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ABSTRACT:

We estimated the economic benefits of the Chesapeake Bay TMDL to commercial fish harvesters and consumers, recreational anglers, and other outdoor recreators. To forecast the impacts of the TMDL on harvested fish and shellfish stocks in the bay and connected Atlantic coast waters, we used a summary of judgments from an expert panel and a multi-species model of Chesapeake Bay fisheries. We estimated benefits to consumers in commercial fish markets using a multi-stage inverse demand system, which models price as a function of exogenous supply and accounts for substitution possibilities between 13 different species and as many as five regions. Models were estimated using monthly harvest data from the years 1991 to 2011. The estimated parameters of the inverse demand systems were then used to calculate compensating and equivalent variation from the changes in harvests between the baseline and TMDL scenarios. To estimate producer surplus changes, we assumed that fishing effort will remain fixed at recent levels in each fishery, so harvesting costs do not increase due to the TMDL. The resulting estimates of commercial fishing benefits range between $3 and $26 million per year. We also examined the implications of alternative assumptions about the management regime in each fishery, including fixed effort, open access, and maximum sustainable surplus.

We calculated benefits to recreational anglers using a linked participation and site-choice recreation demand model. The model was estimated using angler intercept survey data from the Marine Recreational Fisheries Statistics Survey (MRFSS). Catch rates were calculated using historic reported catch from the MRFSS. We accounted for the sample selection bias caused by the non-random intercept survey sampling design using weights based on historic visitation frequencies at each intercept site. The intercept data were used to estimate a random utility site-choice model, and counts of trips from respondent zip codes were used to estimate a negative binomial participation model conditional on the inclusive value of all sites as estimated by the site-choice model. The resulting estimates of recreational fishing benefits range between $5 and $59 million per year.

We used a separate recreation demand model to estimate the benefits associated with other outdoor recreation activities. The model was estimated using aggregate data on the total number of visitors to national and state parks in Maryland, Virginia, and Delaware. The aggregate visitation data alone are insufficient to estimate all parameters of the model, so these data were supplemented with survey data on the number of recreation trips taken to the Chesapeake Bay collected from a random sample of individuals in the study area. The marginal effects of water quality on recreators' site choices were estimated in a second-stage regression, using estimates of site-specific constants from the first-stage site-choice model as the dependent variable and measures of average water quality conditions and other fixed site attributes as explanatory variables. The central estimates of the outdoor recreation benefits (exclusive of recreational fishing) are between $105 to $280 million per year.

1 Introduction

The ecological impacts of the Chesapeake Bay TMDL and subsequent effects on fishery harvests will stem from changed water quality and aquatic habitat conditions for a variety of finfish and shellfish species in the bay and connected waters. By reducing the loads of nitrogen, phosphorus, and sediment to the Chesapeake Bay, the extent and frequency of hypoxia—defined as dissolved oxygen (DO) levels less than 2 mg/l—which creates "dead zones" where fish and shellfish cannot survive, are expected to diminish. Also, water clarity is expected to increase, thereby allowing sunlight to penetrate deeper into the water column supporting the growth of submerged aquatic vegetation. These water quality and habitat improvements may in turn lead to increased abundances of aquatic species that support commercial and recreational fisheries, and enhance the aesthetic character of the water and connected habitats that support other non-consumptive recreational activities including boating, swimming, hiking, and wildlife viewing. At the same time, reductions in nutrient loads to the Chesapeake Bay could lead to diminished productivity of primary producers at the base of the aquatic food web, thereby reducing the biomass of fish species at higher trophic levels that the ecosystem can support. The net effects of the TMDL on aquatic living resources and associated ecosystem services in the Chesapeake Bay will depend in part on the strength of these potentially countervailing influences.

We estimated the market and non-market benefits of the TMDL for commercial fisheries and outdoor recreational activities using: a summary of expert judgments regarding the potential impacts of the TMDL on fish stocks in the Bay; predictions of changes in dissolved oxygen levels and associated habitat volumes for 14 aquatic species using outputs from a mechanistic water quality model of the bay; a multi-species fishery simulation model; and three economic valuation models. To estimate the commercial fishery benefits of the TMDL, we combined predictions of fishery harvest changes due to the TMDL with an inverse consumer demand system model to project changes in market prices of fish and associated changes in consumer surplus and harvester revenues. We estimated the benefits of the TMDL to recreational anglers using a linked participation and site-choice recreation demand model. We calculated the benefits of the TMDL associated with improvements in other outdoor recreation activities using a separate recreation demand model. The model was estimated using data on the total number of visitors to federal and state parks in Maryland, Virginia, and Delaware, plus supplemental data from two independent stated preference surveys that asked respondents how many outdoor recreation trips they took to the Chesapeake Bay in the previous 12 months.

This paper is organized as follows. Section 2 describes the potential impacts of the Chesapeake Bay TMDL on aquatic living resources using three sources of information: 1) a summary of judgments by a panel of experts that was convened to provide guidance at an early stage of this study, 2) estimates of changes in habitat volumes due to the TMDL for a set of 14 key aquatic species in the bay, and 3) an ecological simulation model that relates habitat volumes to steady-state harvest levels for 14 key fish and shellfish species that reside in the bay. Section 3 describes how we estimated the commercial fishing benefits of the TMDL, and Section 4 describes how we estimated the potential benefits of the TMDL to outdoor recreators.

2 Ecological impacts of nutrient loads to the Chesapeake Bay

The most important effects of nutrient loads in estuaries might be classified into two broad categories: the "enrichment" effects of enhanced primary productivity at the base of the food web as more nutrients are added to the system (Nixon and Buckley 2002), and various "over-enrichment" effects that can occur when the rate of nutrient inflows exceed the capacity of the ecosystem to assimilate those nutrients in living biomass (e.g., Caddy 2000, Cloern 2001, Baird *et al*. 2004). Specifically, if increased nutrient inputs lead to phytoplankton growth that outstrip the capacity of grazers to consume them, then the surplus blooms of phytoplankton will subsequently die and be decomposed by microbes in the water column. This decomposition process consumes dissolved oxygen from the water column, thereby depleting the oxygen available for respiration by other aquatic organisms such as fish and shellfish. The resulting hypoxic conditions lead to physiological stress for many aquatic species, and so diminish the available habitat for mobile species that can avoid hypoxic areas and may reduce the growth rates or increase the mortality rates of sessile species that are not able to avoid hypoxic waters.

Water quality problems related to nutrient over-enrichment—including depressed dissolved oxygen levels, increased frequency of algal blooms, reduced water clarity, and loss of submerged aquatic vegetation—have been observed in over 60% of estuaries in the United States (Bricker *et al*. 1999). A wide range of studies have documented various ecological impacts of elevated nutrient loads to the Chesapeake Bay (e.g., Breitburg 1992, Boesch *et al*. 2001, Kemp *et al*. 2005, Seitz *et al*. 2009). However, a comprehensive ecological simulation model relating nutrient loads to water quality conditions and ultimately to aquatic species' population dynamics and fishery outcomes for the Chesapeake Bay was not available at the time of this study.[1]

To examine the effects of nutrient loads on aquatic living resources in the Chesapeake Bay, we began by assembling a panel of six water quality and fishery experts at a workshop held in January, 2013. Based on the proceedings of that workshop, we developed a summary of the experts' judgments regarding the potential changes in stock sizes of 15 key aquatic species in the bay. We also developed an ecological simulation model to project the long-run average (steady-state) fishery stocks and harvest levels in the Chesapeake Bay and connected coastal waters with and without the TMDL. The key model inputs include estimates of the area or volume of suitable habitat for each species associated with simulated water quality conditions under a baseline scenario, intended to represent status quo policies, and a TMDL scenario, which involves reductions of nitrogen, phosphorus, and sediment loads to the bay of 25%, 24%, and 20%, respectively, relative to 2009 levels.

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[1] The leading contenders for existing models that might be used to assess the ecological impacts of the TMDL are the Chesapeake Bay Fisheries Ecosystem model (CBFEM) which is described in Appendix A, and Atlantis, which is described in Appendix B. The CBFEM is based on the Ecopath with Ecosim (EwE) platform (Christensen *et al.* 2009) and simulates the growth, reproduction, fishery harvests, and predator-prey relationships among more than 50 aquatic species, life-stages, or species groups in the Chesapeake Bay. The model was developed mainly to evaluate existing or proposed fishery management options, and so focuses on "top down" influences including fishing mortality and interactions among species at high trophic levels. The "bottom up" influences on the ecosystem stemming from nutrient loads, habitat conditions, and the details of species interactions at lower trophic levels are highly simplified. In particular, the linkages between nutrient loads and water quality conditions and their influence on species' rates of growth, reproduction, and survival are absent or not well represented in the CBFEM. A means of linking habitat volumes to foraging arenas in the CBFEM is still under development. Appendix A contains a more detailed description of the CBFEM, including some preliminary model experiments conducted in the early stages of this study and proposed innovations to incorporate forcing functions that would better represent the influence of water quality conditions on species' interactions in the model.

The following sub-sections describe each of these elements in more detail. Section 2.1 provides a summary of judgments by the expert panel. Section 2.2 describes the methods used to calculate changes in habitat volumes for each species. Section 2.3 describes the multi-species fishery model for the Chesapeake Bay that was developed for this study.

2.1 Summary of judgments from a panel of experts

At the time of this study, no ecological simulation model that includes the potentially detrimental over-enrichment effects of nutrient loads on fisheries in the Chesapeake Bay was available. To help fill this information gap, we invited seven experts on fisheries and water quality in the Chesapeake Bay to participate in a two-part workshop. Part one of the workshop was a webinar during which several EPA staff described the main elements of our study, and NOAA staff described some exploratory ecological modeling results from the Chesapeake Bay Fisheries Ecosystem Model (CBFEM) (Christensen *et al.* 2009) and the need for more reliable predictions of the effects of water quality conditions on living resources in the bay. In a second meeting, held two weeks later, the experts presented their preliminary responses to the charge questions posed in the webinar and participated in a facilitated discussion of the potential impacts of the TMDL on the major fish and shellfish species in the bay. Panelists also discussed how the ecological impacts could be assessed through specific modifications of existing models or by other approaches. The supplemental information contains a summary of the workshop proceedings plus all relevant workshop materials, including the webinar presentations, charge questions, notes from the facilitated panel discussion, and preliminary and final responses by the panelists to the workshop charge questions.

During the course of the facilitated open discussion, the expert panelists were asked to provide their best professional judgments of potential changes in Chesapeake Bay species' stock sizes with the TMDL relative to current water quality conditions, holding all other influences constant. These judgments were offered in qualitative form—e.g., "large increase," "small increase," etc.—for 15 key species in the Chesapeake Bay, which are listed in the first column of Table 1. The first 12 species were included in this assessment because together they account for roughly 90% of the annual average total commercial fishing revenues for all fish and shellfish species landed in the Chesapeake Bay between 2007-2011. The final 3 species do not currently account for a significant fraction of commercial fishing revenues, but are historically or ecological important for other reasons. American shad supported an important commercial fishery in the Chesapeake Bay region as recently as the 1950s, but current abundances are near an all-time low and the species is the target of a variety ongoing restoration efforts.[2] Blue catfish were introduced to Chesapeake Bay tributaries from mid-western rivers starting in the 1960s. They are an invasive species that have few natural predators and in some areas are crowding out many native fish species.[3] American eels supported a large commercial fishery in the region in the past but have been in decline for several decades. Their main cause of decline is thought to be upstream habitat loss due to dams, but they are also sensitive to low DO conditions and so may benefit from water quality improvements.[4]

Table 1 shows the qualitative assessments by the expert panelists for all 15 species. Most panelists offered a judgment for most species, but "no opinion" was indicated in 17 of the 105 cases. To convert the panelists' qualitative judgments into quantitative assessments, we assigned values between -25% to 25% changes to each qualitative indicator of change as specified in the caption for Table 2, which shows the transformed quantitative assessments and the corresponding averages across all panelists for each species in the final column. The upper bound of 25% was based on one panelist's conjecture that most species would likely respond less than proportionally to nutrient load reductions, and so because the TMDL nutrient load reductions targets are 25% or less, we assumed that a 25% increase in stock size would be a "large" response. Similarly, we mapped "small increase [decrease]" responses to 10% [-10%] changes, and "no measurable change or small increase" responses to 5% changes. We chose these figures because they are convenient round numbers that, in our judgment, seemed consistent with the implied intent of the expert panelists. The set of averaged quantitative assessments in Table 2, based on the transformed qualitative judgments of the expert panelists shown in Table 1, represents our central informed guess of the potential impacts of the Chesapeake Bay TMDL on commercially harvested fish and shellfish species. We will refer back to these assessments in later sections for comparison to the predicted

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[2] http://www.bayjournal.com/article/despite_massive_restoration_efforts_american_shad_have_not_ rebounded

[3] http://www.dnr.state.md.us/fisheries/catfish/pdfs/Invasive_Catfish_%20Fact_Sheet.pdf

[4] http://www.delawareriverkeeper.org/resources/Factsheets/Action_Plan_American_Eel_March_2012.pdf

changes in habitat volumes for each species (section 2.2) and the predicted changes in steady-state stock sizes from the ecological simulation model (section 2.3).

2.2 Changes in habitat volumes

An important causal pathway through which nutrient and sediment load reductions may affect the health of aquatic living resources in the Chesapeake Bay is through the influence of water quality conditions on the amount of suitable habitat for those species. To calculate habitat volumes, we used predictions of water quality conditions from the Chesapeake Bay Estuary Model (CBEM) and water quality tolerance ranges for one or more life stages of 14 important aquatic species in the Chesapeake Bay based on a review of the literature by Schlenger (2012) plus our own supplemental review of other studies and reports. The CBEM is based on the CH3D (curvilinear-grid hydrodynamics three-dimensional) model (Kim 2013) coupled with the CE-QUAL-ICM water quality model (Cerco and Cole 1993, Cerco and Noel 2013). The CBEM predicts a suite of water quality parameters, including temperature, salinity, and dissolved oxygen (DO) levels, on an hourly time step. The simulation domain covers the Chesapeake Bay and the adjoining tidal waters of its major tributaries. The water quality conditions projected by the model are interpolated to a three-dimensional network of roughly 30 thousand cells, each with a vertical dimension of 1 meter and representing a total volume of nearly 10^{11} cubic meters (about 20 trillion gallons).[5] The temporal resolution of the water quality model output corresponds to the long-standing environmental monitoring program that collects the data used to calibrate and validate the model. Water samples are taken from 175 locations in the bay, including points along a transect that runs the length of the bay, generally twice per month during May through September and once per month at other times (Tango and Batiuk 2013). The model is calibrated to match these water quality measurements, and the standard outputs for all simulated scenarios also match the temporal resolution of this monitoring program.

Schlenger (2012) conducted a systematic literature review of the physiological tolerances of Chesapeake Bay living resources. All available studies for 30 species life-stages were

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[5] The precise figures for our calculations of habitat volumes are 29,950 cells and 7.9×10^{10} m^3. These figures are a few percent smaller than the total volume represented in the model, but we only use model cells for which projections for dissolved oxygen, salinity, and temperature are all available.

synthesized to define ranges of “required” and “optimal” habitat conditions. “Required habitat was defined as a range of environmental conditions outside of which mortality of a specific species will occur. Optimal habitat was defined as a range of environmental conditions outside of which physiological stress will occur, which may manifest through reductions in growth, movement, or other metabolic processes” (Schlenger 2012 p 14).

We used outputs from the CBEM to calculate habitat volumes in each month for all 14 species represented in the multi-species fishery model (MSFM), which is described below in section 2.3. The species represented in the MSFM are listed in the first column of Table 3. To calculate habitat volumes, we used the water quality tolerances synthesized by Schlenger for as many species life-stages as possible. For those species life-stages not included in Schlenger’s review, we either transferred values from other closely related species reviewed by Schlenger or extracted tolerance ranges from other primary studies, as described in the notes at the bottom of Table 3. In each time step, the required and optimal habitat volumes for each species life-stage were calculated by summing the volume of all cells in the CBEM whose predicted levels of temperature, salinity, and DO all fell within the required and optimal ranges for each species life stage in turn, divided by the total volume of all cells in the model containing water quality predictions for that month.[6] These are normalized habitat volumes, indicating the fraction of the water in the bay that meets the required or optimal habitat conditions for each species life-stage in each month.

Figure 1 shows graphs of the estimated habitat volumes in each month under simulated baseline and TMDL conditions based on rainfall data spanning the years 1991-2000, which are taken as representative of current climate conditions, for the juvenile stages of the 14 species that appear in the MSFM. Table 4 lists the average monthly habitat volumes during the months of May through September ignoring DO limits, under historic conditions, and under simulated baseline and TMDL conditions for all life stages of all 14 species. The increases in the average habitat volumes between the baseline and TMDL scenarios range from

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[6] Blue crabs, eastern oysters, northern quahog clams, and summer flounder are benthic species that live on or near the bay floor and typically avoid shallow waters, so when calculating habitat volumes for these species we considered only the deepest model cells (each of which are 1 meter in height) at each *x*-*y* location in the bay where the depth is greater than 2 meters.

close to 0% to roughly 5%, so these results suggest that the percentage increases in the volume of suitable habitat for aquatic species in the Chesapeake Bay generally will be substantially less than the nutrient and sediment load reductions, which are between 20% and 25%.[7]

2.3 Multi-species model of Chesapeake Bay fisheries

With no off-the-shelf model suitable for this study, we developed a new ecological simulation model designed specifically to examine the effects of changes in habitat volumes on harvested aquatic species in the Chesapeake Bay to supplement the expert judgments described in section 2.1. This model comprises a set of single-species models, each in the mold of a highly simplified fishery stock assessment model (Kilduff *et al*. 2009), but with a two-patch structure to account for regular migrations of fish in and out of the bay. The model is highly simplified since it excludes any explicit representations of predator-prey or other interactions between species. However, it does include explicit (quasi-)mechanistic relationships between habitat volumes and the survival rates of the early life-stages of each life species.

The purpose of the model is to translate a set of spatially delineated changes in water quality conditions in the Chesapeake Bay into changes in habitat volumes and exposure to low dissolved oxygen (DO) levels, and to estimate the potential long-run impacts of those changes on the average stock sizes and harvest levels for a set of important aquatic species. The predicted changes in average harvest levels were used as one set of inputs to the commercial and recreational fishing valuation models described below in sections 3 and 4. The model is inherently dynamic, but in this study we focused exclusively on steady-state outcomes.[8] To calibrate the model we used estimates of recent average harvest levels and fishing mortality rates, assuming that these levels and rates are consistent with steady-state conditions, plus several other strategic simplifying assumptions described below.

