem possible. Also note that *SF* will depend on the scale at which site fidelity is measured. It will increase from a minimum value when measured at the smallest appropriate scale – which in our case might be the average dispersal distance for an individual over the course of the summer – up to a value of one when measured at the scale of the entire range of the species.

harvested, survived over the winter, and migrated into the study area. The number of adults out of the study area is determined by similar reasoning. Thus:

$$A_t^{in} = \left[\left(A_{t-1}^{in} - HR_{t-1}^{in}\right)s_a + M_{t-1}^{in}s_m\right]SF + \left[\left(A_{t-1}^{out} - HR_{t-1}^{out}\right)s_a + M_{t-1}^{out}s_m\right](1-SF)g \quad [8a]$$

$$A_t^{out} = \left[\left(A_{t-1}^{in} - HR_{t-1}^{in}\right)s_a + M_{t-1}^{in}s_m\right](1-SF) + \left[\left(A_{t-1}^{out} - HR_{t-1}^{out}\right)s_a + M_{t-1}^{out}s_m\right](1-(1-SF)g) \quad [8b]$$

In Equations [8], s_a is the survivorship of adults over the winter (we assume that harvesting in the recreational fishery is the main source of mortality in the summer months), and s_m is the survivorship mature juveniles over a full year.

The recreational harvest in the study area in year *t* is the product of the number of trips, T_t^{in}, and the average catch per trip, C_t^{in}, which we model as $C_t^{in} = h^{in} A^{in q}$ (and similarly for out of the study area):

$$HR_t^{in} = T_t^{in} \cdot h^{in} A^{in q} \quad [9a]$$

$$HR_t^{out} = T_t^{out} \cdot h^{out} A^{out q} \quad [9b]$$

As described in Section 3 below, we use data on a sample of summer flounder recreational fishing trips in the study area to estimate q and the potential responses of h^{in} and h^{out} to changes in DO.

To accommodate the use of commercial harvest data for calibrating the model, we follow convention in the fisheries science literature and specify the adult survival rate over the winter as a function of the instantaneous mortality rates from natural causes, *M*, and commercial fishing, F, so $s_a = e^{-M-F}$ (Haddon 2001). It then follows that the total commercial harvest level will be:[7]

$$HC_t = \left(A_t^{in} - HR_t^{in} + A_t^{out} - HR_t^{out}\right)[1 - s_a]\frac{F}{M+F} \quad [10]$$

Finally, we assume that only those adults not harvested in either the recreational or the commercial fisheries reproduce. Thus, the spawning stock in year t, i.e., the stock that will contribute to the cohort of juveniles in year $t + 1$, is:

$$S_t = A_t^{in} - HR_t^{in} + A_t^{out} - HR_t^{out} - HC_t \quad [11]$$

Young juvenile summer flounder are known to be especially sensitive to dissolved oxygen (DO) conditions, particularly in the summer months when DO can reach very low levels during the morning hours and when high nutrient loads flushed from agricultural fields after storm events can spur algal growth and subsequent DO depletions in the water column (Able and Kaiser 1994, US EPA 2002, Brady pers comm). To quantify the effects of DO conditions on summer flounder, we use information from US EPA (2000), which summarizes results from laboratory tests for a variety of aquatic organisms in the mid-Atlantic region. The 24-hour LC_5 and LC_{50} and values -- the DO concentrations at which on average 5% and 50% of young juvenile summer flounder died within 24 hours of exposure -- were 1.90 and 1.59 mg/L. We assume that the maximum survivorship over the 122 days from May through August is 0.75,

[7] The reader can confirm by taking the integral $HC_t = \int_0^1 N(t)F\,dt$ where $N(t) = \left(A_t^{in} - HR_t^{in} + A_t^{out} - HR_t^{out}\right)e^{-Mt-Ft}$.

and we use the two LC values to solve for the parameters in a double exponential function: $s_{24} = a\exp(-\exp(b_1 + b_2 DO))$.[8]

One of the key difficulties in transferring laboratory results to natural settings is that water quality conditions will vary significantly both temporally and spatially in real environments, whereas most laboratory studies hold conditions constant in a homogeneous artificial environment for the duration of each test. We make two adjustments to the double exponential function to incorporate it into the population model. First, to accommodate natural temporal variations in DO levels we assume that the probability of mortality is independent across each increment of time. Thus, the survivorship over each quarter-hour time interval (corresponding to the temporal scale at which the available water quality data were collected) is $s_{1/4} = (s_{24})^{1/96}$. Second, to accommodate the potential behavioral adjustments summer flounder can make in response to local spatial variations in DO levels, another parameter, x, is added to the model, so the survivorship over each quarter-hour time interval becomes: $s_{1/4} = x(s_{24})^{1/96} + (1-x)$. The parameter x can be given a loose physical interpretation. It can be thought of as the proportion of the young juvenile stock exposed to the measured DO conditions in each successive quarter-hour time interval, where the remainder of the stock, the proportion $1-x$, is exposed to DO levels above the sensitive range for the species, and where the particular individuals exposed from one time interval to the next are independent. Thus, x incorporates the combined effects of spatial variation in DO levels and the ability of individuals to move to better locations in response to unfavorable local conditions. Combining the

[8] a is the maximum survivorship for one day during the summer, and there are 122 days in the months May through August, so we set $a = 0.75^{1/122}$.

laboratory results with these additional assumptions gives the model for the summer survivorship of young juvenile summer flounder:

$$s_j = s_{jw} \prod_{d=121}^{242} \left[\prod_{q=1}^{96} \left\{ x s_{1/4}\left(DO_{dq}\right) + (1-x) \right\} \right] \qquad [12]$$

where s_{jw} is the survivorship of young juveniles over the non-summer months (September through April), d indexes days of the year between May and August, q indexes the quarter-hour time intervals in a day, and DO_{dq} is the DO level during time interval q on day d.

In the application below, repeated random draws from the empirical distribution of DO based on water quality data from two continuous monitoring stations in the Isle of Wight Bay in 2002 and 2003, shown in Figure 2, are used in Equation [12] to simulate the survivorship of young juveniles. Note that the survivorship function will be responsive not only to changes in the annual average DO levels, but also to changes in the variation and the frequency of extremely low DO levels on time scales as short as fifteen minutes.[9] For example, Figure 3 shows the effect on summer survivorship of young juveniles from proportional changes in DO for all time intervals (panel a) and of increasing the minimum DO level that is reached in any time interval (panel b). These graphs are based on repeated random draws from the empirical DO distribution shown in Figure 2 and Equation [12]. The top graph shows that uniform proportional changes in DO near baseline conditions will lead to less than proportional changes in survivorship, but large decreases in DO would eventually lead to much greater than proportional decreases in survivorship (the dotted line is the 1:1 line). The bottom graph shows

[9] It will not be sensitive to the *duration* of low DO events, however, because of our simplifying assumption that the probability of mortality is independent from one time interval to the next.

that eliminating occurrences of DO below a minimum threshold value increases young juvenile survivorship rapidly through a fairly narrow range of DO. The ability to model these or similar types of water quality changes, based on any time series of DO levels on time scales as short as 15-minute intervals, provides the flexibility to evaluate a much wider variety of water quality policies than would be possible with models that include only annual average water quality measures.