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[7] The scope for improvements indicated by the predicted changes in habitat volumes is substantially smaller than what we inferred from the judgments expressed by the panel of experts, as summarized in section 2.1. Possible reasons for this difference include that the experts may have anticipated larger changes in habitat volumes than indicated by the water quality model (predictions which were not available at the time of the workshop), or that the experts anticipated improvements in other dimensions of habitat quality not represented by the modeled habitat volumes, such as expanded cover of submerged aquatic vegetation.

[8] One reason we focused on equilibrium conditions is that the simulated water quality data represent future steady-state conditions after the various hydrologic lags (involving groundwater transport, among other processes) have played out, the timing of which are highly uncertain.

We refer to the model as a multi-species model because it comprises a set of independent single-species population models, all based on a common functional form and set of simplifying assumptions but distinguished by species-specific data on landings, fishing mortality rates, and life-history parameters. The model does not account for predator-prey or other species interactions, so it is not a community or ecosystem model.

The model is designed to account for two modes of action by which dissolved oxygen may affect aquatic species: the direct lethal effects of low DO levels on eggs and larvae, and the indirect effects of elevated competition among juveniles who avoid hypoxic areas by crowding into suitable normoxic (non-hypoxic) habitat. The model also is designed to accommodate two key processes that can lead to spillovers of the benefits of water quality improvements in the Chesapeake Bay to other Atlantic bays and estuaries and coastal waters: passive larval transport and fishery overlaps.

There are at least two other potentially important omissions from the model. First, the model does not account for the enrichment effect of nutrients on lower trophic levels that may increase the overall productivity of the ecosystem (e.g., Nixon and Buckley 2002, Breitburg 2002, Breitburg *et al*. 2009, Rose *et al*. 2009). This is important because if the enrichment effect is strong enough then reducing nutrient loads to the bay could reduce the bottom-up productivity of the ecosystem by an amount sufficient to outweigh ecological benefits of the reduction in hypoxia, which would lead to a net effect of lowered stocks and fishery harvest levels. Second, the model does not include any mechanisms that could lead to hysteresis associated with eutrophication (Carpenter *et al*. 1999, Scheffer *et al*. 2001). As discussed by Smith and Crowder (2011 p 2259), explicitly accounting for the possibility of hysteresis could lead to higher or lower estimated benefits of nutrient load reductions, since this could delay the policy impact of nutrient reductions but also would account for the reduced probability of regime shifts associated with the dynamics of oxygen consumption through the process of phytoplankton decomposition. Appendix C sketches a simplified mathematical model that illustrates this dynamic. These mechanisms were not included in the model used in this study due to data limitations and time constraints, so future work should examine the implications of these processes in a more comprehensive model.

The following sections describe the structure of the multi-species fisheries model and the various simplifying assumptions that underpin it, the data used for calibration, and results from applying the model to estimate the fishery impacts of the Chesapeake Bay TMDL. We will introduce notation as it is needed, but for convenience Table 5 includes a complete list of variables and parameters with their definitions and measurement units. All equations in the following sub-sections 2.3.1—2.3.3 are species-specific, but to avoid notational clutter we will refrain from appending a species index on all parameters and variables in each equation.

2.3.1 Model

To account for potential spillovers of the benefits of water quality improvements in the Chesapeake Bay to connected waterbodies, we partition the total Atlantic coast stock of each species into two compartments, or "patches," one comprised of adults originating in (i.e., either spawned in or that resided in during an early sub-adult stage) the Chesapeake Bay, N_B, and the other comprised of adults originating in other Atlantic bays or estuaries, N_A. If an early sub-adult stage occurs in coastal waters, then the model assumes that a fixed fraction, γ_A, of the larvae that are spawned by other Atlantic estuary adults and survive to adulthood, R_A, will be dispersed to other estuaries (with the remainder dispersed to the Chesapeake Bay), and a fixed fraction γ_B of the larvae that are spawned by Chesapeake Bay adults and survive to adulthood, R_B, will be dispersed to the Chesapeake Bay (with the remainder dispersed to other estuaries). We refer to γ_A and γ_B as "larval retention coefficients," since they indicate the fraction of larvae spawned by adults in patch A or B that return to patch A or B, respectively.

At steady-state the number of larvae that survive and return as recruits to patches A and B equals the number of adults in patch A and B, respectively, that die each year, i.e.,

$$\gamma_A R_A + (1-\gamma_B)R_B = N_A(1-e^{-M-F}) \text{ and } \gamma_B R_B + (1-\gamma_A)R_A = N_B(1-e^{-M-F}), \qquad (1)$$

where M and F are the adult natural and fishing mortality rates, assumed common to both patches and constant over time.

The model incorporates two potential mechanisms by which hypoxia may affect the survival rate of fish and shellfish during their early life stages when aquatic species are thought to be most sensitive to low DO conditions. The first mechanism involves elevated density-independent mortality due to the acute lethal effects of low DO on early sub-adult life stages (eggs, larvae, and some young juveniles) that are unable to avoid hypoxic areas. The second mechanism involves elevated density-dependent mortality due to intensified competition for food resources among later sub-adult stages (older juveniles) who are able to actively avoid hypoxic areas by crowding into normoxic habitat. Movement patterns and behaviors vary among the species in the model, but they all spend some portion of their early life stages in the Chesapeake Bay so we assume both mechanisms will affect all but one species in the model. (The sole exception is bluefish, which is affected by water quality in the Chesapeake Bay during the juvenile stage but not the egg and larval stage. See Appendix D.) We further assume that the effects of water quality on each species can be represented by the estimated species-specific habitat conditions during the months of May through September for both early life stages. The following sub-sections describe in detail the simplifying assumptions for the two sub-adult life stages that are treated as vulnerable to low levels of dissolved oxygen.

Eggs and larvae

We assume that eggs are deposited in a pulse once per year, and the number of spawners, S, is equal to the average number of adults over the year:

$$S = \left(\frac{1 - e^{-M-F}}{M + F}\right) N. \tag{2}$$

We denote the average number of eggs deposited per spawner as a, so the number of eggs deposited is αS. The available water quality data are resolved on a monthly time scale and we make no attempt to downscale the water quality data to a lower temporal resolution, so we make the simplifying assumption that water quality conditions change from month to month, as indicated in the data, but are constant within each month. We denote the fraction of the water in the Chesapeake Bay in month k that is suitable for survival and growth for a species during a particular life-stage, i.e., the species' life-stage "habitat volume," as V_k. This fraction varies by species and life-stages as it is based on the volume of water in the bay with DO, salinity, and temperature conditions that lie within the tolerance range for each species life-stage, as described in section 2.2. We denote the instantaneous mortality rate in suitable habitat as β [yr $^{-1}$]. We assume that deposited eggs are effectively sessile; i.e., any movements due to drift are negligible compared to changes in the size of their habitat volumes. Next we assume that water quality conditions, including DO levels, that are outside of a species' "required" tolerance range are surely fatal while water quality conditions within a species' required tolerance ranges have no adverse effects.[9] Putting these assumptions together implies that the average survival rate over the egg and larval stage is proportional to the extent of habitat that is always suitable (never experiences conditions outside of the species' "required" tolerance ranges) over the entire duration of the egg and larval stage, which we will label $\tilde{V}$, i.e.,

$$J_0 = \alpha e^{-\beta K/12} S\tilde{V} = aS\tilde{V}, \tag{3}$$

where J_0 is the number of individuals that survive to the end of the larval stage (i.e., the beginning of the juvenile stage), a is a composite parameter comprising the average egg production per spawner and the average survival rate over the K-month duration of the egg and larval stage under suitable water quality conditions.

Juveniles

Unlike eggs and larvae, which we treat as immobile or passively drifting particles, we assume that juveniles are active swimmers that can avoid unsuitable habitats and so will experience no lethal effects of direct exposure to low DO conditions. However, we assume that during this stage competition for food resources is pronounced and so the instantaneous per capita mortality rate of juveniles, $-\frac{dJ/dt}{J}$, during month k is proportional to the density of juveniles in the suitable habitat, J/V_k, i.e., $dJ/dt = -\frac{b/K}{V_k}J^2$ (Gurney and Nisbet 1998 p 125).[10]

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[9] These two assumptions—eggs are sessile, and exposure to monthly average water quality conditions outside of a species' "required" tolerance range is surely fatal—are among the key simplifying assumptions in the model. A more realistic model would account for spawning habitat preferences with respect to water depth and water quality conditions. While admittedly crude, the simplifying assumptions used here are the most parsimonious, transparent, and biologically plausible assumptions we could devise that allow us to close the model of the egg and larval stage while avoiding the need for detailed spatially-explicit modeling of movements of water, eggs, and larvae or a continuous dose-response relationship for each species.

[10] Note that b/K is the constant of proportionality between the per capita mortality rate and density. We have normalized this rate by K, the length of the juvenile stage in months, for convenience, so that K itself does not appear in the final stock-recruitment equation. See equation (4) in the main text.

(Note that we are re-using the notation $k = 1, 2, \dots, K$ to indicate months that comprise the juvenile stage, and in this section t has units of months.) To determine the number of juveniles that survive to the end of month k we separate variables to get $dJ/J^2 = -\frac{b/K}{V_k}dt$. Next, we integrate both sides to get $-1/J = -\frac{b/K}{V}t + D$, where D is a constant of integration. Then we solve for the number of juveniles as a function of time get $J(t) = 1/\left(\frac{b/K}{V_k}t - D\right)$. The number of juveniles at the beginning of the first month of the juvenile stage is J_0, so we use this initial condition to solve for the constant of integration to get $D = -1/J_0$, then plug this back into the equation of motion for juveniles to get $J_1 = J_0/\left(1 + \frac{b/K}{V_1}J_0\right)$. By the same logic, the number of juveniles that survive to the end of the second month is $J_2 = J_1/\left(1 + \frac{b/K}{V_2}J_1\right)$. We can combine these equations to write J_2 as a function of J_0, V_1, and V_2: $J_2 = J_0/\left(1 + \frac{b}{K}\left[\frac{1}{V_1} + \frac{1}{V_2}\right]J_0\right)$. Repeating this for all months of the juvenile stage and using equation (3) to substitute for J_0, we arrive at the following stock-recruitment function:

$$R = \frac{aS\tilde{V}}{1 + abS\,\tilde{V}/\hat{V}}, \tag{4}$$

where $\hat{V} = K/\sum_{k=1}^{K} V_k^{-1}$ is the harmonic mean habitat volume over the duration of the juvenile stage (which should not be confused with $\tilde{V}$, the "always suitable" habitat volume relevant for the egg and larval stage, as defined above). Note that (4) has the form of a Beverton-Holt function, which follows from the assumption that individuals continuously track their suitable habitat and their mortality rate continuously adjusts to match their density throughout the juvenile life stage (e.g., Turchin 2003 p 55). Also note that the harmonic mean will always be less than the arithmetic mean habitat volume, so using the arithmetic mean as a sufficient statistic in the juvenile survival model would understate the influence of transitory poor water quality conditions on the population and harvest outcomes. This is a consequence of the serially multiplicative nature of the density-dependent survival process assumed to characterize the juvenile stage.

The model includes a stock-recruitment function for each species in the form of equation (4) for both patches distinguished by patch-specific parameters a_A, b_A, a_B, and b_B. The always suitable and harmonic mean habitat volumes, $\tilde{V}$ and $\hat{V}$, will appear only in the equation for patch B (Chesapeake Bay) recruits because we assume that water quality conditions in other bays and estuaries will remain unchanged, so water quality conditions affecting the survival of sub-adults in estuaries other than the Chesapeake Bay are subsumed in a_A and b_A.

We can gain insight into the behavior of this model by considering a closed population confined to a single patch with constant habitat volume, V. In this case, the steady-state abundance is

$$N = \frac{aV - M - F}{ab(1 - e^{-M-F})}. \tag{5}$$

Assuming the fishing mortality rate is held fixed, the proportional effect of an increase in habitat, which we will refer to as the "habitat elasticity" of the stock, is

$$\frac{\partial N}{\partial V}\frac{V}{N} = \frac{aV}{aV - M - F}. \tag{6}$$

which will be greater than 1 as long as the condition for persistence, $aV > M + F$, is satisfied. Therefore, increasing the amount of suitable habitat by one percent in this model will generally lead to an increase in the steady-state stock abundance that is greater than one percent.[11] It is also easy to show that as V decreases from one (all available habitat is suitable for survival and growth) toward the minimum level consistent with persistence, the habitat elasticity will increase, so the proportional effect of water quality improvements will be higher when water quality is low or when the fishing mortality rate is high.

Fishery overlaps

Some fisheries are partly or wholly prosecuted in coastal waters where adults from the two patches may be mixed, and fish from patch A or B harvested in coastal waters may be landed at ports in either patch, not necessarily in their parent estuary. To accommodate such potential harvest spillovers in the model, we assume that a fixed fraction ω_A of har-

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[11] Gurney and Nisbet (1998 p 142-143) found an analogous result using a Beverton-Holt model to examine the impact of a power plant on a coastal fishery.

vested fish that originated in other bays are landed at ports in other bays (with the remainder landed at Chesapeake Bay ports), and a fixed fraction ω_B of harvested fish that originated in the Chesapeake Bay are landed at ports in the Chesapeake Bay (with the remainder landed at other ports). We will refer to ω_A and ω_B as "harvest retention coefficients." Fishery harvests, H_A and H_B, and landings, Q_A and Q_B, are related as follows:

$$Q_A = \omega_A H_A + (1 - \omega_B) H_B \ \text{ and } \ Q_B = \omega_B H_B + (1 - \omega_A) H_A. \tag{7}$$

Finally, Baronov's catch equation (Ricker 1975 p 12-13) for the total harvest is

$$Q_A + Q_B = B(N_A + N_B), \tag{8}$$

where $B = \frac{F}{M+F}(1 - e^{-M-F})$ is the fishing exploitation rate (the fraction of the stock abundance at the beginning of the period that is harvested during the period).

2.3.2 Data and calibration procedure

We assembled data and parameter estimates from the fisheries literature sufficient to specify all model state variables and parameters except a_A, a_B, b_A, and b_B. The information used to calibrate the free parameters of the model are presented in Table 6 and Table 7. These include average Chesapeake Bay and Atlantic coast commercial and recreational fishery landings between 2008 and 2012, estimates of current fishing mortality rates, fishing mortality rates that would achieve maximum sustained yield (MSY), adult mortality rates, and larval and harvest retention coefficients for each species in the model. Detailed descriptions of the data sources and parameter input assumptions are provided in the notes at the bottom of Table 6 and Table 7. In brief, we obtained data on fishery landings from NOAA's commercial and recreational fishery statistics online database, and we obtained estimates of natural mortality rates and current and MSY fishing mortality rates from the most recent stock assessments for each species or other sources when no stock assessment was available. We could find very little information to specify the larval and harvest retention coefficients, so the values we use for these parameters are rough estimates based on our interpretation and broad extrapolation of a small number of relevant studies; in many cases these are little more than crude guesses. Given the large uncertainties surrounding the spillover effects, we calibrated the model both with and without the spillovers included. With this in mind, when calibrating the model with spillovers included we set the retention coefficients near the low end of what seem to be plausible ranges for each species (to maximize the strength of the spillovers) while still allowing the model to fit the available data on landings given the other more reliably estimated biological parameters for each species. Therefore, the results from the with- and without-spillovers cases are intended to span a wide range of possible outcomes conditional on the loosely constrained larval and harvest retention coefficients.

Given the data and parameters in Table 6 and Table 7, the remaining unknown parameters of the model—a_A, a_B, b_A, and b_B—can be calibrated using a simple search algorithm such as that described in Text box 1.

Text box 1. Calibration algorithm

1. Calculate the stock abundances in each patch using:

   $$N_A = \frac{1}{B}\left[\frac{(1-\omega_B)Q_B-\omega_B Q_A}{1-\omega_A-\omega_B}\right] \text{ and } N_B = \frac{1}{B}\left[\frac{(1-\omega_A)Q_A-\omega_A Q_B}{1-\omega_A-\omega_B}\right].$$

   These expressions can be derived from equations (7) and (8). Note that we must have $\omega_A + \omega_B \neq 1$.

2. Set the initial values for the juvenile-per-spawner coefficients to:

   $a_A = (F_{MSY} + M^2)/M$ and $a_B = (F_{MSY} + M)^2/(M\hat{V})$.

   These expressions correspond to the a's that would obtain in the model without spillovers.

3. Calculate the density-dependent juvenile mortality coefficients using:

   $$b_A = \frac{M+F}{1-e^{-M-F}}\left\{\frac{1-\gamma_A-\gamma_B}{[N_B-\gamma_B(N_A+N_B)](1-e^{-M-F})} - \frac{1}{a_A S_A}\right\}$$
   and
   $$b_B = \hat{V}\left\{\frac{1-\gamma_A-\gamma_B}{[N_A-\gamma_A(N_A+N_B)](1-e^{-M-F})} - \frac{1}{a_B S_B \hat{V}}\right\}.$$

   These expressions can be derived using equations (1) through (4).

4. Use numerical simulation to calculate the predicted values of the fishing mortality rate that achieves MSY, $\hat{F}_{MSY}$, and the predicted landings in both patches, $\hat{Q}_A$ and $\hat{Q}_B$, conditional on the current values of a_A, a_B, b_A, and b_B.

5. Calculate $SSE = \left(\frac{\hat{F}_{MSY}-F_{MSY}}{F_{MSY}}\right)^2 + \left(\frac{\hat{Q}_A-Q_B}{Q_B}\right)^2 + \left(\frac{\hat{Q}_B-Q_B}{Q_B}\right)^2$.

6. Adjust a_A and a_B, repeating steps 3-6, to minimize SSE .

2.3.3 Results

The model results are shown in Table 8 and Table 9. Table 8 shows the habitat elasticities for each stock with and without spillovers. The elasticities were estimated by using the calibrated model to simulate steady-state landings for each species conditional on a 1% increase in the habitat volume for each species in each month, subtracting from this the predicted landings (which, by construction, match the recent historic average landings), and dividing the difference by the predicted landings. Note that these elasticities are conditional on the recent historic stock sizes and habitat volumes for these species and will generally decrease (increase) as the stocks and habitat volumes increase (decrease). The first column of numbers in Table 8 confirms the analytical result implied by equation (6): the habitat elasticities of the Chesapeake Bay stocks are all greater than one percent when no spillovers are included. The elasticity estimates in this case range from 1.1% (Northern quahog clams) to 7.2% (eastern oyster). The second and third columns of numbers in Table 8 show that the elasticities are affected by the inclusion of spillovers. The direction and magnitude of the changes vary by species and apparently depend on the share of the total Atlantic harvest that is represented by the Chesapeake Bay harvest. For example, the Chesapeake Bay harvest for both black sea bass and bluefish represent a relatively small fraction of their respective total Atlantic coast harvests, and for both of these species the habitat elasticities are significantly lower in the model with spillovers than without. The elasticities in the with-spillovers case range from 0.2% (American shad) to 5.6% (eastern oyster).[12]

Table 9 shows the results of applying the model to the Chesapeake Bay TMDL. These results were generated by using the calibrated model to estimate landings conditional on habitat volumes associated with the baseline and TMDL water quality conditions, as estimated by the CBEM (see section 2.2). As in Table 8, the results are presented as proportional

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[12] Note that the Chesapeake Bay stocks of Northern quahog clams, eastern oysters, white perch, and blue catfish are all assumed to be independent of other Atlantic coast stocks of these species, so all retention coefficients for these species were set to 1. Therefore, any differences in the results between the without- and with-spillovers cases for these species are solely due to variability in the iterative calibration procedure, which includes a random search component.

changes in long-run steady-state landings, i.e., baseline landings were subtracted from the TMDL landings and the difference was divided by the baseline landings for each species. The first and second columns of numbers show the change in the always-suitable and harmonic mean habitat volumes during the months of May through September for each species. The next four columns contain the estimated proportional changes in Atlantic and Chesapeake Bay landings both with and without spillovers. The final two columns show the share of total Atlantic landings and harvest, respectively, contributed by the Chesapeake Bay harvest under the baseline scenario. Comparing these columns gives an indication of the strength of the spillover effects. The proportional changes in Chesapeake Bay landings range between 0.3% (Atlantic croaker) and 4.5% (spot) without spillovers and 0.3% (summer flounder) and 4.4% (spot) with spillovers. The predicted changes for each species, with and without spillovers, were used in the commercial and recreational fishery valuation models described in section 3 and section 4.1.

3 Commercial fishing benefits

Changes in consumer and producer surplus in the commercial fishing sector will depend on the changes in the abundance of the fish stocks, which were described in section 2, as well as the slopes of the demand and supply curves and the nature of the management regime in each fishery. We estimated consumer and producer surplus changes in the commercial fishery sector using a statistical model of consumer demands for 14 of the most important commercially harvested species in the Bay.

For the central estimates of commercial fishery benefits developed in this study, we assumed that fishing effort will not change as a result of the TMDL. This implies that in each fishery the fishery management authority imposes binding restrictions on effort and will not adjust these restrictions after the TMDL is implemented. It also implies that the nature of the restrictions and the fishing harvest production function in each fishery are such that positive fishing rents can be sustained. This means that the fishing mortality rate imposed on each harvested species and the cost of harvesting in each fishery will remain fixed. This assumption greatly simplifies the estimation of commercial fishery benefits because it ignores any behavioral responses on the part of the harvesters or fishery managers. This allows us to estimate the change in stock size and harvest level independent of the economic details of each fishery, as in section 2.3. With estimates of harvest changes in hand, we then calculated price changes using our estimated system of demand equations, and finally we calculated changes in producer surplus (which equal the change in harvest revenues under this simplifying assumption) and consumer surplus using the baseline and policy prices and quantities.

The assumption of fixed fishing effort that underlies our primary estimates of commercial fishing benefits is consistent with the instructions given to the panel of experts described in section 2.1, and it was the basis of the estimates of harvest changes using the ecological simulation model described in section 2.3. Under this assumption, the cost of fishing is the same under the baseline and TMDL scenarios for each species, so the benefits to producers—the change in producer surplus, ΔPS—corresponds to the difference in fishery revenues between the scenarios, i.e.,

$$\Delta PS = p_1 Q_1 - p_0 Q_0 \,. \tag{9}$$

Our commercial fishing benefit estimates are based on the consumer surplus changes estimated using the inverse demand system described in section 3.3 plus the producer surplus (harvest revenue) changes calculated using equation (9).

3.1 Open access and rent dissipation

The assumption of fixed fishing effort makes the model more tractable but arguably sacrifices an important element of realism, so we also investigated how our results could change under an alternative assumption that may be more realistic. Most major Atlantic coast fisheries, including those in the Chesapeake Bay, are managed by the Atlantic States Marine Fisheries Commission (ASMFC). The ASMFC develops a fishery management plan (FMP) for each major stock or group of closely related stocks. Each FMP is developed using data from fishery stock assessments in light of biological reference points for each species (Kilduff *et al.* 2009). Biological reference points are target or threshold fishing mortality rates or spawning stock biomass levels, which, if transgressed, indicate that the stock is being overfished (Gabriel and Mace 1999). The management measures set forth in the FMPs typically include a combination of fishing gear restrictions, season limits, annual quotas allocated among states and between commercial and recreational sectors, and recreational fishing size and bag limits. In a fishery with an annual quota, commercial harvesters are required to report their landings on a continual basis as the season progresses, and when the total reported landings reaches the annual quota the fishery is closed for the remainder of the year.

Fisheries managed in this way can be described as "regulated open access" fisheries (Homans and Wilen 1997). If no restrictions are imposed on who participates in the fishery—anyone who qualifies to obtain a license can participate, and there are no binding limits on the number of licenses that can be awarded—then there will be an incentive for harvesters to enter the fishery as long as the expected profits from fishing are positive. Each additional harvester takes a portion of the quota, thereby increasing the scarcity of fish and as a consequence increasing the cost of catching the remaining fish for all other harvesters. The optimal response by other participants is to increase their effort by, for example, fishing longer hours using more gear or labor or other variable inputs. The end result of this process can be an overcrowded fishery and a "race for fish" (Hilborn 2007) where, in the extreme, profits for all harvesters are driven down to zero, i.e., all rents from fishing are dissipated (Gordon 1954, Scott 1955). In such cases fishing will be a break-even proposition, where the total revenues from fishing just cover the total costs of fishing.[13] An arguably more realistic assumption for many of the fisheries examined in this study is that all rents are dissipated both before and after the water quality improvements caused by the TMDL, in which case only consumers would realize a surplus gain.[14]

Under the fixed effort assumption the total surplus gain corresponds to the area *under* the inverse demand curve between the initial and final quantities, whereas under an open access assumption the total surplus gain corresponds to the area *behind* the inverse demand

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[13] Rents may accrue to the producers during the transition from the initial to the new steady-state water quality conditions if the rate of entry of additional fishermen is not too fast, but eventually the extra rents will be dissipated under the new steady-state conditions (e.g., Smith and Crowder 2011). Also, under certain conditions some profits may be sustained in regulated open access fisheries, for example if the variable cost of effort is increasing in effort (Anderson 1985) or if variable fishing inputs are poor substitutes (Deacon *et al*. 2011).

[14] A notable exception to the generalization that Chesapeake Bay fisheries are managed under a regulated open access regime is the Atlantic menhaden fishery, which is prosecuted almost exclusively by a single firm, Omega Protein. Depending on the nature of the relationship between the firm and the management authority, the Atlantic menhaden fishery might be best characterized as a sole-owner fishery. The closest approximation to a sole-owner fishery model that we examine in this study appears in Appendix D, where we use a simplified Shafer-logistic framework to estimate commercial fishery benefits under three alternative assumptions about the management regime in each fishery, including one in which the total economic surplus (consumer + producer surplus) is maximized.

curve between the initial and final prices. As shown in Appendix E, in a simplified logistic-Schaefer fishery the relative size of these alternative surplus measures depends on the price elasticity of demand and the fishing mortality rate relative to the species' biological growth rate; specifically:

$$\frac{\Delta S_{OA}}{\Delta S_{FE}} = \left[\frac{\left(1+\sigma_{OA}\frac{\Delta K}{K}\right)^{1-\varepsilon}-1}{\left(1+\frac{\Delta K}{K}\right)^{1-\varepsilon}-1}\right] \text{ and } \sigma_{OA} = \frac{\partial Q}{\partial K}\frac{K}{Q} = \frac{1-\frac{F}{r}}{\varepsilon+\frac{F}{r}(1-2\varepsilon)} \qquad (10)$$

where ΔS_{OA} and ΔS_{FE} are the changes in total (consumer + producer) surplus in the open access and fixed effort models, respectively, ε is the own price flexibility of demand, σ_{OA} is the elasticity of the harvest level with respect to the carrying capacity of the stock under open access, $\Delta K/K$ is the proportional change in the carrying capacity due to the water quality improvement, F is the fishing mortality rate, and r is the intrinsic growth rate of the species. As shown in Appendix E , considering our estimates of the ε's, F's, and r's for the species examined in this study, we would expect an open access model to produce smaller estimates of total surplus changes than the fixed effort model. Therefore, to the degree that the fisheries examined here are more realistically characterized as open access fisheries than restricted effort fisheries, our primary estimates of commercial fishing benefits could be biased upward. However, an important caveat to this result is that we derived equation (10) under the assumption that the demand for each species is independent of demand for other species, i.e., no species is a complement to or substitute for any other species in the representative consumer's utility function. If some species are complements (substitutes), then an increase in the supply of one species can shift out (in) the demand curve for other species, in which case equation (10) would need to be generalized.[15]

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[15] It would be desirable to conduct a more comprehensive analysis that links the multi-species fishery model, or a more realistic ecosystem model, to the system of consumer demands described in section 3.3 to examine several alternative assumptions about the management regime in each fishery. However, we would not expect that the reasonable alternatives would give drastically different results, so due to time and resource constraints we have left such an analysis for future work. In the meantime, Appendix D contains a preliminary supplemental analysis based on a simplified framework comprised of a Schaefer-logistic model with a constant elasticity demand curve applied to each fishery but no cross-price effects among species. We used the framework to compare the value of the TMDL under a fixed effort regime, an open access regime, and a regime in which the sustainable economic surplus is maximized in each fishery.

3.2 *Preliminary ballpark estimates*

Before proceeding to the detailed analysis based on our empirical consumer demand model, we generated a set of preliminary ballpark estimates of the commercial fishing benefits by multiplying the baseline price by the predicted change in harvest for each fishery and summing these products across all fisheries: $\sum_i p_{0i} \Delta Q_i$. We know that this will not be an accurate estimate because it does not account for any anticipated changes in prices.[16] However, we would not expect it to be very far away from estimates derived from a more realistic model because we do not anticipate very large price changes in this setting. (This conjecture can be examined using results from our statistically estimated inverse demand model, as reported in section 3.4.)

Table 10 shows the ballpark estimates under three ecological impact scenarios: averaged expert judgments and the MSFM with and without spillovers. We estimated the baseline prices and quantities, p_0^A, p_0^B, Q_0^A, and Q_0^B, using the average revenues and landings at Chesapeake Bay ports and at all Atlantic coast ports for each species between 2008-2012. The bottom row of the table shows the ballpark estimates of the total surplus changes under these scenarios, which range from roughly \$2 to \$15 million per year.

The differences among the results in Table 10 follow directly from the differences among the ecological scenarios. The summary of expert judgments suggests the blue crab stock could increase significantly, the Atlantic menhaden stock could decrease due to the reduced nutrient inputs to the Bay, eastern oysters could increase substantially (in percentage terms) from their very low current levels, and several species may not be measurably

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[16] This ballpark estimate would be an upper-bound on the change in total (consumer + producer) surplus under the following conditions: 1) fishing effort, and therefore the cost of fishing, does not change between the baseline and TMDL scenarios; 2) no harvests will decrease under the TMDL scenario; and 3) any complementary relationships (i.e., positive cross-price effects) in the system of fish and shellfish demands are negligible. To see why $\sum_i p_{0i} \Delta Q_i$ gives an upper-bound under these conditions, first note that condition 1 means that the change in producer surplus will equal the change in fishery revenues and (in a single-market setting, ignoring any cross-price effects) the change in total surplus will equal the full area under the demand curve between the initial and final quantities. Therefore, if all demand curves were flat then $\sum_i p_{0i} \Delta Q_i$ would equal the change in total surplus. Condition 2 means that if one or more demand curves are not flat, then some prices will go down, in which case the increase in total surplus would be less than $\sum_i p_{0i} \Delta Q_i$. Condition 3 means that no prices will increase (as long as no quantities decrease), and some prices could be reduced even further than with no cross-price effects, in which case the increase in total surplus would be reduced further still. The upper-bound estimates are preliminary because at this point in the report we do not yet know if all three of the stated conditions will hold.

impacted (largely due to their wide distributions outside of the Chesapeake Bay). Due mainly to the high value of the blue crab harvest and the large predicted increase in blue crabs by the experts on average, the ballpark estimate of benefits is largest under this scenario. The ballpark benefits based on outputs from the MSFM are significantly smaller, mainly due to the relatively modest predicted increases in habitat volumes, which are on the order of 5% or less. The proportional changes in stock sizes are larger than the predicted increases in habitat volumes for each species in the MSFM without spillovers, as explained in section 2.3.1, but the predicted stock increases are still substantially smaller than the averaged quantified expert judgments. Furthermore, the largest predicted increases are not for the highest valued stocks (blue crabs and Atlantic menhaden).

The following subsections describe more refined estimates of consumer and producer surplus changes based on an empirical model of consumer demands for fish and shellfish species harvested in the Chesapeake Bay and nearby regions. We applied the consumer demand models to the same set of ecological impact scenarios used above.

3.3 Market demand for fish and shellfish

Barton and Bettendorf (1989) argue that demands for perishable goods like produce, meat, and seafood should be examined using inverse demand equations in which supply is treated as exogenous and price adjusts to clear the market. In addition to perishability, seafood harvest depends in a large part on gear and effort restrictions and capital investments that are made long before the boat leaves the dock. For these reasons we choose to treat quantity changes exogenously and analyze demand for Chesapeake Bay harvest using inverse demand systems.

We use a distance function approach to estimate the associated welfare impacts to consumers resulting from the three fish and shellfish harvest projections: expert elicitation, multi-species model and the multi-species model with spillovers to the rest of the East Coast. Kim (1997) show how the distance function can be used to estimate the exact welfare measures compensating and equivalent surplus from a change in supply by determining what scaling of the quantity vector is necessary to reach a reference utility level. Conveniently, the parameters of the distance function are estimated with an inverse demand system which we will use to forecast changes in the price vector as part of the producer surplus calculation.

When estimating consumer welfare impacts of increased Chesapeake harvest it is important to account for two types of substitution that may occur as a result of improved water quality. First, consumers may substitute between species as their relative prices change with the supply vector. The analysis must also account for the availability of substitutes from other harvest regions and consumer preferences among those regions. Substitution between harvest regions becomes more important when water quality improvements in the Chesapeake Bay result in spillover effects elsewhere on the East Coast. The most straightforward approach would be to treat the harvest of species from each region as an individual commodity and estimate a single inverse demand system with an equation for each. Many of the species in this analysis are harvested in 4 or 5 different regions which would create a demand system with 80 equations or more and thousands of parameters in order to account for cross-quantity effects. A more tractable alternative is to use a two-stage budgeting approach (Edgerton 1997). In the first stage consumers allocate total expenditures among the species of interest. In the second stage, consumers allocate the expenditures on each species among the different regions from which they are harvested. The two-stage budgeting approach requires that we assume weak separability between harvests from different regions. That is to say, a change in harvest of a given species from a particular region can affect the market clearing price for a different species from a different region but only through the allocation of expenditures among species. The multi-stage approach allows us to model changes in how consumers allocate income among different types of seafood and account for preferences among regions. Harvests from other regions may be less than perfect substitutes for Chesapeake harvest and the two-stage approach will capture those asymmetries and account for them in the welfare analysis.

To better represent consumers' substitution patterns and simplify estimation, the Chesapeake species are divided into three groups: high value fish, shellfish, and low value fish. Other popular species that are not harvested from the Chesapeake Bay but could substitute for Chesapeake Bay species are included in each group. High value fish are finfish with an average price over $1 per pound while low value fish have a price of less than $1 per pound. The constituents of each group are shown in Table 11. A two stage demand system is estimated for each group. The presumption here is that changes in the supply of Chesapeake Bay species will affect demand for other species within that group but not across groups. Further, we assume that total expenditures on each group will remain constant in real terms. As real income increases over time, real expenditures on normal goods are expected to increase. In that regard, assuming constant real expenditures will result in a conservative welfare estimate.

3.3.1 Consumer Welfare Estimation in Quantity Space

The distance function is dual to the expenditure function and can be considered a normalized money metric utility function (Kim 1997) that measures how the quantity vector must be scaled in order to reach a reference utility level. Given the quantity vector X and a direct utility function $U = U(X)$, the distance function $D(U, \boldsymbol{X})$ is defined as

$$D(U, \boldsymbol{X}) = max_t\{t > 0: U(\boldsymbol{X}/t) > 0\} \tag{11}$$

which gives the maximum amount by which the quantity vector must be divided in order to just reach the indifference surface.

Exact measures of consumer welfare can be calculated from the distance function by finding the difference between $D(U, \boldsymbol{X})$ evaluated at different quantity vectors, say $\boldsymbol{X}^0$ and $\boldsymbol{X}^1$, given a reference utility level U^0 or U^1. Specifically, the normalized compensating and equivalent surplus from exogenous quantity changes are

$$cs = D(U^0, \boldsymbol{X}^1) - D(U^0, \boldsymbol{X}^0), \tag{12}$$

and

$$es = D(U^1, \boldsymbol{X}^1) - D(U^1, \boldsymbol{X}^0). \tag{13}$$

Since (12) and (13) are normalized by total expenditures, the non-normalized, or absolute, consumer welfare measures are found by multiplying by total expenditures, Y.

Utility is not observable so the distance function cannot be estimated directly. To estimate the parameters of the distance function it is necessary to choose a functional form and derive the resulting system of compensated inverse demands. These demands can be estimated empirically and then used to recover the parameters needed for welfare analysis.