Parameters for the population model used in the application below are taken directly from government reports, internet databases, and the peer-reviewed literature, or they are estimated by calibrating the model so the equilibrium recreational and commercial harvest levels under baseline conditions are equal to the average historic harvest levels. Table 1 summarizes the parameters and the information sources or calibration procedures used for each.

4.2 The catch model

We use three datasets from Maryland's Department of Natural Resources (MD DNR) to estimate the relationships between the average catch per trip, water quality conditions, and fish abundance. Water quality data were obtained from the MD DNR Eyes on the Bay program. Water quality measurements taken at twenty-three of the program's monitoring stations located throughout the four coastal bays were used to calculate average levels for DO, water temperature, salinity, and secci depth in each of the four bays in each month April through October 2002. Catch data were obtained from the MD DNR Summer Flounder Volunteer Angler Survey. Anglers who fished for summer flounder in Maryland were asked to report the details of all of their fishing trips in 2002. The information collected includes the date of each trip, total catch, location fished, fishing method used, party size, hours fished, and several demographic characteristics. Our analysis uses data on the 612 trips taken to the four Maryland coastal bays. Fishery-independent data on fish abundance were obtained from the Maryland Coastal Bays Finfish Project, which performs bottom trawl surveys at twenty sites in the coastal bays every month April through October. The average number of summer flounder captured per haul is taken as an index of fish abundance for each bay in each month in 2002. Trip and angler characteristics from the Volunteer Angler Survey were matched with water quality measures from the Eyes on the Bay program and the fish abundance index from the Coastal Bays Finfish Project to create the dataset used for estimation.

Because the data on catch per trip come in the form of non-negative integers, we use a negative binomial count regression model to estimate the catch function (Cameron and Trivedi 1998), and we use the standard exponential mean function to specify the average catch per trip:

$$C_i = \exp\left(\mathbf{Z}_i \mathbf{d} + q \ln \tilde{A}_i\right) \qquad [13]$$

where C_i is the reported summer flounder catch, $\mathbf{Z}_i$ is a vector of angler characteristics and water quality conditions, $\tilde{A}_i$ is the index of fish abundance, and $\mathbf{d}$ and q are parameters to be estimated. The regression results are shown in Table 2. All angler characteristic and fishing method variables are significant and possess their anticipated signs. The coefficient estimates on all water quality variables also are of the expected sign, but only the coefficients on water temperature and secci depth are statistically significant. The abundance index is positive and statistically significant. When the model is estimated with the abundance index excluded (results not shown), all water quality variables become highly statistically significant. Thus, with catch per trip is conditioned on fish abundance there appears to be only a modest direct "catchability effect" of water quality on catch.[10] Only the dummy variable for Chincoteague Bay was statistically different from the control bay, Assawoman Bay, which suggests that the measures of water quality and fish abundance explain most of the variation in catch between the bays.

Finally, note that because the natural log of the abundance index is used in the exponential mean function, our estimating equation corresponds to the functional form posited for the average recreational catch in Equations [9] of the population model. If the abundance index is proportional to the actual adult abundance, A_i, then $\tilde{A}_i = \boldsymbol{n}_i A_i$ and we can define $h_i = \exp(\mathbf{Z}_i \mathbf{d}) n_i^{-q}$ and rewrite our catch equation as $C_i = h_i A_i^q$.

4.3 The recreation demand model

To estimate the relationship between average catch per trip and recreation demand we use stated choice data from the 2000 Survey of Northeast Recreational Anglers (Hicks 2002). The study was conducted by the National Marine Fisheries Service to evaluate angler responses to different fishery management alternatives. Each respondent was presented with four different contingent choice questions in a conjoint format concerning summer flounder fishing in the Mid-Atlantic. In each question, respondents were asked to choose between two alternative summer flounder fishing trips and a "do something else" option. The attributes that could differ between the two trips included travel costs, likely total catch, the minimum size

[10] Several other functional forms were for the catch equation were estimated but those results are not reported here. The coefficient estimates on the DO variable(s) were weakly significant in a few of the alternative specifications, but none of the models clearly performed better that the specification reported here. Furthermore, this functional form facilitated a more natural linkage between the catch model and the population models, as discussed in the main text.

limit, the catch limit, the likely number of legal-sized summer flounder caught, and the likely total catch of other fish species. The recreation demand model estimated below uses responses from the 2,392 participants who answered all four stated choice questions.

We model the survey respondents' stated choices using a repeated choice mixed logit model. The utility individual *i* would receive from selecting alternative *k* on choice occasion *t* is specified

$$U_{ikt} = Z_{ik} f_i + e_{ikt}. \quad [14]$$

Individuals are assumed to choose the alternative that provides the maximum utility on each choice occasion. Each individual is assumed to have a unique vector of coefficients f_i, however coefficients are assumed to vary normally across individuals with a known distribution $f(f \mid y)$, where y contains the means and variances of the f_i coefficients. We further assume coefficients are uncorrelated and constant across choice occasions.[11]

The error term e_{ikt} is assumed to be distributed identically and independently according to an extreme value distribution, which gives rise to the logit probability function. The probability that individual *i* makes a sequence of choices y_i over a series of choice occasions *t* = 1, 2, … *T*, given their preferences as described by f_i, is then the product of logit formulas:

$$L(y_i \mid f_i) = \prod_{t}^{T} \frac{\exp(Z_{ik} f_i)}{\sum_j \exp(Z_{ij} f_i)}. \quad [15]$$

However, because the true coefficient values are unknown and assumed to vary across individuals according to $f(f \mid y)$, the logit probability function must be integrated over all

[11] This is a reasonable assumption if tastes are thought to be stable over time.

possible values of f_i. Due to the analytical difficulty of evaluating multiple integrals, results are most easily obtained through simulation. The simulated probability is:

$$SP(y_i \mid y) = \frac{1}{D}\sum_{d=1}^{D}\left[\prod_{t}^{T} \frac{\exp(Z_{ik} f_i^d)}{\sum_j \exp(Z_{ij} f_i^d)}\right] \qquad [16]$$

where D is the number of draws and f_i^d is draw d from the given mixing distribution for f_i.[12] These simulated probabilities are used to construct a simulated likelihood function that is then maximized to produce estimates of y, the parameters of the f distribution (Train 2003).

Results from the repeated choice mixed logit model are shown in Table 3. The travel cost coefficient is negative and statistically significant, as expected, indicating that all else equal anglers prefer to fish at sites closer to their home.[13] The results also indicate that anglers are more likely to visit sites with higher total catch, bag limits, legal take-home catch of summer flounder, and total catch of other fish. Owning a boat, being non-white, male, and working full time are found to increase an angler's probability of taking a trip on any given choice occasion. Only the college education and income variables have statistically insignificant mean coefficient estimates, although all three variables have their anticipated signs. All coefficients except male, work fulltime, and income squared are found to vary significant across the population.

The results of the trip demand model allow us to calculate the value of changes in water quality, catch rate, and site availability to anglers. Following Small and Rosen (1981) and Hanemann (1999), changes in expected utility are monetized using the coefficient estimate on

[12] We use 200 Hess, Train, and Pollock draws in estimation. See Train (2003) for a discussion of the advantages of the Hess, Train, and Pollock draws.