A number of functional forms for the distance function have been used in the literature. A popular choice is the Inverse Almost Ideal (IAI) specification (Eales and Unnevher 1994, Moschini and Vissa 1992),

$$ln[D(U,\boldsymbol{X})] = a(\boldsymbol{X}) - Ub(\boldsymbol{X}) \tag{14}$$

where $a(\boldsymbol{X}) = \alpha_0 + \sum_i \alpha_i ln(X_i) + \frac{1}{2}\sum_i \sum_j ln(X_i)ln(X_j)$, $b(X) = \beta_0 \prod_i X_i^{\beta_i}$ and X_i are elements of the quantity vector X. Because $D(u,X)$ is homogeneous of degree one in X, the following restrictions apply: $\sum_i \alpha_i = 1$, $\sum_j \gamma_{ij} = \sum_i \gamma_{ij} = 0$, and $\sum_i \beta_i = 0$. Also, without loss of generality, $\gamma_{ij} = \gamma_{ji}$ (the symmetry property).

According to Shephard's theorem, the first derivatives of the distance function with respect to the quantities yield normalized compensated inverse demands

$$\widehat{\boldsymbol{P}} = \partial D(U,\boldsymbol{X})/\partial \boldsymbol{X} = a(u,\boldsymbol{X}), \tag{15}$$

where $\widehat{\boldsymbol{P}} = \boldsymbol{P}/Y$, $\boldsymbol{P}$ is a vector of prices, and Y is total expenditure on $\boldsymbol{X}$. Applying Shephard's theorem to the IAI form of the distance function, recognizing that $\partial lnD/\partial lnX_i = (\partial D/\partial X_i)(X_i/D)$, and that utility maximization requires that $D = 1$, yields the compensated inverse demand system in expenditure share form

$$W_i = P_i\boldsymbol{X}_i/Y = \alpha_i + \sum_j \gamma_{ij}\ln(X_j) - U\beta_j b(X). \tag{16}$$

But u is unobservable, so using $D = 1$, invert the distance function in (11) for $U = a(X)/b(X)$ and plug into the inverse demands, yielding

$$W_i = \alpha_i + \sum_j \gamma_{ij} ln(X_j) - \beta_i lnQ, \tag{17}$$

where $lnQ = a(X)$. Expression (17) represents a non-linear system with cross-equation restrictions on the parameters to satisfy homogeneity and symmetry.

3.3.2 Accounting for preferences among species and harvest regions with a two-stage budgeting model

The welfare impact of a change in X_i will depend on the total expenditures for that species and changes in the supply of other species to the extent that there are cross-quantity effects. The two-stage demand system allows us to account for consumers reallocating expenditures among species and harvest regions in response to the change in supply from one or more regions. A two-stage demand system is estimated for each of the three species groups. In the model description that follows no distinction is made for species groups in order to simplify the notation. The estimation and operations that follow are performed in the same way for each group. Time subscripts have also been omitted from the following model descriptions but estimation relies on monthly harvest and price data.

First Stage: Species-level inverse demand system

Within each species group, species cross-quantity effects will be captured by the first stage system which includes an inverse demand function for each species. If a supply shift for one species has an effect on the market clearing price, and thus the expenditure share, for another species in the same group, this information is required to project welfare changes based on the results of the second stage. An IAI demand system with an equation for each species $i = 1,2, \ldots , n\, i = 1,K\ ,n$ will provide estimates of expenditure shares

$$W_i = \alpha_i + \textstyle\sum_j \gamma_{ij} ln(X_j) - \beta_i lnQ, \tag{18}$$

Evaluating expression (18) at the new quantity vector will provide the expected reallocation of total expenditures on all species of interest via the new expenditure shares, w_i. The baseline and policy expenditures on each species are required to estimate compensating and equivalent surplus, as we will show below.

Second Stage: Regional inverse demand systems

The second stage will estimate an inverse demand system for each species and each system will contain an equation for each region from which that species is harvested. This stage will capture the substitutability between harvests of the same species from different regions. For example, if blue crabs harvested from any region other than the Chesapeake are poor substitutes for Chesapeake blue crab than welfare impacts from a change in Chesapeake harvest will be larger than if very close substitutes were available. In fact, if consumers were completely indifferent regarding the source of the commodity, only the first stage inverse demand system would be necessary to estimate welfare impacts of a change in Chesapeake harvest because all that would matter to consumers is the change in total supply of that species.

For species *i*, expenditure shares for region *r* = (1,…,R) will be estimated with the same linear inverse demand system described for the second stage

$$w_{(i)r} = \alpha_{(i)r} + \sum_j \gamma_{(i)rs} lnX_{rs} - \beta_{(i)r} lnQ_r \,. \tag{19}$$

Projecting expenditures and welfare impacts

Equations (18) and (19) provide normalized welfare impacts. Absolute household welfare impacts require forecasts of expenditures on each species. In this case, the first stage model allows us to estimate changes in the expenditures within each species group as a result of the TMDL. This difference in baseline and TMDL expenditures must also be accounted for when estimating welfare impacts (Kim 1997). Non-normalized, or absolute, compensating and equivalent surplus are calculated using

$$CV = \sum_i Y_i^0 \{D(U^0, \boldsymbol{X}_i^1) - D(U^0, \boldsymbol{X}_i^0)\} - (Y_i^1 - Y_i^0) \tag{20}$$

$$EV = \sum_i Y_i^1 \{D(U^1, \boldsymbol{X}_i^1) - D(U^1, \boldsymbol{X}_i^0)\} - (Y_i^1 - Y_i^0) \tag{21}$$

where Y_i^0 (Y_i^1) is forecasted expenditures on species i assuming baseline (TMDL) supply, $\boldsymbol{X}_i^0$ ($\boldsymbol{X}_i^1$) is baseline (TMDL) supply vector for species i, and $U(U^1)$ is baseline (TMDL) utility. The results of the first stage model are used to provide baseline and policy expenditure shares

$$W_i^0 = \alpha_i + \sum_j \gamma_{ij} (ln\boldsymbol{X}_i^0) - \beta_i lnQ^0, \tag{22}$$

$$W_i^1 = \alpha_i + \sum_j \gamma_{ij} (ln\boldsymbol{X}_i^1) - \beta_i lnQ^1. \tag{23}$$

W_i^0 and W_i^1 are multiplied by total expenditures Y to provide estimates of Y_i^0 and Y_i^1 which are then used in equations (20) and (21) to generate welfare estimates.

3.3.3 *Total price and scale flexibilities*

In an inverse demand system the price can be used to forecast the new market clearing price vector after a marginal change in supply and provide a theoretical check on the sign and magnitude of the estimated parameters. Eales and Unnevher (1994) derive expressions for the price and scale flexibilities of the IAI demand system and show that they are a fairly straightforward translation of their direct demand counterparts. The multi-stage structure of this analysis, however, adds a layer of complexity to the calculation of flexibilities.

Partial flexibilities can be calculated for any single-stage inverse demand system but only total flexibilities will account for both types of substitution captured by the two-stage model. Moore and Griffiths (2017) derive formulae for total elasticities from a two-stage indirect demand system. One simplifying aspect of this analysis is that all data are reported in the same units (pounds), obviating the need for quantity indices that would be necessary if the units varied among commodities (pounds and gallons, for example).

Flexibilities can be interpreted in a way similar to elasticities. Demand for a commodity is said to be inflexible if a 1% increase in its consumption leads to less than a 1% increase in normalized price. Commodities with negative cross-price flexibilities are said to be gross q-substitutes. Positive cross-price flexibilities are considered gross q-complements.

For the IAI demand model, the partial uncompensated own and cross-price flexibilities for goods *i* and *j* are given by

$$E_{ij} = \frac{\partial \ln[P_i]}{\partial \ln[X_j]} = \frac{\gamma_{ij}+\beta_i(W_j-\beta_j lnQ)}{\mathrm{W}_j} - \delta_{ij} \,, \tag{24}$$

where $\delta_{ij} = 1$ if $i = j$ and $\delta_{ij} = 0$ otherwise. The second stage partial own- and cross-quantity flexibilities for two regions, *r* and *s*, supplying the same species, *i*, would be

$$\mathcal{E}_{(i)rs} = \frac{\partial \ln[p_{(i)r}]}{\partial \ln[x_{(i)s}]} = \frac{\gamma_{(i)rs}+\beta_{(i)r}(w_{(i)s}-\beta_{(i)} lnQ_{(i)})}{\mathrm{w}_{(i)r}} - \delta_{rs} \,, \tag{25}$$

The total own and cross price flexibilities that account for substitution across species and regions are a function first and second stage flexibilities, expenditure shares, and indirect demand parameters. A derivation of the total flexibilities is provided in the appendix to Moore and Griffiths (2017). The own and cross-price flexibility for the i^{th} species in the r^{th} region and the j^{th} species in the s^{th} region is

$$\begin{aligned} \varepsilon_{(i)r,(j)s} = \delta_{ij} &\left[-\delta_{rs} + \{\gamma_{(i)rs} - \beta_{(i)r}[w_{(i)s} + \beta_{(i)s}\ln(Q_{(i)})]\}\left(\frac{1}{w_{(i)r}}\right)\right] \\ &+ \left[\left\{\gamma_{ij} - \beta_i\left(W_j + \beta_j \ln(Q)\right)\left(\frac{1}{W_i}\right)\right\}\right]\left(\frac{x_{(j)s}}{X_j}\right) \end{aligned} \tag{26}$$

The total flexibilities will be used to find the new market clearing price vector after the change in Chesapeake harvest which is needed for the producer surplus calculation.

3.3.4 Data for inverse demand estimation

NOAA National Marine Fisheries Service (NMFS) compiles landings and ex-vessel value data for all commercial fisheries. Both stages of the demand model are estimated with monthly harvest and price data from the years 1990 to 2010. None of the species harvested commercially in the Chesapeake Bay is harvested from the Pacific so only Atlantic regions and Gulf of Mexico are included in the analysis. Further, not all species are harvested to a significant degree in all regions. Table 12 summarizes the harvest and price data.

3.3.5 Inverse demand model estimation results

The first and second stage systems of the inverse demand model are estimated with the Stata statistical package using the *nlsur* estimation function which performs non-linear seemingly unrelated regression. The homogeneity and symmetry restrictions are imposed by estimating a subset of the model parameters and solving for the restricted parameters.

With eighteen equations in the first stage and ten systems of three to five equations each in the second stage, hundreds of individual parameters are estimated in the three two-stage models. To present those results in a more manageable and meaningful way we report the own-price flexibilities for each system of equations. We use Stata's *nlcom* command to evaluate expressions (24) through (26) and report the results in Table 13, 14, and 15. For all 52 equations estimated (18 in the first stage and 34 in the second) the point estimates and their 95% confidence intervals lie below zero, which is what we would expect for downward sloping demand curves.

In order to evaluate consumer welfare we first have to forecast the new expenditure allocations among species Y_i^1 using the results of the first stage expenditure share equations and the policy quantity vectors resulting from the four ecological scenarios. Table 16 shows the baseline expenditure share (means over the range 1990 to 2010) and the predicted change in allocation based on each of the new quantity vectors for each of the species groups.

Of the three fish harvest projections, the expert panel provides estimates with the largest increases in the populations of commercially harvested species with a few exceptions. For black sea bass the expert panel projections show no effect whereas the multi-species model predicts modest increases. Menhaden, however, are a notable exception because the expert panel projections are negative and the multi-species model predicts small but positive changes in population. Also worth noting is that the projections of the multi-species model predicts smaller population increases in the Chesapeake Bay when migrating species are allowed to populate other regions of the east coast. These differences across ecological models are captured in the consumer welfare model and generate markedly different benefit estimates as a result.

Table 17 shows the compensating variation resulting from each of the harvest projections for each of the species groups. The decline in menhaden harvest under the expert elicitation projection results in a negative welfare impact in the low value species group but the gains in the other two groups are large enough to make the compensating variation estimate an order of magnitude larger than the gains from the multi-species model. The difference in welfare estimates between the expert elicitation and multi-species model projections is primarily attributable to the blue crab harvest. The value of that harvest, the importance of the Chesapeake Bay harvest relative to other regions, and a forecast of harvest increases by the expert panel that are roughly an order of magnitude larger than the multi-species models result in a large difference in welfare estimates. Narrowing focus to the two multi-species models, when spillover effects are included the harvest gains in the Chesapeake are smaller but the gains elsewhere on the east coast are such that the compensating variation from this model is more than twice that of the model without spillovers.

3.4 Final benefit estimates

Estimates of total consumer and producer benefits are shown in Table 18. The scenario based on the summary of expert judgments produces the largest benefit estimates of roughly $26 million per year. The results from the multi-species fishery model lead to estimates between $2.9 and $4.1 million per year, depending on whether or not spillovers are explicitly included. The explicit inclusion of spillover effects increases the total benefits by roughly 40%.

As explained in section 3.1, many Chesapeake Bay and other Atlantic coast fisheries may be better described by a model of “regulated open access,” in which case we would expect the harvest increases to be greater than those predicted under the fixed-effort model used here. The total surplus gain in an open access model would consist only of consumer benefits (in steady-state), but still could be larger or smaller than the total surplus gain in a fixed-effort model because the consumer benefits under open access typically will be greater than those under a fixed-effort regime. The illustrative calculations reported in Appendix E suggest that in this setting the total surplus gains in an open access model are likely to be smaller than those in a fixed-effort model. However, in that exercise we ignored any complementarities among species in the system of demand equations, so questions of the size and even direction of the difference between the total benefits in a fixed-effort versus open access model remain open. However, in light of the magnitude of the commercial fishing benefit estimates summarized in Table 18 relative to the TMDL benefit estimates in other categories and the cost estimates, we do not expect that further refinements of the fishery models would have a material impact on the final estimates of the net benefits of the Chesapeake Bay TMDL.

4 Outdoor recreation benefits

4.1 Recreational fishing benefits

In this section we estimate the value of water quality changes resulting from the Total Daily Maximum Load (TMDL) established by the U.S. Environmental Protection Agency in December of 2010 to recreational anglers inside and outside the Chesapeake Bay. Water quality is assumed to affect fishermen positively through its effect on fish populations and therefore expected catch. It may also have a positive or negative site amenity component. These water quality driven changes in species abundance and site characteristics may cause changes in fishermen's per trip utility, the numbers of trips taken, or both.

The most common approach for investigating the welfare effects of water quality changes on recreational fishing valuation is to estimate one of the many versions of recreation demand models in the literature (Freeman 1995, Van Houtven *et al*. 2001.) The common trait of these recreation demand models is that they model trip choice as a function of travel costs and site characteristics. 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), McConnell and Strand (1999), and Morey and Waldman (1998) all employ variations of this strategy. 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 estimates of nitrogen and phosphorus discharge, biochemical oxygen demand, and suspended solids, along with a measure of average catch, are examples of the small number of studies that include both catch rates and water quality. Even fewer studies take the additional step of modeling catch rates as a function of water quality. One example is provided by Massey *et al*. (2006) who model summer flounder population levels, catch rates, and site characteristics as a function of dissolved oxygen. Despite the different ways in which water quality is assumed to enter fishermen's decision processes, all recreational fishing studies we are aware of incorporating water quality find a positive willingness to pay for improvements in water quality.

We estimate a linked random utility maximization (RUM) and negative binomial recreation demand model utilizing several different expected catch specifications and compare their results. The models rely on either a summary of judgments by a panel of experts (section 2.1) or the results of an ecological simulation model (section 2.3) to determine the changes in recreational fish stocks anticipated to happen as a result of the TMDL. Recreational fishing trip data used is from NOAA's National Marine Fisheries Service's (NMFS) Marine Recreational Information Program (MRIP).[17] The first component of the model is a standard random utility maximization site choice travel cost model that includes expected catch and water quality as site characteristics. The model estimates fishermen's choice of which fishing site to visit conditional on the angler already having decided to take a trip. Given that there is no participation decision modeled, welfare effects from this model are estimated on a per person per trip basis. To capture the seasonal participation effect we link the RUM model to a Negative Binomial (NB) trip demand model. The addition of the linked NB model allows estimation of changes in the number of trips taken over a season.

4.1.1 Site Choice Model

The first component of the linked model is a RUM site choice travel cost model. The strength of the model is its ability to model fishermen's choices among a large number of

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[17] See http://www.st.nmfs.noaa.gov/recreational-fisheries/index for in-depth information on the survey.

substitute sites on a given choice occasion. The RUM model estimates changes in fishermen's utility per trip conditional on already having decided to take a trip.

The RUM model may be specified as follows. On given choice occasion in year t, it is assumed the individual *i*'s indirect utility for visiting site *j* can be specified as

$$U_{ijt} = D_{ijt}\alpha + X_{jt}\beta + \xi_{jt} + \varepsilon_{ijt} \,, \tag{31}$$

where D_{ijt} is an individual's travel cost to a site *j* in year t and X_{jt} is a vector of characteristics from site *j* in year t that do not vary across visitors including expected catch.[18] ξ_{jt} represents unobserved site characteristics and ε_{ijt} is an iid extreme value error term. α, β, and ξ_{jt} are parameters to be estimated.[19] Following Murdock (2006), site characteristics that only vary across sites may be characterized as site specific constants

$$\phi_{jt} = X_{jt}\beta_j + \xi_{jt} \,. \tag{32}$$

Representing all site characteristics with alternative specific constants has the advantage of avoiding issues with missing or excluded variables that could potentially bias estimates. Inserting the site specific constant ϕ_{jt} back into equation (31) yields

$$U_{ijt} = D_{ijt}\alpha + \phi_{jt} + \varepsilon_{ijt} \,. \tag{33}$$

The utility function may then be specified

$$V_{ijt} = D_{ijt}\alpha + \phi_{jt} \,. \tag{34}$$

Assuming the error term ε_{ijt} is independently and identically distributed according to the extreme value distribution lead to a probability function for fishermen i in year t at site *j* that may be specified

$$P_{ijt} = \frac{e^{D_{ijt}\alpha + \phi_{jt}}}{\sum_{k=1}^{J} e^{D_{ikt}\alpha + \phi_{kt}}} \,. \tag{35}$$

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[18] Expected catch may also vary over anglers and over choice occasions but data limitations prevent us from attempting to model those dynamics.

[19] The utility function may also contain individual socioeconomic characteristics such as age of the angler or avidity interacted with site characteristics. These are excluded here for notational simplicity.

The probability function specified above relies on a representative sample which may not be present in data being used for this analysis. The MRIP uses a targeted onsite intercept strategy based on expected fishermen usage rather than a random sampling method. Failing to correct for this sampling strategy would therefore result in biased estimates of the population parameters (Breidt *et al.* 2012).

Following Haab and McConnell (2003), a logit estimated with an on-site sample may be weighted using information on sample proportion of trips from each site and the population portion of total trips expected to be taken to each site.[20] Formally, the weighted logit probability function may be specified:

$$P_{ijt} = \frac{\frac{\rho_{jt}}{W_{jt}} e^{D_{ijt}\alpha + \phi_{jt}}}{\sum_{k=1}^{J} \frac{\rho_{jt}}{W_{jt}} e^{D_{ikt}\alpha + \phi_{kt}}}. \tag{36}$$

ρ_{jt} equals the number of observations from site *j* during year *t* divided by the total number of observations collected across all *K* sites in year *t*. ρ_{jt} may be written

$$\rho_{jt} = \frac{N_{jt}}{N_t}. \tag{37}$$

The number of observations from each site comes directly from the data used in estimation. W_{jt} is the total expected number of trips taken by individuals in the population to site *j* in year *t* divided by the total number of expected trips taken by the population across all *K* sites in year *t*

$$W_{jt} = \frac{x_{jt}}{x_t}, \tag{38}$$

where $x_{jt} = \sum_i x_{ijt}$, which is the expected number of total trips taken by individuals to site *j* in year *t*, and x_t is the total number of trips taken by the population to all *J* sites in year *t*. Expected total numbers of trips are calculated from pressure codes found in the NOAA's site registers. Pressure codes, which give a prediction of the mean number of angler fishing trips that an assigned interviewer would expect to encounter at a fishing site, are estimated for

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[20] Another option would be to use the weighted exogenous stratification maximum likelihood and propensity score estimation method developed by Hindsley *et al.* (2011).

both weekend/holidays and weekdays in two month waves and represent the average number of anglers expected to fish in one of four possible fishing modes at a site on an average good weather day during the eight-hour peak use period. The four possible fishing modes are shore fishing, charter fishing, private or rental boat fishing, and head or party boat fishing. The number of expected anglers is first calculated for each mode by month accounting for the types of day found in that month in that year (weekend/holiday and weekday) for each fishing mode. Monthly mode totals are then aggregated to arrive at monthly total expected trips for each site.

The probability function in equation (36) may be used to construct the likelihood and log likelihood functions. The log likelihood function for year t is given by

$$ln(L) = \sum_{i=1}^{N} \sum_{j=1}^{J} \delta_{ijt} ln(P_{ijt}), \tag{39}$$

where $\delta_{ijt} = 1$ if fisherman i visited site j during year t. The log likelihood function is maximized to recover unbiased estimates of α and all ϕ_{jt}'s. For estimation, one alternative specific constant and weight and must be normalized on one alternative.

Again following Murdock (2006), site specific characteristics such as fishermen's expected catch rate and site specific water quality can then be regressed on the estimated site specific constants, ϕ_j, to recover estimates of the fishermen's preferences, β. In order to increase the power of the second stage regression, the year 2008, 2009, and 2010 annual RUM models are first estimated separately and then the estimated annual alternative specific constants are stacked into a single vector and regressed on annual expected catch rates, water quality measures, and year fixed effects. Because the estimated site specific constants appear on the left hand side of the second stage regression it produces unbiased coefficient and standard error estimates.

Calculation of angler's per trip willingness to pay (WTP) for quality changes using the RUM model's estimated parameters begins with calculating recreators' expected maximum utility. Expected maximum utility is simply the log of the exponentiated sum of all site utilities. The expected maximum utility may be defined

$$I_{it} = ln\left(\sum_{j=1}^{J} e^{V_{it}}\right). \tag{40}$$

Changes in the expected maximum utility (the difference in I before and after a change) may then be monetized by dividing by the marginal utility of income which is given by the travel cost parameter α.[21] Specifically, average per person per trip WTP in year t resulting from changes in water quality may be calculated

$$Average\ per\ trip\ WTP_t = \frac{1}{N_t}\sum_{i=1}^{N_t}\left(\frac{I_{it}^{*} - I_{it}^{0}}{\alpha}\right), \tag{41}$$

where N_t is the total sample size in year t. Multiplying the average per person per trip WTP by the number total number of trips taken in a season yields an estimate of aggregate annual WTP for a given change in utility per trip.

Many previous analyses that have used MRIP data have commonly used nested models that investigate fishermen's choice of fishing mode and then fishing site. For examples see Strand *et al*. (1991), McConnell and Strand (1994), Hicks *et al*. (1999), and Haab *et al*. (2003). The previously mentioned studies that utilized nested models generally focused on much larger and diverse study areas that allowed for more variations across sites which likely aided estimation. In those larger area models the Chesapeake Bay was often characterized by county level access sites that would not capture the variability in water quality and catch rates that this study attempts to investigate. We instead try to allowing for fishing mode effects in our second stage OLS regressions by utilizing mode specific expected catch rates.[22]

4.1.2 Angler Participation Model

Ideally, the available recreational fishing trip data would include all the trips taken by a respondent during a given year or time period. This type of data would allow the specification of a model that looks first at the decision of whether to fish, and then conditional on fishing, where to go in the same model.[23] Instead, the available data consists of a relatively

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[21] The complete expected maximum utility expression contains a known constant term (Euler constant) that has no effect on utility differences and is therefore dropped from the expression in Equation (40). For more detail on the constant see Haab and McConnell (2003 p 199).

[22] We considered several more general or flexible versions of the RUM such as the nested logit or mixed logit that would allow for richer substitution patterns between sites and possibly modes of fishing but convergence problems restricted the analysis to use a more basic model.

[23] Given data on fishermen's participation decision (whether or not to go fishing) as well as where to fishing once they have decided to go, it would be possible to estimate a nested logit or Kuhn-Tucker (K-T) model. In

large sample of one-time observations of fisherman who have already decided to take a trip in a given year. Given the available data we attempt to estimate changes in participation on a seasonal level. Specifically, we estimate the percentage change in the number of trips taken over the course of a season in response to a change in water quality. To do this we utilize counts of the number of trips taken over a given period of time as the dependent variable of a Negative Binomial (NB) fishermen participation model.[24] We estimate a NB model to allow for the possibility of overdispersion within the data (i.e. the variance of the data is greater than the mean). If there is no overdispersion then the model will collapse to a standard Poisson model.

The literature provides several possible functional forms for specifying linkage between the site selection and trip demand model, ψ_{it}.[25] The most direct linkage is provided by Hausman *et al*. (1995) who monetize the negative of the maximum expected utility term by dividing it by the RUM TC travel cost parameter:

$$\psi_{it} = -\frac{I_{it}}{\alpha}. \tag{42}$$

The monetized expected value term is a measure of the attractiveness of the sites. This linkage allows the attractiveness of sites to enter the participation decision. See Herriges *et al*. (1999), Bockstael and McConnell (2007), and Parsons *et al*. (2009) for summary discussions of the linked RUM and NB model.[26]

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the nested model, the first stage of fishermen decision could be whether to go fishing or do something else. Assuming the fisherman chooses to go fishing the second stage then models where the fishermen will choose to fish. In the K-T model anglers are assumed to determine the portion of their budget they will allocate to fishing trips and then decided where to take those trips. Both types of models would allow the estimation of utility changes per trip and from increased trips in a utility theoretic model. The RUM model estimated earlier in this analysis estimates only the second stage of the described nested model.

[24] Two potential trip counts are used in the analysis in order cover the full range of potential angler participation. The first trip count come from fishermen's reported annual trip number of trips and the second trip count measure is based on counts of the number of anglers intercepted from each zip code. Both counts are described in the Data section. The model described in this section is specified in terms of individual observations *i* but may be applied to the zip code level count data by thinking of the *i*'s in terms of unique zip codes rather than individuals.

[25] See Parsons *et al*. (1999) for a discussion and comparison of most prevalent linking methods.

[26] We also estimated the separate price and quality linking indexes suggested by Parsons and Kealy (1995) and Feather *et al*. (1995). The linking variables suggested by the authors were not adopted in this analysis because the relative sizes of the price and quality indexes in both cases led to implausible welfare estimates.

The number of fishing trips observed is a positive count y_i where *i* denotes individual *i* of *N*. Using a negative binomial regression the probability of observing y_{it} trips from individual *i* in time *t* is then

$$P(y_{it}) = \frac{\Gamma(\theta + y_{it})}{\Gamma(1 + y_{it})\Gamma(\theta)} r_{it}^{y_{it}} (1 - r_{it})^{\theta} , \tag{43}$$

where $r_{it} = \lambda_{it}/(\lambda_{it} + \theta)$, λ_{it} represents the mean number of expected trips from individual *i* in period *t*, and Γ represents the gamma function. λ_{it} is assumed to be a function of participant and site characteristics and may be specified

$$\lambda_{it} = e^{\omega + G_{it}\theta + \psi_{it}\eta} , \tag{44}$$

where G_{it} is a vector of zip code specific characteristics and ψ_{it} is the monetized RUM expected maximum utility term (equation (40)) that links the RUM and NB models together. ω, θ, and η are parameters to be estimated. Each observation is weighted by W_{jt}/ρ_{jt}, which is the inverse of the weight used in the RUM model. The inverse weight effectively puts a heavier weight on the more lightly sampled sites and should correct for any over or under sampling of sites in the individual observations.

4.1.3 Total Population WTP

Following Haab and McConnell (2003), to calculate the average annual willingness to pay, $\overline{WTP}$, the RUM per trip WTP expression given in equation (41) is multiplied by the predicted number of trips taken to the Chesapeake Bay under policy conditions:

$$\overline{WTP} = TRIPS_t \left(\frac{\lambda_{it}^{*}}{\lambda_{it} H_t} \right) \sum_{i=1}^{N_t} \frac{I_{it}^{*} - I_{it}^{0}}{\alpha} . \tag{45}$$

The total number of trips in a year, $TRIPS_t$,comes from NOAA estimates of the annual number of inland fishing trips taken in Maryland and Virginia.[27] To calculate the total number of trips taken under the TMDL conditions, the Chesapeake Bay trip total is adjusted by the predicted change in the number of trips from the NB model. The percentage change in predicted

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[27] Angler trip data was retrieved from http://www.st.nmfs.noaa.gov/recreational-fisheries/data-and-documentation/queries/index

trips is given by the percentage change in λ_{it} calculated under baseline and policy conditions.[28]

4.1.4 Angler Trip Data

Data on trips by anglers to the Chesapeake Bay and nearby salt water fishing sites come from NOAA's MRIP survey.[29] The intercept survey component of the MRIP focuses on collecting marine angler catch and effort data on a national scale. The information collected is used by fisheries managers to set management regimes and influence the health of marine fish stocks. While recreation demand modeling is not the primary purpose the survey's data collection a number of researchers have successfully used the intercept data to estimate fishermen's WTP for changes in site access and site quality. For examples see Strand *et al*. (1991), McConnell and Strand (1994), Hicks et al. (1999), Haab *et al*. (2000), Gentner (2003), and Lipton and Hicks (2003).

For recreational fishing site choice purposes the Chesapeake Bay region is assumed to contain all sites in the Chesapeake Bay, the Atlantic Ocean coastal sites on the Delmarva Peninsula, and all sites in the Delaware Bay. In this assumed Chesapeake Bay region, there are 670 access sites defined by NOAA that are eligible for sampling in each year. Not every site is visited each year. Interviewers are distributed across a subset of the sites each year in a targeted way based on the expected level fishing effort across the sites. The expected levels of effort are defined by site register pressure codes that are updated every two months throughout the year.

Previous researchers have used county level aggregated site definitions; however the relatively large county level aggregations were found to mask a great deal of the water quality variability found in the Bay. We instead grouped geographically similar sites into forty-

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[28] The NB model's predicted change in the number of trips can also be used to directly calculate per person per season WTP (see Parsons *et al*. 2009 for details). The estimated per person seasonal value was found to be very sensitive to the assumed measure of trip counts. In this analysis where we are forced to rely on potentially over and understated trip count measures it was deemed more appropriate to focus on the predicted percentage changes rather than per person seasonal welfare estimates.

[29] The reauthorization of the Magnuson-Steven Fishery Conservation and Management Act of 2007 established the Marine Recreational Information Program to replace the Marine Recreational Fishing Statistic Survey (MRFSS). The MRIP program is intended to address a number of the known weaknesses of the MRFSS.

one aggregated sites. As shown in Figure 2, thirty-four of the aggregated sites are located in the Chesapeake Bay and remaining seven are along the Atlantic Coast and Delaware Bay.

Because this analysis focuses only on day trips, angler's one-way travel distance from home to the closest site is limited to be no more than 150 miles.[30] This analysis focuses on trips taken between 2008 and 2010 although previous year's data are used in some specifications of expected catch. Table 19 shows the number of anglers intercepted at Chesapeake Bay and nearby substitute sites (i.e. the Alantic and Delaware Bay sites) who live within 150 miles of at least one access point in the Chesapeake Bay in each year, as well as the number of intercepted anglers at each aggregated site over the study years.

Travel costs are computed from each angler's home zip code centroid to the central point of each aggregated access site.[31] The monetary component of travel cost is computed as the round trip travel distance times the annual AAA midsize sedan cost per mile. The opportunity cost component of travel costs is computed as the round trip travel time in hours multiplied by one third of each person's hourly income which is assumed to be equal to the zip code median personal hourly income. Median hourly income per year was calculated by dividing the median annual household income by average number of people per household from the US Census (2.59 people) and then by the average number of hours worked per year in the United States.[32] Total travel cost is then the sum of monetary and opportunity costs. If an angler reported fishing on a headboat then a $50 average headboat trip fee is added to their travel costs. If an angler reports fishing on a charter boat then a $100 average charter boat fee is added to their travel costs. Table 20 shows the average estimated per-person travel costs in each year.

When estimating changes in the number of fishing trips taken due to the TMDL changes in water quality we utilize two different trip counts that should capture the full range of potential angler trip participation. The first count is based on the responses of anglers intercepted at Chesapeake Bay fishing site to the question "Not counting today, within the past 12 months, that is since (insert month) of last year, how many days have you gone

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[30] Expanding the one-way travel distance to 250 miles only adds roughly 2000 more observations over five years or less than roughly 3.5% of the total sample size.

[31] Road network distance and travel times were calculated using MPMileage software.

[32] From the Organization for Economic Cooperation and Development (https://stats.oecd.org/Index.aspx?DataSetCode=ANHRS)

saltwater fishing in this state or from a boat launched in this state?" as a measure of the number of trips taken by each angler to the Chesapeake Bay each year. The first potential problem with using responses to this day's fished question is that the responses may contain trips to saltwater destinations outside of the Chesapeake Bay. Interviewers are more likely to intercept avid users than casual users since the avid users will be on site more often over the course of a season. Both issues could lead to an overstatement of the number of trips taken. As summarized in Table 21, the data seems to support this assumption as a large number of fishermen intercepted between 2008 and 2010 report fishing 300 or more days per year resulting in an annual average number of trips of almost twenty trips.

As a counter to the potentially inflated fishermen reported annual days fished totals, we also utilize a conservative trip count that uses the number of fishermen intercepted from a given zip code over the year across all Chesapeake Bay fishing sites as a proxy for the number of anglers trips taken from a given zip code taken in a year. We include all zip codes within 150 miles one-way travel distance of a Chesapeake Bay site even if no trips were observed from the zip code. We identified 2435 residential zip codes with the 150 mile range and roughly half the zip codes did not have any intercepted trips. An obvious problem with this zip code intercept method though is that while these zip code counts may provide a reasonable estimate of the number of fishermen participating from a given zip code, it cannot account for differences in the average number of trips per year taken by fishermen from different zip codes. Fishermen in the zip codes near the Bay likely take more trips per year than fishermen living farther from the Bay which means the measure likely underestimates the true level of participation in some zip codes. For comparison purposes, using NOAA Mid Atlantic summary statistics we calculated the average number of trips per year between 2008 and 2010 for Mid Atlantic anglers.[33] The Mid Atlantic average numbers of trips were in the six to seven trips per year range which is significantly less than the "days fished" count but greater than the zip code intercept count. While neither the days fished nor the intercepted zip code trip count measures are perfect, the NOAA Mid Atlantic data does seem to suggest that they adequately cover the range of potential trips.

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[33] Estimates of total Mid Atlantic anglers and trips were taken from the 'Fisheries of the United States 2012, Mid-Atlantic Region' report found at www.st.nmfs.noaa.gov/economics/publications/feus/fisheries_ economics_2012.

4.1.5 Expected Catch Data

One of the key variable linking water quality to fishermen's WTP is expected catch. The appropriate way to define expected catch as fishermen think about it is not always obvious however. Some fishermen may care only about catching a specific species of fish while others may be happy to simply catch anything. In addition to the question of how to define expected catch, the amount of information considered by fishermen in forming their catch expectations is not immediately evident either. Do fishermen consider mainly current information, mainly historic information, or some combination of the two? This analysis addresses these questions by testing a number of specifications differing in the way in which species are grouped, the mode of fishing considered, and the amount of current and past information that is used in forming catch expectations.

Expected catch estimates for each site in each year are constructed using the MRIP I2 and I3 catch records from the intercepted trips used in the RUM analysis. I2 records contain each intercepted angler's reported catch that was not available for the interviewer to inspect, while the I3 records denote angler catch that the interviewer was able to inspect. If a fisherman fished alone, or no one else contributed to the reported catch then the fisherman's I2 and I3 records are summed to get the that fisherman's total catch. If more than one fishermen contributed to the reported and observed catch then summed I2 and I3 records are divided by the number of fishermen who contributed to the catch in order to get catch in a per person form. The per fishermen per trip catch rates are then divided by the number of hours fished on the trip so that catch is finally expressed in fish caught per fishermen per hour. The main difference between the expected catch specifications is in what type of fish are included in the fish totals or fish categories. See Table 22.

The first and most basic measurement of a fisherman catch considered is a count of the total number of fish caught regardless of the species. For many fishermen a successful trip may be determined simply by the number of fish caught. Expectations regarding catch may also differ across different fishing methods so we also consider a model that splits total catch into catch from those fishing from a boat and catch by those fishing from shore.