[13] The travel cost coefficient was assumed to be fixed to aid estimation.

the travel cost variable, which is the estimated marginal utility of income. Again, because the mixed logit model only estimates the means and variances of the distributions of the coefficients, calculating the welfare effects of changes to the sites in the choice set requires simulated integration. Using the estimated coefficient distributions, random coefficient values are drawn and expected utility and welfare is computed with each draw. The computed welfare values are then averaged to produce the final welfare estimate. For example, the welfare change associated with a change in quality at some or all of the sites would be computed as:

$$W = \frac{1}{D}\sum_{d=1}^{D}\left[\frac{\ln \sum_{k=1}^{K}\exp(Z_{ik} f_i^d) - \ln \sum_{k=1}^{K}\exp(Z_{ik}^{*} f_i^d)}{f_{TC}}\right] \qquad [17]$$

where D is the total number of draws from the estimated distribution, f_i^d is draw d of f_i from the distribution, f_{TC} is the travel cost coefficient, and Z_{ik}^{*} is a vector of changed quality measures of some or all of the K sites. The value of the loss of one or more sites from participants' choice sets would be calculated in a similar fashion by excluding those sites from the expected utility summation in the second term of Equation [17].

The results indicate that at the average catch rate from the MD DNR Summer Flounder Volunteer Angler Survey, which is approximately two fish per angler per trip, a one percent increase in the catch rate increases the value of each choice occasion by $0.078. Values for extra percentage point increments are roughly constant over a large range, such that the value per choice occasion for a 50 percent increase in the catch rate is $3.93. This estimate is slightly larger than the value of an increase of one fish per trip for bottom fish in Maryland estimated reported by Hicks et al. (1999), which is $2.44, but it is near the low end of the range of values for an increase in catch of one fish per trip found in the marine recreation literature summarized by Freeman (1993), which is \$2.21 to \$85 in 1991 dollars (most values are below \$20). The estimated welfare loss per choice occasion from eliminating all fishing options in our sample of anglers is \$289. This is near the high end of the range of per trip values for single species fisheries found by Freeman (1993), which is \$4.44 to \$346. One reason our estimate is relatively large may be because we simulate the elimination of *all* fishing options using data from a stated choice survey in which respondents choose the trip option over the "do something else" option in 93% of the choice occasions. A large number of the studies surveyed by Freeman valued the loss of single site from a multiple site choice set, and when participation was modeled the studies commonly had higher frequencies of non-participation.

To estimate the total welfare change per year in the recreational fishery, we must multiply the per person per choice occasion welfare estimate by the number of potential anglers, N, and the number choice occasions in a season, O. We can estimate N and O using the identity $T \equiv (N \times O \times p)/n$, where T is the total number of summer flounder fishing trips in a year, n is the average number of anglers per trip, and p is the average probability of taking a trip on any choice occasion. The average number of summer flounder fishing trips to Worcester County, MD between 1994 and 2003 was 76,602 trips per year. The average number of anglers per trip from the MD DNR Volunteer Summer Flounder Angler Survey is 2.23. We calculate the average probability of taking a trip on any given choice occasion using responses to a supplemental question in the stated choice survey. After completing the battery of stated choice questions, respondents were asked how many times they took a fishing trip in the previous two months. The average response was 10.4 of the 61 days in the previous two months, or 17%, which we take as our estimate of p, treating each day as an independent choice occasion.[14] Most summer flounder fishing trips occur between April and August, so we take 153 as our estimate of O. Combining these estimates, we find that the number of potential anglers who fish for summer flounder in the study area is 99,773. Using these values of N and O, the welfare change per year of increasing the average catch per trip in the study area by 50 percent is $601 per potential angler and $60 million in total.

We also use results from the mixed logit model to predict the effect of changes in catch per trip on the probability of taking a trip on any given choice occasion, which is p from above and the complement of the probability of "doing something else" in the stated choice survey. Using the mixed logit parameter estimates and Equation [16], we calculate that a one percent change in average catch per trip increases the probability of taking a trip on a given choice occasion by 0.0007 percent, and a 50 percent change in the average catch per trip increases the probability of taking a trip on a given choice occasion by 0.032 percent. The very small change in participation rates is again most likely due to the 93% initial participation rate per choice occasion in the stated choice survey. Because respondents in some sense already seem decided to take trips, the estimated model predicts that changes in site attributes will have little effect on the decision of whether or not to take a trip.[15]

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[14] Several respondents reported fishing every day in the previous two months.

[15] Other researchers have found larger effects. For example, Englin et al. (1997) find a small to moderate effect of catch rate on trip demand. They estimate two versions of a joint catch and trip demand function based on a Poisson count regression model. The coefficient estimates on the predicted catch variable in the two versions of their model are 0.081 and 0.138. The mean value of predicted catch per trip is not reported, but using a plausible range of 1 to 5 fish per trip gives an elasticity of trip demand with respect to catch per trip between 0.08 and 0.7.

4.4 Calibration

Before the models can be put to use in evaluating simulated water quality changes, they must first be calibrated to baseline conditions. The unknown parameters and state variables to be estimated are: x, h^{in}, h^{out}, A_0^{in}, A_0^{out}, $\boldsymbol{b}$, and g. First, x is set such that the young juvenile survivorship over the summer, calculated using Equation [12], equals the young juvenile survivorship reported in the literature, which is presumed to correspond to the average survivorship across the range of the species under current conditions. s_j is calculated by repeated random sampling from the empirical DO distribution shown in Figure 2 and using Equation [12]. Next, $\boldsymbol{a}$ is set such that the dominant eigenvalue of the annual transition matrix with no density dependence or fishing mortality equals the geometric growth rate for the species at low population sizes (Gotelli 2001), which was estimated independently using stock-recruitment data from NFSC (2002a) and using the approach of Myers et al. (1999). Next, $\boldsymbol{b}$ is set such that the equilibrium commercial harvest level predicted by the model under baseline conditions equals the historic average harvest levels (NFSC 2002b). Finally, we assume $HR^{in}/A^{in} = HR^{out}/A^{out}$ and set g such that the recreational harvest levels predicted by the model under baseline conditions in and out of the study area match historic average levels (ASMFC 2003).[16] The calibration of $\boldsymbol{b}$ and g simultaneously determines the baseline values for

[16] The main motivation for this assumption is a practical one: another equation is needed to identify all of the remaining unknown parameters when calibrating the model to baseline conditions. However, we can provide a superficial theoretical justification. The overall harvest pressure from recreational fishing will be equal throughout the range of the species if the fish and the anglers jointly distribute themselves across sites such that all arbitrage opportunities (for avoiding sites with high harvest pressure for the fish, and for fishing at sites with high catch rates for the anglers) are exhausted. Clearly this is a strong assumption and in the face of significant variations in environmental conditions, habitat quality, and angler characteristics across the species' range it will hold, at best, only approximately. This assumption would not be needed if comparable fishery-independent data on fish abundance were available for all bays and estuaries in the range of the species.

the abundances of all life stages in and out of the study area. Refer back to Table 1 for a summary of the calibration steps.

4.5 Evaluating water quality changes

With all parameters estimated and the starting values for all state variables specified, the effects of water quality changes can now be simulated and the resulting impacts on the fish stock and the anglers evaluated. We investigate two types of water quality changes, proportional increases in DO and reductions in the frequencies of low DO events. These changes are simulated by modifying the empirical DO distribution: proportional increases expand the distribution shown in Figure 2 to the right, and reductions in the frequencies of low DO events shift the portion of the distribution below the simulated threshold downward.

The change in DO conditions first causes a change in the survivorship of young juvenile flounder over the summer months, and after two years the first new larger cohort of young juveniles is recruited to the adult population. This increases the availability of catchable fish and therefore the average catch per trip, which in turn leads to an increase in demand for trips. The increase in catch per trip and demand for trips combine to yield a new, higher total harvest in the recreational and commercial fisheries, which partly offsets the increase in fish abundance from the improved DO conditions. After some transition phase, the system will reach a new steady state where the fish abundance and the harvest levels and will remain stable from year to year.[17]

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[17] We could introduce variability in young juvenile survivorship by taking a new set of draws from the empirical DO distribution and using them in Equation [12] every year. This would produce qualitatively similar behavior to the results presented here, but the new "steady state" would exhibit year to year fluctuations around a long term average.

The results are summarized in Table 4 and Figure 4. First, Table 4 shows results from four scenarios based on the two water quality changes applied in the study area only (columns 1 and 3) and throughout the entire range of the species (columns 2 and 4). The model predicts that if water quality is improved only in the study area then the impacts on the fish population and the benefits to the anglers will be small. Recreational harvests in the study area comprise approximately 2% of the total recreational harvest for the species, and the simulated water quality improvements in the study area alone increase catch rates by approximately 0.5% in the study area 0.1% out of the study area. However, if water quality improvements of the same magnitude are extended over the entire range of the species, catch rates and harvest levels increase on the order of 5 - 10%. The differences between the scenarios where water quality is changed only in the coastal bays versus throughout the entire fishery come about because there is assumed to be some mixing of the adult stock in the open ocean during the winter. Only a portion of the juveniles subject to the improved water quality conditions in the study area will return to the study area the next year as adults (recall that site fidelity, *SF*, is set to 0.5), and the next year only a fraction of those return, and so on. Thus, the density of adult fish available to be caught by the anglers in creases by only a fraction of what would occur if either site fidelity were absolute or if the water quality improvements affected the entire population, as shown by the results in columns 2 and 4. One of the advantages of the model is that questions about the effect of the scope of water quality improvements can be investigated directly.

The results also indicate that recreational harvest levels will increase with fish abundance at a less than proportional rate. Because $q = 0.328$, a 1% increase in fish abundance will lead to a 0.328% increase in the average catch rate. Following convention from models commonly used for commercial fisheries (Haddon 2001), we have assumed that the commercial harvest is directly proportional to the stock size (i.e., $q = 1$), which means that responses to increases in fish abundance in the commercial fishery would be greater than those in the recreational fishery. However, this must be considered a working hypothesis only. Firm conclusions would require data for the commercial fishery analogous to the data used here for the recreational fishery: measures of harvest effort and contemporaneous and fishery-independent measures of fish abundance.

The small increases in trip demand predicted by the model are a direct consequence of the predicted change in probability of choosing the "do something else" option from a change in the average catch rate across all available fishing sites, as calculated from the mixed logit recreation demand model. The survey was designed to illicit anglers' preferences for various combinations of catch and size limits for summer flounder, not to predict changes in participation rates. The survey is appropriate, however, for estimating per angler welfare effects of changes in the catch rate, which is our main use for it. To more accurately estimate the effect of changes in the catch rate on trip demand revealed preference data on actual trips taken or an entirely different survey would be preferred.

The estimated changes in recreational harvest levels are a consequence of the combined effects of the response of the fish population to changes in DO conditions, the response of the average catch rate as a function of fish population density, and the response of changes in trip demand as a function of changes in the average catch rate.[18] These changes are appreciable for what seem to be reasonable assumptions about potential improvements in water quality conditions, but such gains may require improvements in water quality across large portions of the species' range.

The estimated aggregate welfare effects are reported as the constant annual equivalent of the stream of benefits from year one onward discounted at 3%. (It generally takes about 25 years for the system to reach a new steady state; see Figure 4.) If V_t is the current value of the benefits in year *t*, and the new steady state is reached by year *T*, then the annualized aggregate benefits are:

$$Benefits = \left[r \sum_{t=1}^{T} \frac{V_t}{(1+r)^t} \right] + \frac{V_T}{(1+r)^T} \qquad [17]$$

Note that even in the cases where water quality improvements are confined to the study area, the aggregate benefits are much larger out of the study are than in it. The per angler welfare effects are an order of magnitude smaller out of the study area, but the number of anglers to which these external benefits accrue is two orders of magnitude larger than in the study area. The benefits estimates reported in Table 4 depend directly on the estimated values for a change in the catch rate per angler per choice occasion and the estimated number of "angler-choice occasions" ($N \times O$). The estimated value of changes in the catch rate per choice occasion appear reasonable. We have less confidence in the estimated number of angler-choice

[18] The estimated changes in commercial harvest levels are a consequence of the simple assumption that commercial harvest is directly proportional to the stock size and does not change. The commercial fishery component of the model could be improved by incorporating a behavioral response of the commercial harvesters based on information about the prevailing mana gement regime.

occasions. This is one of the key limitations of recreation demand models that focus on site choices alone. It is not immediately clear how best to estimate the appropriate number of choice occasions in a season for the representative angler or household for the purpose of calculating aggregate welfare values.

Finally, Figure 5 plots the aggregate benefits in and out of the study area for two qualitatively different types of DO changes (analogous to Figure 3). Panel (a) shows the effect of proportional increases in DO during all time intervals throughout the entire fishery, and panel (b) shows the effect of eliminating occurrences when DO levels drop below a minimum threshold. Figure 5 suggests that the maximum possible benefits from water quality improvements to the recreational summer flounder fishery are on the order of $60 million per year. The maximum benefits will be delivered if all occurrences of DO below about 1.75 mg/L could be eliminated. Analogous to Figure 3, benefits accrue much more rapidly when low DO events are eliminated than when DO is increased proportionally. This suggests that accurately characterizing the precise nature of the changes in water quality conditions that will be brought about by a policy will be one of the keys to producing accurate benefits estimates.

5 Summary and conclusions

In this paper we have developed and applied a bioeconomic model of a coastal recreational fishery for estimating the value of water quality changes. A variety of information sources were used to specify a biological model of summer flounder population dynamics, and several unique data sets were used to estimate a count regression model of average catch per fishing trip and a mixed logit recreation demand model. The models were combined in a bioeconomic framework and calibrated to baseline conditions using historic recreational harvest levels from both in and out of Maryland's coastal bays and commercial harvest levels for the entire fishery. Several water quality improvement scenarios based on simulated changes in the empirical DO distribution were evaluated. The results indicate that substantial increases in summer flounder populations and associated benefits to recreational anglers may be possible if water quality conditions are improved throughout the range of the species. If water quality improvements are confined to a small area, however, only small increases in recreational summer flounder fishing benefits will be possible. It should be noted that this study does not estimate the effects of water quality changes on other fish species or any other environmental resources, nor does it value water quality changes for people who do not (potentially) participate in the recreational summer flounder fishery, so our benefits estimates capture only a portion of the total value of water quality improvements.