A more realistic specification may be one that considers both the total number of fish caught and the type of each fish caught. In order to account for fishermen who target specific species of fish we also test versions of the first two models that also include the catch totals of individual targeted recreational species. The major recreational species are Atlantic Croaker, Spotted Seatrout, Black Sea Bass, Striped Bass, Spot, Bluefish, Summer Flounder, White Perch, and Tautog. These species comprise roughly ninety percent of the total recreational catch. Table 5 gives an idea of the relative popularity of different species among fishermen targeting a particular species. It is interesting to note that Striped Bass and Summer Flounder comprise roughly half of all targeted trips between 2008 and 2010. See Table 23.

The individual species catch rates are comprised of catch from trips where the angler stated they were targeting that particular species. Individual species catch rates for each site are weighted by the percentage of total trips taken targeting a given species that originate from a site. This weighting is intended to put more weight on catch rates at sites where fishermen take a lot of targeted trips and to put less weight on the catch at sites where fishermen take fewer targeted trips.

Two other catch rate measures based on different aggregate groupings of fish species are also considered. The first of these groupings considered is adapted from McConnell and Strand (1994) and Hicks *et al*. (1999). In those works the major mid-Atlantic recreational species are grouped into small game fish, bottom fish, and flat fish.[34] The work of McConnell and Strand focuses on the Mid and South Atlantic region while Hicks *et al*. focus on the Mid Atlantic and New England region. Both areas are much larger and contain a significantly more diverse habitat and larger numbers of species than the Chesapeake Bay.

To address the Bay's uniqueness a Chesapeake Bay specific fish grouping was also created. The Bay specific catch rate groupings first group fish based on whether they are most commonly caught in the main Bay or one of its brackish tributaries. If the fish are most commonly caught in the main Bay they are then divided into predominately bottom fish and other game fish. Attempts were made to group species according to the simple metric of "if you are fishing in a certain part of the Bay (main stem vs. tributary) using a certain method of fishing (bottom fishing vs. other) what species would likely expect to catch?" Significant judgement went into defining the catch rate categories because there were very few clear

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[34] The authors also included a Big Game and Other fish group category however the almost none of the species in those groups are found in the Bay itself so they were excluded from this analysis.

dividing lines along which to group fish. For example, Striped Bass can be caught in both the main Bay and its tributaries. Striped Bass can also be caught bottom fishing, trolling, or casting.

Even assuming that the researcher knows the appropriate definition of expected catch that fishermen care about there is still the question of what information fishermen use to form their catch expectations. Do they rely on current year catch reports, the previous year's reports, or some combination of current and historic information? In this application we calculate the average annual catch per hour per fisherman for each site for each expected catch specification for (1) the current year, (2) the previous year, and (3) the combined current and two previous years. The historic average annual hourly catch per angler across all sites in the bay for all expected catch measures is presented in Table 24. The average trip length in the data was a little over four hours which translates into most fishermen catching between two and four fish per trip on average.

While the diverse set of expected catch specification and combinations of information used to calculate expected catch should help address two large expected catch specification questions, there is still be a great deal of uncertainty and potential measurement error in the expected catch estimates. First, the observed and reported catch collected during the intercept surveys used to construct catch rate estimates is observed only at the fishermen's intercept location and in most cases not where they were actually fishing (except for shore fishermen). While the Bay is a large waterbody, it is not uncommon for fishermen to leave the dock on one side of the Bay and fish on the other side. They may also travel several miles either north or south from the location their trip originated. This analysis relies on the assumption that catch data collected at the intercept sites should be highly correlated with the nearby expected catch. It is possible however that fishermen leaving from multiple sites around the Bay may end up actually fishing in the same general area which would mean that we might not see significant difference in expected catch between many sites.

This analysis also makes several other simplifying assumptions. For example, it is assumed that fishermen value take home catch and catch-and-release catch in the same way. It may be the case that different fishermen have different preferences for keeping their catch. Similar to the previous point, this analysis does not consider the effects of catch regulations on fishermen. It is assumed that no fisheries are closed and fishermen value fish they are forced to release the same as fish they are allowed to catch. This analysis also does not account for how the size of fish caught influences angler WTP. An angler may be much happier catching one large trophy fish than catching dozens of smaller fish. Finally, data limitation force the analysis to rely on annual averages that will not be able capture within season catch fluctuations.

4.1.6 Water Quality Data

The water quality at a site may also have effects on fishermen's WTP beyond its influence on expected catch. For example, fishermen may avoid sites with poor water quality for aesthetic reasons. Several measures of water quality are estimated by the Chesapeake Bay Estuary Model (CBEM) including the concentrations of dissolved oxygen (DO) and chlorophyll-a (CL), and the level of water clarity K_d (KD). DO is necessary for survival of aquatic species so we would expect the effect of increased DO to always be positive although at higher levels it may have a decreasing effect. CL is an indirect measure of the biomass of algae in the water which may be beneficial at low levels for supporting life but also potential harmful at higher level if it leads to over-enrichment and discolored cloudy water. KD is a measure of clarity or light attenuation expressed in inverse meters in which a lower value corresponds to increased clarity. Annual water quality estimates for each water quality measure were constructed using the distance-weighted average of interpolated values from the three closest cells in the Chesapeake Bay Estuary Model (CBEM) in a given year.

As shown in Table 25, there is significant variation in water quality values across the sites in the Chesapeake Bay, but the three water quality measures are found to be highly correlated. CL is strongly positively correlated with both KD and DO, while KD and DO have a significant negative correlation. The high levels of multi-collinearity of present in the data mean that including all three water quality measures will likely introduce large errors in prediction and WTP calculations. Given the need to narrow the number of water quality parameters included in the second stage regressions, in models with catch rates included it makes the most sense to include KD and exclude CL and DO. CL and DO can both be thought of roughly as proxies for aquatic health which should be adequately represented by the sites expected catch rate. Water clarity is also the most directly observable water quality measure which should make it easier for fishermen to identify differences across sites.[35]

4.1.7 Site Choice Model Results

The RUM site choice model coefficient estimates for years 2008 through 2010 are presented in Table 26. Coefficient standard errors were estimated by bootstrapping in which the model was re-estimated 200 times using sample datasets of the same size as the original data set drawn with replacement. As expected, the Travel Cost parameter (TC) is negative and highly significant and relatively stable across years.[36] The result confirms that all else equal fishermen would rather visit a closer site than a more distant one all else equal. The majority of the alternative specific constants (ASC's) are also significant signaling that the characteristics of each site are important to fishermen when choosing where to go fishing. Some ASC's do vary in magnitude year to year. The coefficient on aggregate fishing site 41 was set equal to zero in estimation.

The estimated ASC's from all three years are then stacked and used as the dependent variable a second stage ordinary least squares regression to recover fishermen's expected catch and water quality preference coefficients. All models also include 2008 and 2009 year fixed effects as well as the NSITES variable that measures the number of MRIP intercept sites included in each aggregated site. The results of the second stage regressions using different expected catch metrics are presented in Table 27, Table 28, and Table 29.

Table 27 presents the results arising from measuring expected catch in its most basic manner as the total number of fish caught. Model 2 differs from Model 1 in that it divides total catch into catch caught from boats and catch caught from shore. In Model 1, the TOTAL CATCH is positive, significant, and relatively stable across the expectation time frames considered signaling that fishermen do tend to visit sites with higher expected catch more often. In Model 2, the BOAT CATCH variable is also positive and significant and somewhat similar in magnitude to the Model 1 TOTAL CATCH coefficient. Across specifications the SHORE

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[35] Given the uncertainty in catch rates noted earlier, we also estimate several models that exclude catch rates and attempt to rely on the water quality measures to proxy for species abundance. The water quality only models focus on KD and DO since they had the lowest level of correlation among the water quality measures. In all models and all specifications including DO it return coefficient estimates with the sign opposite of the expected.
[36] To aid estimation, travel costs were scaled by dividing them by one hundred.

CATCH variable has an unexpected negative sign and is insignificant in all but the three-year average model. The insignificance of the SHORE CATCH variable may suggest that estimating the RUM model in a fashion that nests choices by mode may not be necessary.[37] In both Models 1 and 2 the KD variable is negative and significant across all specification signaling that fishermen prefer clearer water all else equal. Similarly, fishermen are found to prefer sites with more water access points all else equal.

Models 3 and 4 extend the first two models by also including targeted individual species specific catch rates. In contrast to Model 1, the TOTAL CATCH variable in Model 3 is significant only in the current year average specification. In Model 4 the BOAT CATCH measure is again positive and significant while the SHORE CATCH is unexpectedly negative across all expectation time frames. The KD variable is again negative, significant, and relatively stable as expected across all specification of Models 3 and 4. Individual species catch rate coefficient estimates vary greatly and in some cases in unexpected ways. All species except striped bass and white perch also have positive coefficient estimates, although only a few species have significant catch rate coefficients. Croaker is significant across all catch rate measures while sea bass, sea trout, and flounder are significant under a few measures. Despite being a popular recreational species, white perch is found to have a significant negative catch coefficient across all Model 3 and 4 specifications. Even with the weighted species specific catch rates the coefficient estimates appear to be very sensitive to the presence of several very large reported catch totals at infrequently visited sites.

In the Models 5 and 6 fish group expected catch specifications, the Mid Atlantic grouping and the Bay specific grouping perform similarly poorly. Very few coefficients are significant in either model although the KD and NSITES variables again return correctly signed and significant coefficients across all models and specification. The FLATFISH variable in Model 5 is the only positive and significant catch coefficent across all specification. Model 6 returns no significant catch coefficients.

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[37] Several reviewers suggested more flexible nested or mixed logit models but estimation difficulties limited the analysis to a standard flat logit.

Overall, Models 1 through 4 seem to be preferable to Models 5 and 6 because of their anticipated positive and significant coefficients on TOTAL CATCH and BOAT CATCH. Compared to Models 1 and 2, Models 3 and 4 do provide a lot of additional explanatory power although their species specific catch rate results are not overwhelmingly significant and often do not fit a priori expectations. There also does not appear to be much significant difference in results across the different expected catch time frames. Since no model clearly dominates all others we calculate WTP for models 1-4 in order to demonstrate the potential range of predicted WTP.

4.1.8 Per Trip Willingness to Pay for the TMDL Catch and Water Quality Changes

The results of the first stage RUM and the second stage regressions are then used to calculate per trip WTP for changes in water quality and catch rates. We rely on two estimates of the changes in catch rates in the Chesapeake Bay due the implementation of the TMDL. The first comes from the panel of Chesapeake Bay fisheries experts described in section 2.1. The second set of predictions comes from the Multi-Species Fisheries Model (MSFM) described in section 2.3. The MSFM also produces estimates for changes in catch outside of the Bay. Both the expert panel and the MSFM predict very small changes in catch rates for most species. Within the Bay, the expert panel's predictions translate into an average increase of .08 fish per trip, while the MSFM's predictions translate into an average increase of 0.04 fish per trip. Using the baseline and with TMDL predictions of water quality at each of aggregated Chesapeake Bay intercept sites estimated by the CBEM, we calculate the percentage change in KD due the TMDL at each site. The average change in KD across all sites is 11%. A summary of all predicted changes is provided in Table 30.

Mean per person per trip WTP estimates and 95% confidence intervals for each RUM and second stage regression combination are presented in Table 31 and Table 32. Table 31 presents angler per trip WTP for the changes in catch rates in the Chesapeake Bay predicted by the expert panel and the MSFM. Within the Bay, the expert panel and MSFM predictions result in very similar per trip WTP predictions. Per trip WTP ranges from a minimum average of a little more than a dollar up to nearly three dollars per trip with an overall average across all models, specifications, and catch predictions of $1.91. The similarity of the predicted WTP value is not surprising given that the expected catch predictions are similar in magnitude.

The majority of the per trip WTP for the TMDL changes comes from changes in KD across the sites. Between $.10 and $.20 of each per trip WTP estimate is attributable to the change in catch rates.[38] The limited effect of small changes in catch rates on WTP is demonstrated clearly with the MSFM's outside of the Bay changes in catch rates. Outside of the Bay there are assumed to be no changes in water quality and any changes in catch rates are due to fish migrating in and out of the Bay. As shown in Table 32, The model predicts a change in total expected catch of 0.6% which translates into an increase of slightly less than .02 fish per trip. The average increase in WTP for fishing trips outside of the Chesapeake Bay across all models and specifications is slightly less than $.06 per trip.

4.1.9 Changes in Angler Participation due to the Changes in TMDL Water Quality

The results of the 2008 through 2010 linked NB models are presented in Table 33. The results show that across years both trip count models perform as anticipated and that their results are highly significant. In all cases, increased income leads to increases in the number of trips demanded. In the zip code intercept count model, POPULATION is also found to increase the number of trips. In both models and in all years, the RUM IV index is positive and highly significant signaling that as the attractiveness or expected utility of the Bay increases the number of trips demanded will increase as well.

Using the results of the linked model it is also possible to calculate the expected change in trips per person. Despite the potentially large differences in the absolute magnitudes of the "Days fished in the last 12 months" and the "zip code intercepts" counts they both results in very similar predictions of the expected percentage change in the number of trips. On average, the two model's predictions differed by only roughly 1.5%. The average percentage increase in trips across all specifications of the "Days fished in the last 12 months" count model was 2.26% when using the expert panel catch predictions and 2.02% when using the MSFM catch predictions. The average percentage increase in trips across all

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[38] In situations such as this with very small changes in expected catch, a "reduced form" with only water quality measures and no catch rate measures will produce almost the same WTP estimates.

specifications of the “zip code intercepts” count model was 3.84% when using the expert panel catch predictions and 3.59% when using the MSFM catch predictions. The high value of 3.84% and the low value of 2.02% are used as bounds later when calculating total population WTP values. Focusing on trips taken outside of the Chesapeake Bay the “Days fished in the last 12 months” model predicts a 0.9% average percent change in the number of trips taken, while the “zip code intercept” model predicts a 0.13% change in trip taken. These results are shown in Table 34 and Table 35.

4.1.10 Annual Population Welfare Results

In the final step, the average per person per trip WTP estimates for each potential catch measure for each year for trips inside and outside the Chesapeake Bay are multiplied by the anticipated number of trips per year under the TMDL conditions inside and outside the Bay to calculate total WTP for Bay and non-Bay fishermen. The Bay and non-bay fishermen’s total WTP is then added together to get total population WTP for the water quality and catch rate changes resulting from the Chesapeake Bay TMDL.

Trip total estimates come from NOAA’s Recreational Fishing Statistics. [39] Annual angler trips to the Chesapeake Bay are assumed to be equal to the sum of Maryland and Virginia inland trips. As shown in Table 39, annual trip totals range from 5.1 to 6.1 million trips between 2008 and 2010. The number of outside of the Bay trips taken was calculated by summing NOAA’s state level trip estimates from the states stretching from Massachusetts to North Carolina for each year from 2008 through 2010 . [40] These states were chosen because they cover the range of the major species expected to migrate in and out of the Bay whose population levels and catch rates may be affected by the TMDL. The estimates of the number of Chesapeake Bay trips is then subtracted from the sum of the state trips to give an estimate of the total number of trips taken outside of the Bay.

To determine how the number of trips will change both in and out of the Chesapeake Bay in response to TMDL changes we use the low and high estimated average percentage

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[39] Angler trip total were downloaded from http://www.st.nmfs.noaa.gov/recreational-fisheries/data-and-documentation/queries/index

[40] State level trip total were downloaded from http://www.st.nmfs.noaa.gov/st1/recreational/queries/

change in participation across NB count models (Table 34 and Table 35) to bound the potential responses. In the Bay, the lowest average response was a 2.02% increase in trips while the highest average response was w 3.84% increase. For trips outside of the Bay the low percentage change was 0.09% and high percentage change was 0.13%. The estimated number of new trips is then added to the estimated number of existing baseline trips to get total trips under the TMDL conditions.

The estimated per person per trip WTP values estimated from the RUM model for fishing trips inside and outside of the Bay are then multiplied by the appropriate number of trips expected under the TMDL to calculate total population WTP. As shown in Table 37, WTP for existing trips is found to range from roughly half a million to $22.8 million per year with an average across all specifications of $10.4 million. The WTP for trips outside the Bay was much smaller due to the very small change in expected catch due to the TMDL. Outside the Bay WTP ranged from roughly $450 thousand to $5 million per year with an average across specifications of $1.8 million. On average around 85% of the total WTP from existing trips comes from trips taken in the Chesapeake Bay.

To give an idea how total population WTP breaks down between new and existing trips, WTP for new trips is presented in Table 38. WTP is calculated for using both the estimated "Days Fished" and Zip Code" trip count participation changes. As stated earlier, the two measures provide a reasonable upper and lower bound for participation changes and the WTP for new trips due the TMDL improvements. . WTP for new trips in the Bay due to changes from the TMDL range from $11 thousand to roughly $650 thousand per year with an average of $310 thousand. The average WTP for new trip outside of the Bay is roughly $1.5 thousand. Values outside of the Bay are considerably smaller because the expected changes in catch rates are much smaller and there are no assumed improvements in water quality.

Finally, new and existing trip from both within and outside the Bay are summed together to estimate the total population WTP for the water quality and fish population changes driven by the Chesapeake Bay TMDL. The combined WTP for existing and new trips under TMDL conditions range from $1.1 million to $28.7 million per year with an average across all specifications of $12.6 million. The central estimate of WTP for the TMDL driven changes in catch rates and water quality across all models is near $22.9 million per year. The model mean estimates range from \$4.9 to \$58.8 million per year. Full results for all model specifications may be seen in Table 39.

4.2 Other outdoor recreation activities

In addition to recreational fishing, other water-based outdoor recreational activities also might be affected by changes in water quality, including swimming, boating, and possibly hiking, camping, picnicking or other activities that take place in the water or near the shoreline. This section describes a participation and site-choice model for visits to outdoor recreation sites with water access to the Chesapeake Bay. We estimated the model using census data on the number and location of households (at the zip code level) that live within an approximately three-hour driving distance from Chesapeake Bay, plus aggregate site visitation data, which comprise records of the total number of visitors per year to a collection of outdoor recreation areas in Maryland, Virginia, and Delaware between 2001-2011. These aggregate data contain no information on the types of activities pursued by the visitors on each trip, so it is not possible to estimate activity-specific preference parameters using these data. Therefore, the preferences for water quality improvements estimated using these data pertain to a generic undifferentiated class of water-based recreation activities. We used supplemental data from two stated preference surveys to help estimate the model parameters and to account for (some of) the heterogeneity in preferences for outdoor recreation activities among the target population: a survey on Atlantic menhaden fishing regulations by Kirkley *et al*. (2011), and a new stated preference survey designed and implemented as a companion to the present study to estimate the total economic value of the Chesapeake bay TMDL (Moore *et al*. 2015). We also used these supplemental survey data to estimate the share of outdoor recreation trips devoted to fishing. This allowed us to separately identify the contribution from recreational anglers to the final benefit estimates, thereby helping to avoid double-counting the recreational fishing benefits described in section 4.1.