Finally, because of the limitations of the available data, a number of simplifying assumptions were required to integrate the components of the bioeconomic framework and apply it to the entire recreational fishery. Thus, the estimates of changes in fish abundance, harvest levels, and welfare effects presented in this paper must be considered provisional. The immediate value of this research has been to lay a foundation for further data collection efforts and modeling refinements. First, the primary limitation of the population model arises from the simplifying assumptions necessary to use tests of the effects of DO under laboratory conditions to forecast young juvenile survivorship in their natural environments. The two most uncertain variables in the population model are s_j^{max}, the survivorship of young juveniles under ideal DO conditions, and *SF*, the site fidelity of summer flounder. The primary limitation of the catch model is the lack of compatible data on recreational fishing activities, water quality, and fish abundance throughout the full range of the species. The catch model estimated here used data from only a small portion of the species' range and only for a single year. Maryland's Volunteer Summer Flounder Angler Survey, Eyes on the Bay program, and Coastal Bays Finfish Project are ongoing data collection efforts, so refinements of the catch model over time will be possible. Other coastal states have similar data collection programs, but it may not be possible to integrate all of the available data into a single dataset appropriate for modeling recreational catch throughout the fishery. More standardized data collection efforts on water quality conditions, fish abundance, and angler activities would be helpful here. Finally, the primary limitation of the recreation demand model in this context is that it is not ideally suited for predicting changes in trip demand. A secondary limitation is the difficulty of identifying the appropriate population for extrapolating the per angler per choice occasion values to estimate total welfare changes. Other types of recreation demand models are better suited for this purpose, so the primary limitation here is a lack of appropriate data.

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Table 1 – Information sources and calibration procedures for key parameters and state variables in the population model.

Parameter or state variable	Symbol	Value(s)	Source(s) or calibration procedure
Baseline summer survivorship of young juveniles (May – August)	s_{js}^{0}	0.452	Adapted from US EPA (2002). The value reported for annual proportional survivorship is $e^{-2.38} = 0.093$. We use $e^{-2.38 \times 4/12} = 0.452$ for summer survivorship.
Maximum summer survivorship of young juveniles	$s_{js}^{\max}$	0.75	Assumed.
Parameters of young juvenile survivorship function, Equation [12]	a	0.9976	Set so $s_{js} = s_{js}^{\max}$ when $DO_t >> 3 \quad \forall t$; $a = \left(s_j^{\max}\right)^{1/122}$
	b_1	14.017	Calculated by fitting laboratory test results reported in US EPA (2000) to a double exponential function: $s_{24} = a\exp\left(-\exp\left(b_1 + b_2 DO\right)\right)$
	b_2	-9.056	
	x	0.00532	Calibrated so $s_{js} = s_{js}^{0}$ at baseline DO conditions.
Winter survivorship of young juveniles (September – April)	s_{jw}	0.205	Adapted from US EPA (2002). The value reported for annual survivorship is $e^{-2.38} = 0.093$. We use $e^{-2.38 \times 8/12} = 0.205$ for winter survivorship.
Annual survivorship of mature juveniles	s_m	0.819	US EPA (2002)
		0.27	Grimes et al. (1989)
Instantaneous natural mortality rate for adults during winter	M	0.2	Terciero (2003)
Instantaneous commercial fishing mortality rate	F	0.26	Terciero (2003)
Winter survivorship of adults	s_a	0.632	$s_a = e^{-M-F}$. Also see Grimes et al. (1989, p 6).
Geometric growth rate at low population size	l	1.571	Estimated from a Ricker stock recruitment model using data from NFSC (2002a) and following the approach of Myers et al. (1999). (Note: Myers et al. report a value of 2.203 for the flounder fam ily.)

Parameter or state variable	Symbol	Value(s)	Source(s) or calibration procedure
Number of juveniles surviving to the early summer per adult at very low stock size	a	68.73	Estimated by calibrating the model so the dominant eigenvalue of the annual transition matrix with no density dependence or fishing mortality equaled 1 (Gotelli 2001).
Average catch per trip under baseline conditions	C_0^{in}	13.02	Average catch per trip reported in MD DNR Summer Flounder Volunteer Angler Survey.
Historic average number of summer flounder recreational fishing trips in the study area	T_0^{in}	76,602	Average number of summer flounder fishing trips in Worcester County, MD between 1994 and 2003.
Historic average harvest level out of the study area	HR_0^{out}	5,901,406	Average of total recreational landings between 1994 and 2001 (ASMFC 2003) minus $C_0^{in} \times T_0^{in}$
Historic average commercial harvest	HC_0	7,720,303	Average total commercial harvest between 1994 and 2001, assuming 0.75 kg / fish (NFSC 2002b).
Site fidelity	SF	0.5	Subjective estimate based on Rogers and Van Den Avyle (1983), Able and Kaiser (1994), Grimes et al. (1989), and Kraus and Musick (2001).
Stock size at which egg and larvae survival probability is half the maximum	b	14,820,027	Estimated by calibrating the model so the equilibrium commercial harvest level predicted by the model under baseline conditions equaled historic average levels.
Proportion of new eggs and larvae deposited in the study area every year	g	0.0213	Estimated by calibrating the model so the recreational harvest levels inside and outside of the study area match historic average levels.

Table 2 – Regression results for the catch model.

Variable	Coefficient	t-stat	Mean
Constant	-6.887	-3.159	
ln(hours)	1.509	5.821	0.561
ln(anglers)	0.806	5.271	0.675
Shore	-0.430	-1.395	0.00362
Charter boat	0.501	2.944	0.332
Isle of Wight Bay	-0.121	-0.908	0.275
Sinepuxent Bay	-0.372	-1.088	0.403
Chincoteague Bay	-1.250	-2.53	0.00132
Isle of Wight or Sinepuxent	0.206	0.873	0.220
DO	0.0287	0.431	6.25
Water temp	0.109	4.05	24.2
Salinity	0.132	1.719	30.9
Secci depth	1.067	2.883	0.543
ln(average trawl catch)	0.328	4.404	0.286

Table 3 –Results of the mixed logit recreation demand model.

	Mean Coefficient		Standard Deviation	
Variable	**Estimate**	**t-stat**	**Estimate**	**t-stat**
Travel Cost	-0.0173	-13.052	-------	---------
Total Catch	0.0697	12.357	0.0587	3.117
Bag Limit	0.1027	16.573	0.0730	3.893
Min Size*Legal Catch	0.0128	17.729	0.0101	8.414
Other fishing Good	0.4069	7.165	0.8860	5.705
Other fishing Bad	-0.5628	-10.155	0.7551	6.628
No Trip Dummy (NTD)	-2.5097	-2.568	2.5185	5.316
NTD*Boat Owner	-0.4845	-1.821	1.2588	2.310
NTD*Non-White	-1.4941	-2.941	4.6126	8.042
NTD*Male	-0.9759	-2.060	0.8207	1.029
NTD*Attended College	-0.0935	-0.306	1.2842	2.465
NTD*Work Fulltime	-0.7053	-2.327	1.0310	1.509
NTD*Income	0.1369	0.445	0.1823	2.449
NTD*Income^2	-0.0105	-0.434	0.0092	0.926
Number of Respondents	2392			
Number of Observations	9568			
Mean Log-likelihood	-0.781457			

Table 4 – Results for four water quality change scenarios.

		Baseline conditions	50% increase in DO levels in the study area	50% increase in DO levels everywhere	50% reduction in occurrences of DO < 2 mg/L in the study area	50% reduction in occurrences of DO < 2 mg/L everywhere
		levels	[% chng]	[% chng]	[% chng]	[% chng]
In the study area	*DO*	6.13	50	50	0.4	0.4
	s_j	0.457	7.47	7.47	15.98	16.21
	C	13.0	0.47	4.51	1	9.31
	T	76,600	0.02	0.19	0.04	0.38
	H	107,000	0.49	4.71	1.04	9.73
	Benefits		**$31,700**	**$278,534**	**$67,574**	**$578,673**
Out of the study area	*DO*	6.134	0	50	0	0.4
	s_j	0.457	0	7.47	0	16.21
	C	13.02	0.09	4.51	0.2	9.31
	T	3,591,489	0	0.19	0.01	0.38
	H	5,011,085	0.10	4.71	0.21	9.73
	Benefits		**$261,479**	**$13,059,556**	**$567,045**	**$27,133,613**
	S	29,324,427	0.33	15.75	0.73	34.14
	H	7,720,303	0.33	15.75	0.73	34.14

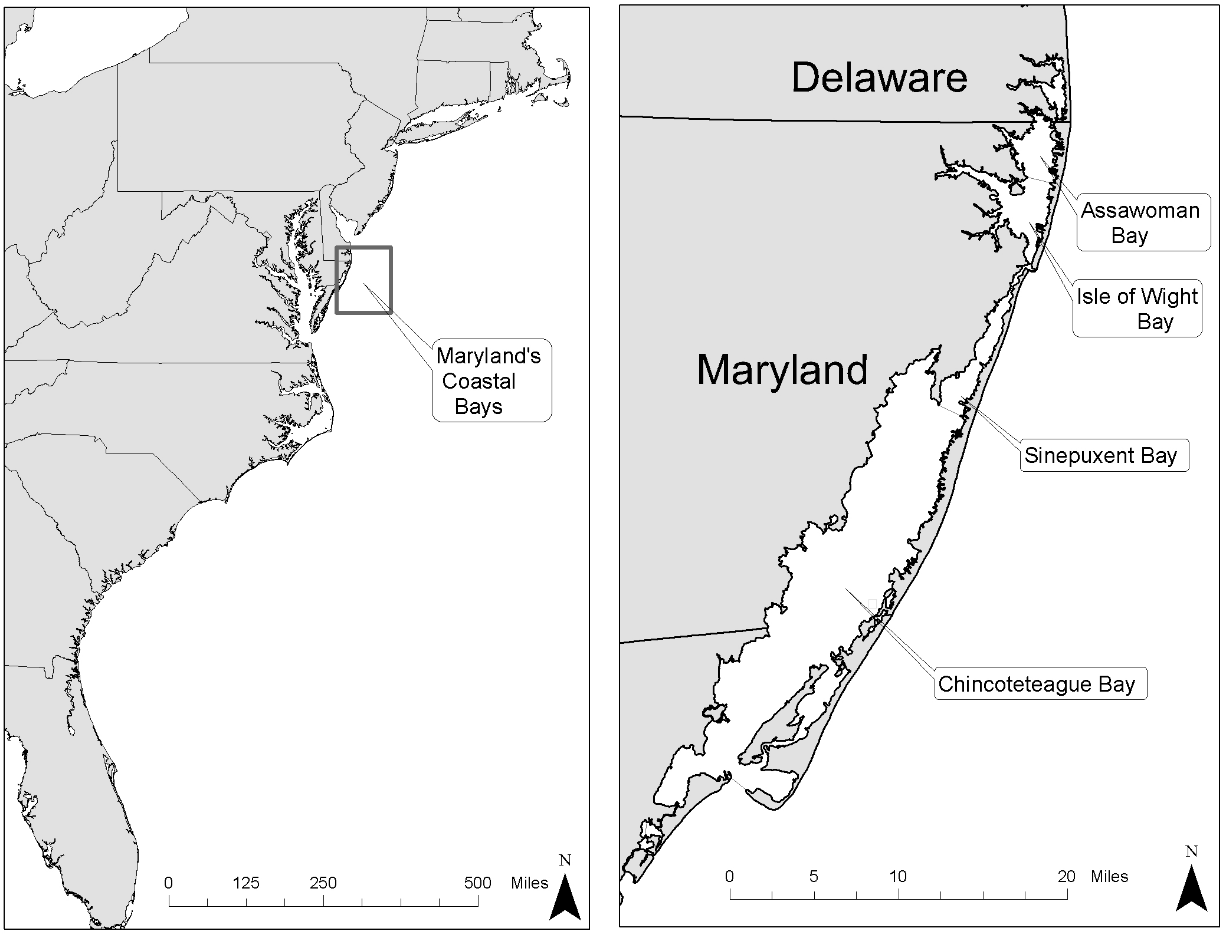
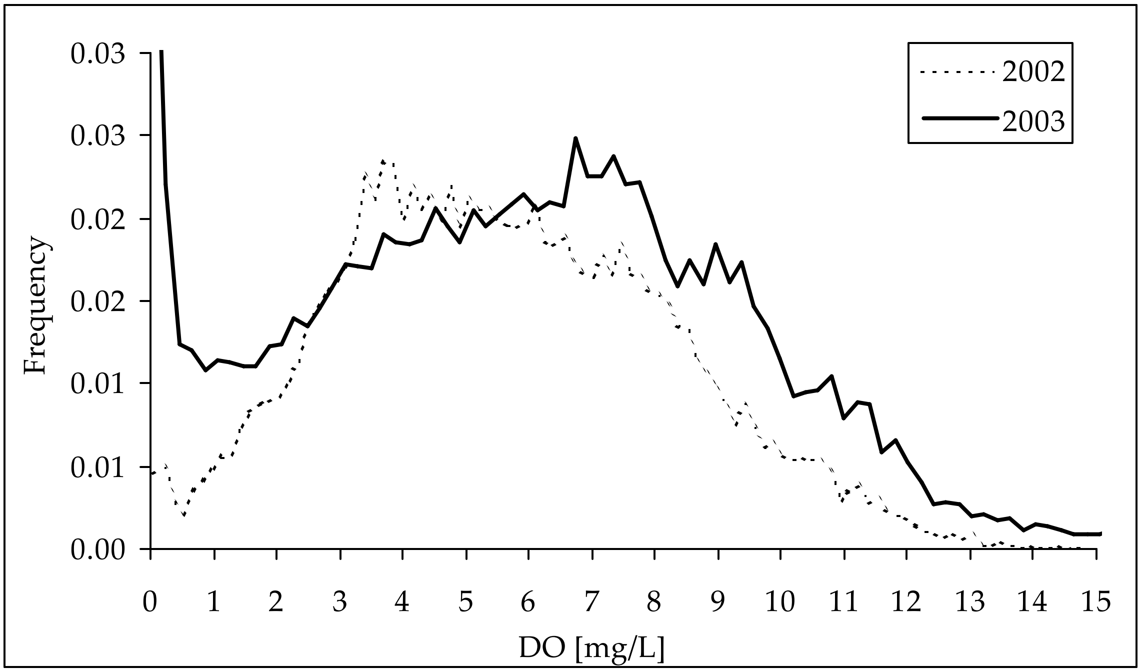


Figure 1 – Maryland's coastal bays



<u>Figure 2</u> – Empirical DO distribution for 2002 and 2003 in the Isle of Wight Bay. The spikes near zero are evidence of relatively frequent DO crashes, which generally occur after large algal blooms.