The starting point for the analysis was a repeated choice multinomial logit recreation demand model (Morey *et al*. 1993). Indirect utility for individual i visiting site i on choice occasion t is represented by

$$V'_{ijt} = Z_i\delta + X_j\beta + \varphi_j - \lambda c_{ij} + \varepsilon_{ijt}\ , \tag{46}$$

where Z_i is a vector of personal or household characteristics for the individual, X_j is a vector of observed site-specific attributes, δ and β are vectors of preference parameters associated with individual- and site-specific attributes, respectively, φ_j is the indirect utility from all unobserved fixed site-specific attributes, λ is the travel cost parameter (the marginal utility of income), c_{ij} is the cost for individual i to visit site j, and ε_{ijt} is an "error" term that is individual-, site-, and occasion-specific, assumed to be observed by the individual but not by the researcher. Assuming that the ε_{ijt}'s are identically and independently distributed across individuals, sites, and choice occasions and follow a type II extreme value distribution, the probabilities that individual i visits site j or stays home on choice occasion t are

$$p_{ijt} = \frac{e^{V_{ij}}}{1 + \sum_k e^{V_{ik}}}, \tag{47}$$

and

$$p_{i0t} = \frac{1}{1 + \sum_k e^{V_{ik}}}, \tag{48}$$

respectively, where $V_{ij} = Z_i\delta + X_j\beta + \varphi_j - \lambda c_{ij}$ is the "observable" component of indirect utility (McFadden 1974).

The model was estimated in two stages, following the approach of Murdock (2006). In the first stage, maximum likelihood and a contraction mapping algorithm were used to estimate a full set of alternative-specific constants (ASCs) for each site, $\theta_j \equiv X_j\beta + \varphi_j$, the travel cost parameter, λ, and the individual attribute coefficients, δ. The ASC for site j, θ_j, is a lumped parameter including all fixed attributes of the site. The model must be estimated in two stages because it is not possible to estimate β and a full set of φ_j's for all sites in a single stage (Murdock 2006).

The contraction mapping component of the first stage of estimation exploits the fact that in a standard (conditional) multinomial logit model the predicted number of trips to each site by all recreators combined using the maximum likelihood parameter estimates, $\hat{Y}_j$, will equal the actual number of trips to each site in the data, Y_j. The contraction mapping algorithm finds the set of $\hat{\theta}_j$'s, conditional on δ and λ that make $\hat{Y}_j = Y_j$ for all sites. The contraction mapping procedure is very fast and highly stable, so this approach effectively reduces the first stage of estimation to the much more parsimonious problem of maximum likelihood estimation of δ and λ .

In the second stage of estimation, we used ordinary least squares to regress the ASCs estimated in the first stage on a set of explanatory variables describing the site attributes. The explanatory variables included measures of water quality conditions in the nearby areas of Chesapeake Bay and dummy variables indicating the presence of amenities at the sites that support various recreation activities, such as trails for hiking, water for swimming, access for boating, etc. The two stages of estimation are described in more detail in sections 4.2.1 and 4.2.2. The data used for estimation are described in section 4.2.3, the benefit calculations are described in section 4.2.4, and the estimation results and associated benefit estimates are presented and discussed in section 4.2.5.

4.2.1 Stage 1

The available aggregate data on total trips to each site alone cannot support estimation of all first-stage parameters of the participation and site choice model in (47) and (48)—θ, λ, and δ—so additional information is required to complete the model. Intuitively, the travel cost parameter, λ, cannot be identified using the aggregate visitation data alone since these data indicate how many people visited each site but they contain no information indicating how far the visitors traveled to reach the sites, and estimation of the other stage 1 parameters requires the identification of λ.

To estimate all stage 1 parameters simultaneously, we supplemented the aggregate site visitation data with data from two stated preference surveys. The first was a survey by Kirkley *et al*. (2011), which was designed to examine people's preferences for restrictions on the commercial harvest of Atlantic menhaden in the Chesapeake Bay. Among the questions included in the survey was the following: "About how many trips did you make from your home to the bay for the main purpose of outdoor recreation during the past 12 months?" The survey also asked each respondent the zip code of their residence plus several demographic questions, including their age, gender, level of education, number of people in their household, and level of income. The second stated preference survey was designed and implemented for the present study to estimate people's total willingness to pay for improvements in four key ecological endpoints in the Chesapeake Bay watershed (Moore *et al*. 2015). Like the survey by Kirkley *et al*., the Chesapeake Bay SP survey also included supplemental questions about the respondent's outdoor recreation behavior within the last 12 months. We did not attempt to combine the data from the two stated preference surveys in this study. Instead, we estimated the model twice using data from each stated preference survey in turn, and we present and compare the results based on both datasets below.

The participation and site-choice decisions are linked in this model by the assumption that they both emerge from the random utility maximization process described above by equations (46)-(48). From the assumption that the ε_{ijt}'s—the error terms in equation (46)—are uncorrelated across choice occasions, it follows that the participation and site choice probabilities in equations (47) and (48) are constant across choice occasions. Therefore, the choice of whether or not to stay home or take a trip on each choice occasion is a binomial process, and so the probability that respondent i takes y_i trips in a year can be approximated by a Poisson distribution,

$$Pr[y_i] = \frac{E[y_i]^{y_i} e^{-E[y_i]}}{y_i!}, \tag{49}$$

where the expected number of trips, $E[y_i]$, is the product of the probability of visiting any Chesapeake Bay recreation site on a choice occasion and the number of choice occasions per year, D:[41]

$$E[y_i] = \frac{D \sum_j A_j e^{Z_i \delta + \theta_j - \lambda c_{ij}}}{1 + \sum_j e^{Z_i \delta + \theta_j - \lambda c_{ij}}}, \tag{50}$$

where A_j is an indicator variable equal to 1 if site j has water access to the Chesapeake Bay and 0 otherwise. Equations (49) and (50) can be combined to create the log likelihood function for the survey data, which can be written as:

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[41] We used $D = 365$ in this analysis. Experiments using $D = 180$ and $D = 90$ suggested that the choice of D affects the scale of the λ and β estimates by a common factor. Therefore, the choice of D does not significantly affect the final benefit estimates.

$$lnL \sim \sum_{i=1}^{I} \left[y_i ln \left(\frac{D \sum_j A_j e^{Z_i \hat{\delta} + \hat{\theta}_j(\hat{\delta},\hat{\lambda}) - \hat{\lambda} c_{ij}}}{1 + \sum_k e^{Z_i \delta + \hat{\theta}_k(\hat{\delta},\hat{\lambda}) - \hat{\lambda} c_{ik}}} \right) - \frac{D \sum_j A_j e^{Z_i \hat{\delta} + \hat{\theta}_j(\hat{\delta},\hat{\lambda}) - \hat{\lambda} c_{ij}}}{1 + \sum_k e^{Z_i \delta + \hat{\theta}_k(\hat{\delta},\hat{\lambda}) - \hat{\lambda} c_{ik}}} \right], \tag{51}$$

where $\hat{\theta}_j(\hat{\delta}, \hat{\lambda})$ is the contraction mapped value of the ASC for site j conditional on $\hat{\delta}$ and $\hat{\lambda}$. Heuristically, the estimation procedure involved the following steps:

0. Choose starting values for $\hat{\delta}$ and $\hat{\lambda}$.
1. Use contraction mapping to find the value of $\hat{\theta}$ that makes predicted site visits equal to observed site visits conditional on $\hat{\delta}$ and $\hat{\lambda}$.
2. Calculate the gradient of the log likelihood function in equation (51). If the gradient is zero, stop; otherwise go to step 3.
3. Increment $\hat{\delta}$ and $\hat{\lambda}$ using the gradient. Return to step 1.

In this approach, the survey data are used to estimate δ and λ (conditional on θ), and the aggregate site visitation data are used in the contraction mapping component of the procedure to estimate θ (conditional on δ and λ).

4.2.2 Stage 2

Stage 1 produced estimates of all θ_{jt}'s, the full set of alternative specific constants in each year. In stage 2 we recovered estimates of the site attribute coefficients, β, by regressing the estimated θ_{jt}'s on yearly water quality measures at the sites interacted with dummy variables that indicate whether the site has amenities to support at least one form of water-based recreational activity, either boating, fishing, or swimming.

Below we report results for several versions of two basic specifications. The first basic specification includes a linear time trend, γt, a full set of site-specific fixed effects, φ_j, and a vector of time varying water quality measures at each site, Q_{jt}, interacted with an indicator variable, w_j, equal to one if amenities to support water-based recreation activities (boating, fishing, or swimming) are present at the site and zero otherwise:

$$\theta_{jt} = \alpha + \gamma t + \varphi_j + w_j Q_{jt} \beta + \nu_{jt} \,. \tag{52}$$

Variations of this specification include versions with different combinations of water quality variables in Q_{jt}. The second basic specification excludes the time trend and site-specific fixed effects but includes a set of year fixed effects and site attribute dummy variables indicating whether the site has amenities to facilitate boating, fishing, swimming, hiking, picnicking, or camping activities, and whether the site has restrooms:

$$\theta_{jt} = \alpha_t + H_j\kappa + w_j Q_{jt}\beta + \nu_{jt}\ , \tag{53}$$

where H_j is a vector of [0,1] variables indicating whether or not site j has facilities to accommodate: boating, fishing, swimming, hiking, picnicking, camping, and restrooms.

The advantage of the first specification in (52) is that the site-specific dummy variables will capture all unobserved fixed attributes of the sites and so reduce the risk of omitted variable bias. However, a potential disadvantage is that the site-specific dummy variables also will capture any influence of the differences in the long-run average water quality conditions among the sites. In this specification the water quality coefficients are identified only by the within-site variation in water quality conditions across years. So this specification risks attributing part of the influence of water quality to the site-specific fixed effects rather than to the water quality variables themselves, which would ultimately bias our estimates of the benefits of water quality changes (presumably downward). In the extreme, if the only considerations of water quality by the recreators' involves comparisons of long-run average water quality conditions among the sites—as in, "I don't know what the precise water quality conditions will be today, but I know that on average site A tends to have better water quality than site B…"—then these influences would be completely absorbed in the site fixed effects and the water quality coefficient estimates would be close to zero even if the differences in long-run average water quality conditions among sites have a strong influence on the recreators' site choices.[42]

The disadvantage of the specification in equation (53) is that it runs a higher risk of omitted variable bias than does the specification in equation (52). If there are important site attributes beyond the short list of site amenities included in the estimating equation, and if one or more of the missing variables are correlated with the average water quality conditions at the sites, then the water quality coefficient estimates will be biased since they will capture

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[42] See Abbott and Klaiber (2011) for a more expansive discussion of spatial fixed effects in hedonic property value models, which can lead to biased estimates of the marginal effects of neighborhood amenities for reasons analogous to those discussed here.

a portion of the influence of these omitted variables. However, the advantage of the specification in (53) is that it eliminates the risk of attributing the influence of differences in long-run average water quality conditions among the sites to the site dummy variables, since no site-specific fixed effects are included. It also allows us to use both cross-sectional and temporal variation in water quality to identify the water quality coefficients. This is important because most of the variation in water quality near recreation sites on the Chesapeake Bay is spatial rather than temporal. (For example, the coefficient of variation of chlorophyll-a calculated for each site across years then averaged across the sites is 0.37, while the coefficient of variation calculated for each year across the sites then averaged across years is 1.67.)

We estimated several versions of both specifications using alternative sets of water quality variables and we calculated the benefits of the TMDL based on each specification. While not producing a definitive model, this approach clearly shows the sensitivity of our benefit estimates to the choice of model specification and functional form of the included water quality variables.

The three main water quality variables we used to specify Q_j for each site are chlorophyll-a concentration (CL), which is an indirect measure of the biomass of algae in the water column, dissolved oxygen concentration (DO), which is an important factor determining the suitability of an area for aquatic life, and Secchi depth (SD), which is the depth at which a disk with contrasting white and black quadrants can be seen from the surface of the water. Our prior expectations regarding the influence of these water quality measures on recreational trip demands and site choices were as follows. At low levels of CL, the marginal effect of CL may be negligible or positive, since initially increases in CL lead to more productive waters that can support more fish and therefore are more suitable for recreational fishing activities. However, eventually the marginal effect of CL may turn negative, since over-enriched waters become cloudy green and unsightly and therefore unsuitable for swimming and unattractive for other water based recreational activities. Persistently high CL levels can lead to frequent hypoxia, which is harmful to fish and shellfish. In contrast to the potentially non-monotonic effect of chlorophyll-a, we expected the marginal effect of DO to be always positive, though possibly diminishing since thresholds between, roughly, 2 to 5 μg/L are thought the be important for many aquatic species. We also expected the influence of water clarity to be always positive but possibly diminishing.

4.2.3 Data

To estimate the model we used data from five sources. First, we collected aggregate data on the total number of visitors each year to a large (though not comprehensive) set of outdoor recreation sites in Maryland, Virginia, and Delaware. Second, we collected data on water quality conditions in the Chesapeake Bay, interpolated to a relatively fine spatial scale from a set of water quality monitoring stations in the bay. Third, we collected data on the number and locations of households within driving distance of the bay from the U.S. Census Bureau. Finally, we collected supplemental individual-level data on outdoor recreation trip demands from two stated preference surveys. These data are described in more detail in the following subsections.

Aggregate visitation data

The dependent variables for the aggregate participation and site choice model are the counts of the total number of visitors to a select set of outdoor recreation sites in the Chesapeake Bay watershed between 2001 and 2010. According to the National Park Service (2013), there are 1,150 public recreation sites[43] on streams and other water bodies in the Chesapeake Bay watershed that provide access for boating, fishing, swimming, or viewing water or shoreline areas. These sites range from small stream side parks and boat launches to large national parks and conservation areas. Boating access is provided at 61% of sites, fishing access at 53% of sites, viewing access at 23% of sites, and swimming access at 7% of sites (National Park Service 2013 p 19). No data were available on the visitation rates for most of these sites. We obtained visitation data, provided by Abt Associates, on 142 state and national parks in Maryland, Virginia, and Delaware, 38 of which have water access to the Chesapeake Bay or a major tributary to the Bay.

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[43] Including all sites owned and managed by a public entity on tidal streams and bays with boating opportunities, streams classified as "fifth-order" and higher, and smaller streams that are part of an established water trail.

The site data are summarized in Table 40. Annual visitation data were not available for all ten years at all sites. The average number of visitors to all sites between 2001 and 2010 was approximately 66 million per year. This figure includes roughly 25 million visitors per year to the Blue Ridge Parkway and George Washington Memorial Parkway combined. The average total number of visitors to all sites with water access to the Chesapeake Bay was approximately 13.5 million visitors per year. We also collected data on amenities available at each of the 38 sites with water access to the bay. Of these 38 sites, 20 have boating facilities, 32 have fishing facilities, 10 have swimming facilities, 34 have hiking trails, 32 have restrooms, and 35 have on-site parking. The data also include latitude and longitude coordinates for each site, which allowed us to estimate the average water quality conditions near each site with water access in each year by cross-referencing the location of each site to a large database of spatially interpolated water quality measurements in the bay collected and maintained by EPA's Chesapeake Bay Program Office, as described below.

Water quality data

To characterize the water quality conditions at each site with water access to the Chesapeake Bay or one of its major tributaries, we used estimates of the concentration of chlorophyll-a (CL), dissolved oxygen (DO), and Secchi depth (SD). Estimates of these water quality indicators were based on the distance-weighted average of interpolated values at the centroids of the three nearest cells in the Chesapeake Bay Estuary Model (CBEM) (Kim 2013). Values in each model cell were interpolated using inverse distance weights based on direct measures of these parameters at the nearest water quality monitoring stations in the bay. The final six columns in Table 40 contain averages for each water quality measure across the 10 years of model outcomes under the simulated baseline and TMDL conditions.

Number and locations of households and travel costs

As explained above, stage 1 of estimation used a contraction mapping algorithm to find the set of ASCs that make the predicted visits to each site, $\hat{Y}_j$, match the observed visits, Y_j. The number of predicted visits to site j was calculated as $\hat{Y}_j = D\sum_{z=1}^{Z} N_z\hat{p}_{zj}$, where D is the number of choice occasions per year, N_z is the number of people living in zip code z, and $\hat{p}_{zj}$ is the predicted probability that an individual living in zip code z will visit site j on any choice occasion, which depends on the travel cost between z and j. Therefore, predicting the total number of trips to each site in each year requires data on the number of people who live within driving distance of the Chesapeake Bay and the distances between their homes and each recreation site.

We used data from the U.S. Census Bureau to estimate the number of adults living in each of the 7,132 zip codes that are within a 3-hour drive of at least one site in the dataset. We used MPMilage to calculate the shortest one-way driving distance between each zip code centroid and each site in the dataset. Two-way travel costs between each zip code and site in each year were calculated as the sum of the vehicle operating cost and the opportunity cost of time. The vehicle operating cost, including the cost of fuel, was estimated as $0.55 per mile multiplied by the two-way travel distance between each zip code and site. The opportunity cost of time was estimated using one half of the average hourly wage rate in the zip code multiplied by the two-way travel distance divided by an assumed average speed of 50 miles per hr. The average hourly wage rate was estimated using the average annual income in the zip code divided by an assumed average of 1,800 hours worked per year.

Supplemental survey data

As described in section 4.2.1, to help estimate the stage 1 parameters we used supplemental data on the number of outdoor recreation trips taken to the Chesapeake Bay by a random sample of nearby residents. For this purpose, we obtained data from two stated preference (SP) surveys. The first was a survey conducted by Kirkley *et al*. (2011). The survey was designed to examine people's preferences for imposing more stringent harvest restrictions on the Atlantic menhaden commercial fishery. The survey was implemented in Maryland and Virginia using three modes: telephone, mail, and internet. See Kirkley *et al*. (2011) for a complete description of the survey. For the dependent variable in the linked participation and site-choice model described in section 4.2.1, we used survey responses to the "trips" question on the survey: "About how many trips did you make from your home to the Bay for the main purpose of outdoor recreation during the past 12 months?" We used each respondent's zip code to estimate their travel costs to each site in the dataset, as described in the previous sub-section, and we use the demographic information provided by each respondent as additional independent variables to help explain the respondents' participation choices. Summary statistics for these data are shown in Table 41.