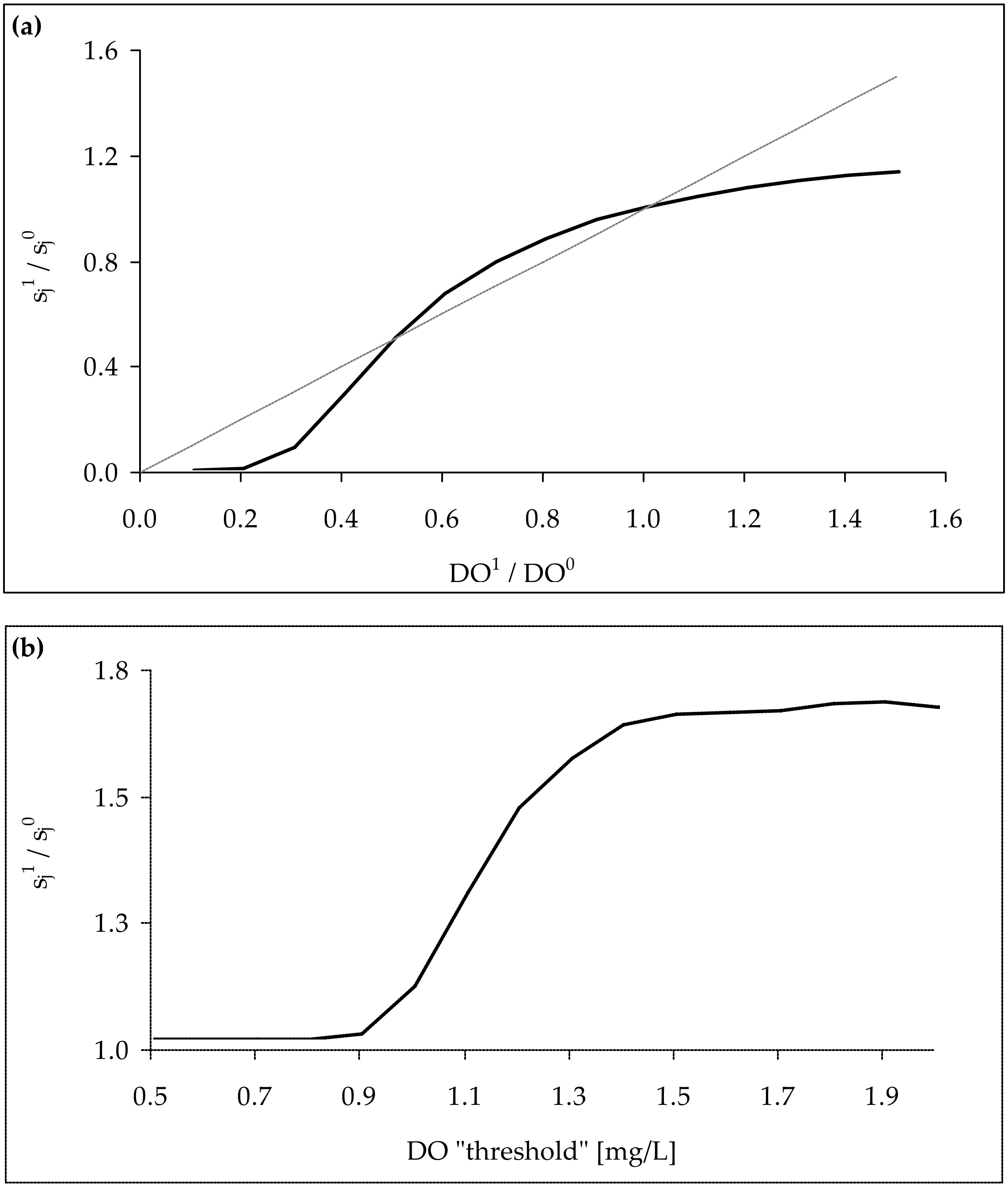


Figure 3 -- Effect of changes in DO conditions on the predicted survivorship of young juvenile summer flounder based on Equation [12]. Panel (a) shows the proportional change in survivorship as a function of proportional changes in DO levels at all times. Panel (b) shows the proportional change in survivorship as the minimum DO level reached at any time is increased.

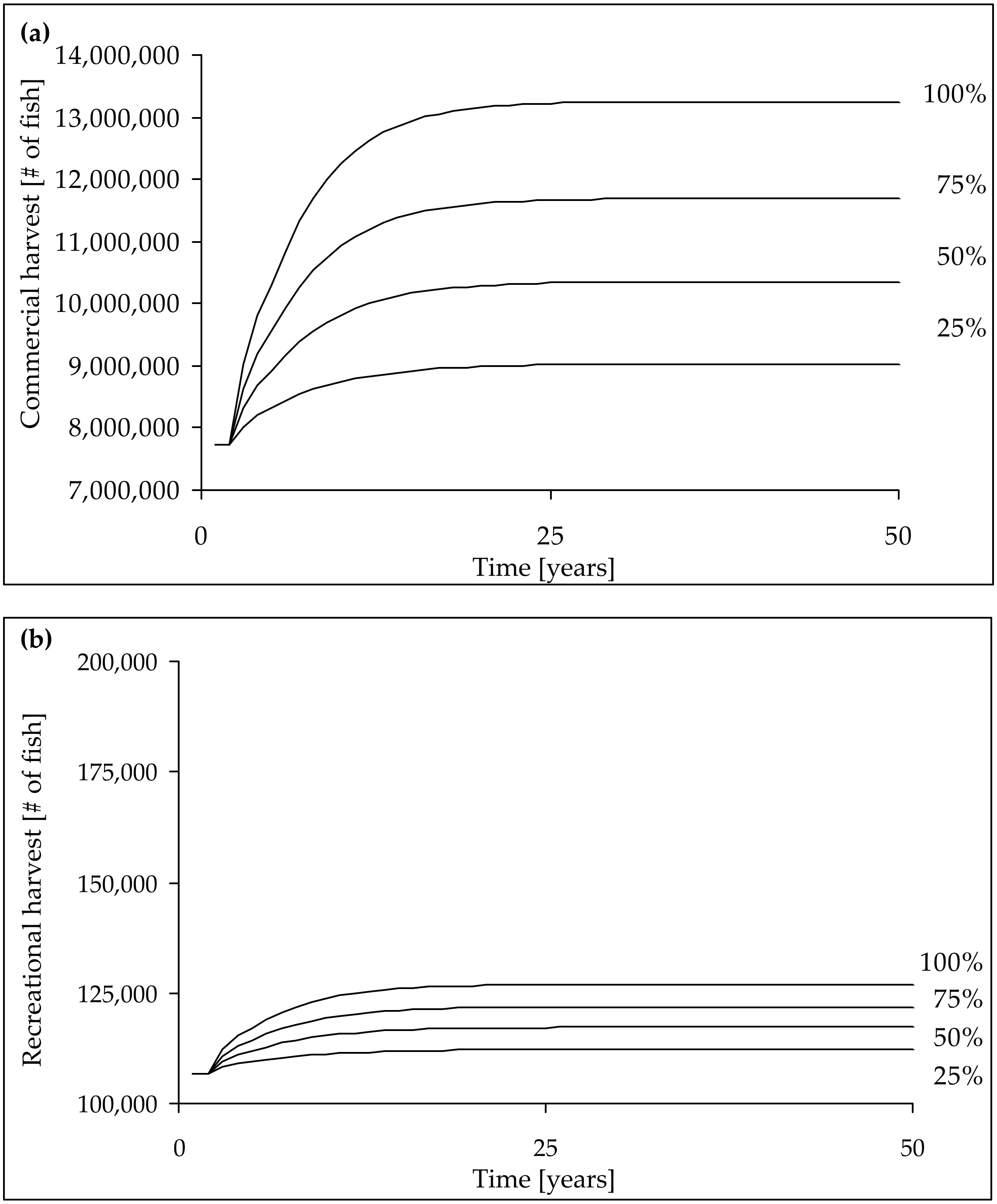
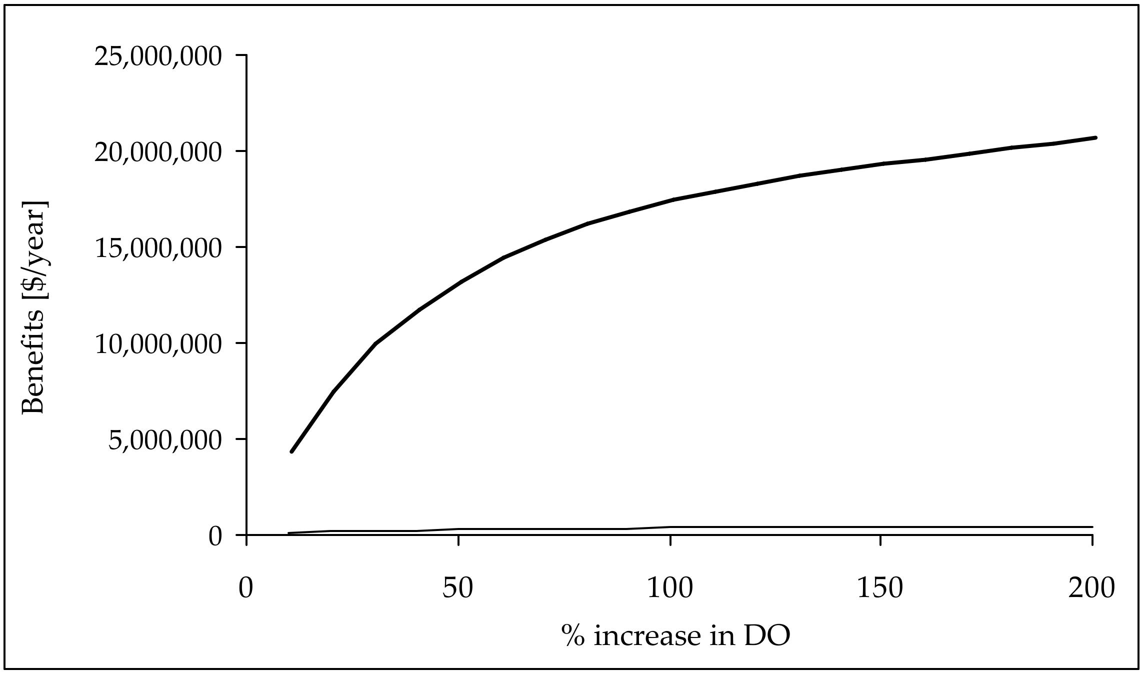
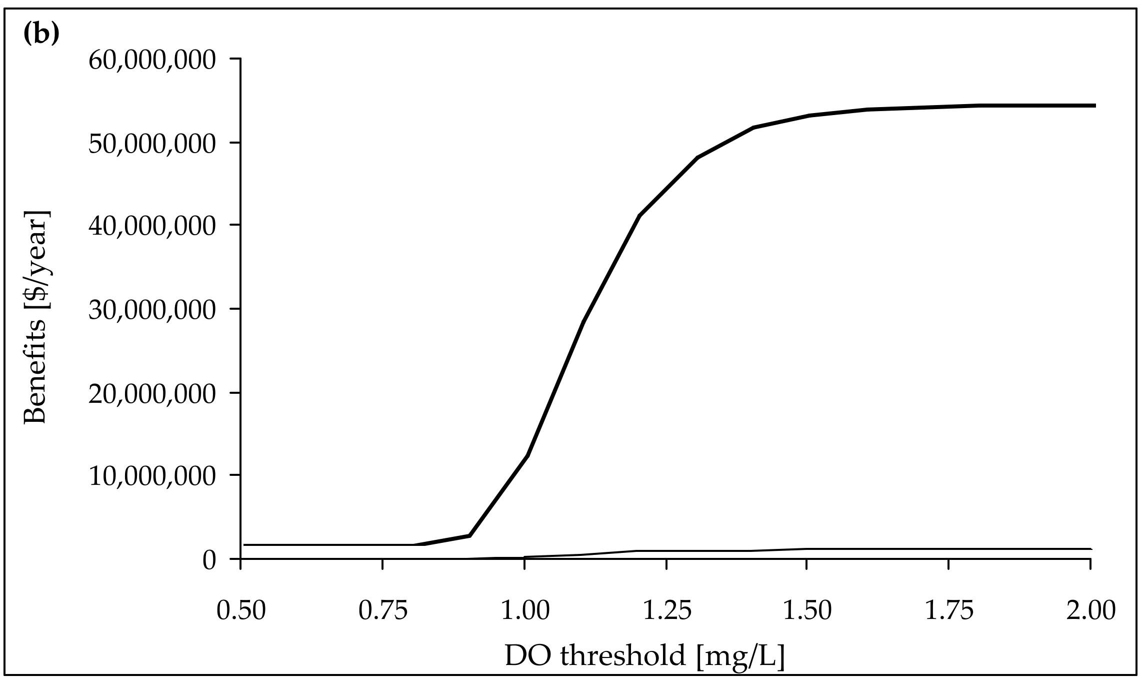


Figure 4 – Forecasts of commercial harvest levels (panel a) and recreational harvest levels in the study area (panel b) from reductions in the frequency of occurrences of DO below 2 mg/L ranging from 25% to 100%.





<u>Figure 5</u> – Aggregate benefits in the study area (light line) and out of the study area (heavy line) from DO increases. Panel (a) shows the effect of proportional increases in DO at all times. Panel (b) shows the effect of eliminating the occurrences of DO below a minimum threshold value.