We also obtained data from a set of supplemental questions added to a new SP survey that was designed and implemented for this study, which is described in detail in Moore *et al*. (2015). For the dependent variable in the linked participation and site-choice model described in section 4.2.1, we used survey responses to a supplemental question on the Chesapeake Bay SP survey that reads as follows: "In the last 12 months, how many times did you visit an outdoor recreation site on the Chesapeake Bay?" As with the Kirkley *et al*. survey data, we used each respondent's zip code to estimate travel costs and the demographic information as additional independent variables in stage 1 of estimation. Summary statistics for these data are shown in Table 42.

4.2.4 Estimation of benefits

To estimate the aggregate willingness to pay for the predicted water quality improvements at outdoor recreation sites on the Chesapeake Bay associated with the TMDL, we first calculated the average levels of chlorophyll-a, dissolved oxygen, and Secchi depth during the months of May-September near each outdoor recreation site as predicted by the Chesapeake Bay Estuary Model under the baseline and TMDL scenarios. The differences between the averages of these water quality measures at each site over the 10 simulated years under each scenario were then used to calculate total WTP under each model specification. The per trip marginal willingness to pay for water quality attribute k is β_k/λ, i.e., the estimated site attribute coefficient (the marginal utility of the attribute) divided by the estimated travel cost parameter (the marginal utility of income), so we calculated the average annual aggregate willingness to pay as follows:

$$WTP = \sum\nolimits_{j=1}^{J} Y_{tj} \sum\nolimits_{k=1}^{K} \left(\frac{\hat{\beta}_k}{\hat{\lambda}} \Delta \bar{Q}_{jk} \right). \tag{54}$$

Equation (54) gives an estimate of total willingness to pay for water quality improvements for all trips represented in the aggregate visitation data. However, the aggregate visitation data represent only a subset of the available sites and therefore only a portion of the total trips to the Chesapeake Bay. To generate an estimate of total WTP to all available sites on the

Chesapeake Bay, some of which are not represented in the aggregate visitation data, we assumed that the WTP per trip estimated using the aggregate visitation data is an unbiased estimate of WTP per trip to all available sites. Then we scaled up the estimate of WTP as calculated using equation (54) by an estimate of the total number of recreation trips to Chesapeake Bay sites divided by the average annual trips represented in the aggregate visitation data, which was 13.5 million per year. We estimated the total number of recreation trips to the Chesapeake Bay using data from supplemental questions on the Chesapeake Bay TMDL stated preference survey. Recall that one question on the survey asked: "In the last 12 months, how many times did you visit an outdoor recreation site on the Chesapeake Bay?" To estimate total trips to the bay, we calculated the average response to this question by the home state of the survey respondents, then multiplied these averages for each state by the total number of residents in each state of age 18 and above and summed these estimates across the states. The resulting estimate of the total number of trips to outdoor recreation sites on the Chesapeake Bay was 42 million per year.

4.2.5 Results

We estimated the ASCs for all sites plus δ and λ using aggregate site visitation and water quality data for 2010, since this was the year closest in time to when the supplemental SP survey data were collected. (The SP survey by Kirkley *et al*. was conducted in 2010, and the Chesapeake Bay TMDL SP survey was conducted in 2013-2014.) Then we estimated the ASCs for all sites in the years 2001-2009 conditional on the estimates of δ and λ based on the 2010 data. Table 43 and Table 44 show the estimates of δ and λ using the Kirkley *et al*. and Chesapeake Bay TMDL SP survey data, respectively. Most coefficient estimates are statistically significant, and we see no obviously counter-intuitive signs on the coefficients. Note that the demographic variables, including household income, were included in the "stay home" indirect utility function, so the negative coefficient on income indicates a positive income elasticity with respect to trip demands, and the indirect utility for each site was computed as written in equation (46), with λc_{ij} subtracted from the indirect utility, so a positive coefficient on the travel cost variable indicates a negative effect off price on demand, as expected.

Results for the stage 2 OLS regressions and associated benefit estimates are shown in Table 45 (based on the Kirkley *et al*. survey data) and Table 46 (based on the Chesapeake Bay TMDL SP survey data). With no strong theory to help guide selection of the most appropriate model specification, we examined a wide variety of specifications distinguished by whether site-specific fixed effects or site amenity indicator variables were included in the estimating equation and by the set of water quality variables that were included. For compactness, we report eight specifications in each table that span most of the full range of results, including those from specifications not reported here.

The advantages and disadvantages of site-specific fixed effects were discussed above in section 4.2.2, and the implications discussed there are borne out in the results in Table 45 and Table 46. When site-specific fixed effects were included, the water quality coefficients capture only the influence of within-site temporal variability in water quality conditions on the site visitation patterns—any influence of the differences in long-run average water quality conditions were absorbed in the site-specific fixed effect—and so in these specifications the overall fit of the model was high but the water quality variables generally had low statistical significance. When site-specific fixed effects were not included, then we included site amenity indicator variables to control for the observable differences among the sites, and in these specifications the water quality variables as a group explained a significant amount of the variation in the site visitation data. In the specifications where site amenity indicators were included in lieu of site-specific fixed effects, many of the water quality variables had t-statistics that indicate statistical significance at the conventional 5% level. Specifically, Table 45 and Table 46 show results for specifications with each of the three water quality variables entered individually as first-order terms, jointly as first-order terms, then adding second-order terms, then adding a natural log transformation to the estimating equation. For each model we also report in the final column of each table a specification based on "testing down:" starting from the most general specification (including includes first-order, second-order, and log terms), the variable with the lowest t-statistic was dropped from the estimating equation and the model was re-estimated. This process was repeated until all remaining water quality variables were individually statistically significant at the 5% level (or nearly so). It is well known that standard t-statistics will be biased upward when undertaking multiple ad hoc specification tests as was done here (Leamer 1983), so we make no claims about having found the best model among the variety specifications that we examined and we understand that the nominal significance levels are conditional on the selected model and so do not account for the uncertainty associated with the model selection process itself. Instead, our approach was to generate results under a variety of alternative specifications that had some potential ex ante plausibility and then examine the range of benefit estimates that emerged from the full suite of considered specifications.

Looking across all specifications including site amenity indicator variables in both tables, more than half of the water quality variables were statistically significant. In most cases, the coefficients on CL and ln(CL) were positive while the coefficients on CL^2 were negative.[44] This was consistent with our prior expectations, discussed in section 4.2.2 above, that when chlorophyll-a is low the marginal effect may be positive but at higher levels of enrichment the marginal effect may be negative. The results for DO were much more mixed than the relatively robust results for CL. We anticipated that DO would have a generally positive effect on site visits, since low DO conditions are detrimental to aquatic species that may attract recreational anglers to the sites. When DO alone appeared in the estimating equation its coefficient was positive, but when other water quality variables also were included the DO and ln(DO) coefficient estimates were positive only about half of the time. The results for SD were also mixed. When SD alone appeared in the estimating equation its coefficient was positive, but it is more often negative when other water quality variables were also included.

Despite the fragility of dissolved oxygen and Secchi depth coefficients, the resulting benefit estimates, which were the ultimate quantities of interest in this study, were fairly robust across most of the alternative specifications. In all specifications reported in Table 45 and Table 46, the central (50th percentile) benefit estimates were positive. In 16 of the 22 specifications, the 90% confidence intervals for the estimated benefits did not include zero, and in 15 of 22 specifications the central estimates of benefits were between \$150 and \$300 million per year.

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[44] For the specifications with site amenity indicators, which had statistically significant CL coefficient estimates, the implied levels of CL where the partial effect turns from positive to negative are between 15 and 30 μg/L. For comparison, the average baseline levels of CL among the sites was 13.8 μg/L and the inner 90th percentile range was 5.3 to 23.3 μg/L. The specifications with site fixed effects did not yield statistically significant CL coefficient estimates and the associated turning points were not in a plausible range.

We estimated the portion of the total value attributable to recreational fishing in a crude way using other supplemental questions on the Chesapeake Bay TMDL stated preference survey, which asked respondents what activities they engaged in on their most recent trip to an outdoor recreation site on the Chesapeake Bay. Of those respondents who answered these supplemental questions, 30% selected “fishing” as one of the activities on their last trip and 6.5% selected “fishing” as the only activity on their last trip. Applying these bounds to the range cited above, we attributed between roughly $10 and $90 million per year to recreational fishing trips and between $105 and $280 million per year to other recreation activities.

5 Conclusions

In this study we estimated the economic benefits of the Chesapeake Bay TMDL to commercial fish harvesters and consumers, recreational anglers, and other outdoor recreators using a variety of data sources and linked models. To forecast the impacts of the TMDL on harvested fish and shellfish stocks, we used a summary of judgments from an expert panel and a multi-species model of Chesapeake Bay fisheries. We estimated price changes and benefits to consumers in commercial fish markets using a multi-stage inverse demand system. The resulting estimates of commercial fishing benefits range between $3 and $26 million per year.

We calculated benefits to recreational anglers using a linked participation and site-choice recreation demand model. We accounted for the sample selection bias caused by the non-random intercept survey sampling design using weights based on historic visitation frequencies at each intercept site. The resulting estimates of recreational fishing benefits range between $5 and $59 million per year.

We used a separate recreation demand model to estimate the benefits associated with other outdoor recreation activities based on aggregate data on the total number of visitors to national and state parks in Maryland, Virginia, and Delaware and survey data on the number of recreation trips taken to the Chesapeake Bay collected from a random sample of individuals in the study area. The central estimates of the outdoor recreation benefits (exclusive of recreational fishing) are between $105 to $280 million per year.

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Tables and Figures

Table 1. Expert judgments of potential changes in Chesapeake Bay species' stock sizes with TMDL relative to current conditions. Columns correspond to individual expert panelists. The meaning of the symbols are as follows: "++" = large increase, "+" = small increase, "0/+" = no measurable change or small increase, "0" = no measurable change, "-" = small decrease, "--" = large decrease, "." = no opinion.

	Expert 1	Expert 2	Expert 3	Expert 4	Expert 5	Expert 6	Expert 7
Blue crab	+	+	+	+	++	++	++
Atlantic menhaden	-	-	-	-	-	.	.
Summer flounder	0	0/+	0/+	+	+	0	.
Striped bass	+	0	0	0	+	+	.
Northern quahog clam	++	.	.	+	0	+	+
Atlantic croaker	0	0	0	0	0	+	+
Eastern oyster	+	++	+	+	++	++	+
Black sea bass	0	0	0	0	0	.	.
Spot	+	+	+	0	0	0	+
Bluefish	0	0	0	0	0	0	0
White perch	.	+	+	0	-	+	0
Tautog	0	0	0	0	0	0	0
American shad	.	.	.	+	0	+	0
Blue catfish	+	.	.	0	--	.	.
American eel	+	.	0	0	0	0	0/+

Table 2. Qualitative expert assessments from Table 1 converted to quantitative estimates (first seven columns of numbers) and averaged (final column), assuming “large increase” = 0.25, “small increase” = 0.1, “no measurable change” or no response = 0, “no measurable change or small increase” = 0.05, “small decrease” = -0.1, and “large decrease” = -0.25.

	Expert 1	Expert 2	Expert 3	Expert 4	Expert 5	Expert 6	Expert 7	Average
Blue crab	0.1	0.1	0.1	0.1	0.25	0.25	0.25	0.164
Atlantic menhaden	-0.1	-0.1	-0.1	-0.1	-0.1	0	0	-0.071
Summer flounder	0	0.05	0.05	0.1	0.1	0	0	0.043
Striped bass	0.1	0	0	0	0.1	0.1	0	0.043
Northern quahog clam	0.25	0	0	0.1	0	0.1	0.1	0.079
Atlantic croaker	0	0	0	0	0	0.1	0.1	0.029
Eastern oyster	0.1	0.25	0.1	0.1	0.25	0.25	0.1	0.164
Black sea bass	0	0	0	0	0	0	0	0.000
Spot	0.1	0.1	0.1	0	0	0	0.1	0.057
Bluefish	0	0	0	0	0	0	0	0.000
White perch	0	0.1	0.1	0	-0.1	0.1	0	0.029
Tautog	0	0	0	0	0	0	0	0.000
American shad	0	0	0	0.1	0	0.1	0	0.029
Blue catfish	0.1	0	0	0	-0.25	0	0	-0.021
American eel	0.1	0	0	0	0	0	0.05	0.021

Table 3. Required and optimal ranges of temperature, salinity, and dissolved oxygen for species stages represented in the multi-species fishery model (MSFM). Information sources and assumptions are given in the notes at the bottom of the table.

		Temperature [^{o}C]				Salinity [‰]				DO [mg l^{-1}]		
		required		optimal		required		optimal		req.	opt.	
		min	max	min	max	min	max	min	max	min	min	Notes
Blue crab	larv	16	30	20	29	20	35	21	30	0.9	0.9	a
	juvn	-99	-99	15	30	3	56	10	30	1.2	2.8	b
	adlt	-99	-99	15	30	3	56	10	30	0	2.8	b
Atlantic menhaden	larv	5	33	14	30	0	35	5	10	1.1	3	c
	juvn	5	33	14	30	0	35	5	10	1.1	3	c
	adlt	5	33	14	33	0	35	3.5	35	1.1	1.1	d
Northern quahog clam	larv	4	36.5	9	31	10	35	21	30	0.5	2.4	e
	juvn	4	36.5	9	31	10	35	21	30	0.5	2.4	e
	adlt	4	36.5	9	31	10	35	21	30	0.5	2.4	e
Striped bass	larv	12	24	16	21	0	25	1	11	4	5	f
	juvn	2	35	24	27	0	33	1	15	1.4	4	f
	adlt	0	31	14	25	0	35	0	35	2	3	f
Eastern oyster	larv	15	34	25	33	8	39	13	33	0	1.5	g
	juvn	17.5	35	20	30	0	42	12	27	1	1	h
	adlt	-2	41	20	32	5	44	10	30	0	1	g
Atlantic croaker	larv	1	36	13	28	0	36	5	20	0.5	1	i
	juvn	1	36	13	28	0	36	5	20	0.5	1	j
	adlt	5	35	10	28	0	36	15	19	2	4	k
Summer flounder	larv	4	28	9	27	10	60	28	60	1	2	l
	juvn	4	28	9	27	10	60	28	60	1	2	l
	adlt	4	28	9	27	10	60	28	60	1	2	l
Spot	larv	4	31	17	25	0	60	0	60	2	4	m
	juvn	4	31	17	25	0	60	0	60	2	4	m
	adlt	4	31	17	25	0	60	0	60	2	4	m
Black sea bass	larv	6	28	13	21	1	36	14	36	2	4	n
	juvn	6	28	13	21	1	36	14	36	2	4	n
	adlt	6	28	13	21	1	36	14	36	2	4	n
White perch	larv	3	34	12	33	0	22	0	16	0.8	2.9	o
	juvn	3	34	12	33	0	22	0	16	0.8	2.9	p
	adlt	3	34	5	18	0	22	5	18	2	4	q
Blue catfish	larv	0	40	26	29	0	17	0.5	3	2	5	r
	juvn	0	40	26	29	0	17	0.5	3	2	5	r
	adlt	0	40	26	29	0	17	0.5	3	2	5	r
Bluefish	larv	17	26	21	25	30	38	30	32	2	4	s
	juvn	10	35	15	27	5	36	18	31	2	4	t
	adlt	8	35	14	30	5	36	25	35	5.1	8	t
Tautog	larv	6	28	13	21	1	36	14	36	2	4	u

	juvn	6	28	13	21	1	36	14	36	2	4	u
	adlt	6	28	13	21	1	36	14	36	2	4	u
American shad	larv	8	26	14	21	0	60	0	60	2	5	v
	juvn	8	26	14	21	0	60	0	60	2	5	v
	adlt	8	26	14	21	0	60	0	60	2	5	v

NOTES:

a. Schlenger (2012 Table 1.2).

b. Schlenger (2012 Table 1.2). Min and max temperatures depend on salinity: Min T = 0.0484S+5.1548, Max T = 0.0735S+36.5.

c. Schlenger (2012 Table 1.18).

d. Values in common with juvenile menhaden transferred from Schlenger (2012 Table 1.18). Other values from Funderburk *et al*. (1991 p viii).

e. All life stages assumed to have same tolerance ranges.

f. Schlenger (2012 Table 1.6).

g. Schlenger (2012 Table 1.4).

h. Funderburk *et al*. (1991 p viii).

i. Assumed same as juveniles.

j. Schlenger (2012 Table 1.20).

k. Lassuy (1983).

l. All life stages assumed to have same tolerance ranges. Temperature ranges from Packer *et al*. (1999 p 17-180). Required min and max based on range of temps associated with catch in any season; optimal min and max based on range of temps associated with summer catches. Salinity required min based on Able and Kaiser (1994 p 40). Salinity optimal min based on Able and Kaiser (1994 p 58). Max salinity values set to 60 to ensure high salinity is not constraining in the model. Dissolved oxygen values based on Massey *et al*. (2006), Figure 4.

m. All life stages assumed to have same tolerance ranges. Min and max optimal temperature based on spawning temperature range as reported by the Smithsonian Marine Station at Fort Pierce, available online at http://www.sms.si.edu/irlspec/Leiosto_xanthu.htm. Required temperature, salinity, and Dissolved oxygen min and max based Howard Townsend pers. comm.

n. All life stages assumed to have same tolerance ranges. Values for northern searobin from Roberts (1978).

o. Assumed same as juveniles.

p. Schlenger (2012 Table 1.24).

q. Required temperature and salinity thresholds transferred from juvenile white perch, based on Schlenger (2012 Table 1.24). All other figures from Funderburk *et al*. (1991 p viii).

r. Howard Townsend pers. comm.

s. Schlenger (2012 Table 1.10). DO values transfered from juveniles.

t. Schlenger (2012 Table 1.10).

u. All life stages assumed to have same tolerance ranges. Values for northern searobin from Roberts (1978).

v. All life stages assumed to have same tolerance ranges. Temperature and dissolved oxygen values based on Stier and Crance (1985). Temperature ranges based on observed swimming activities. Min required dissolved oxygen level based on detrimental responses among juveniles. Min optimal dissolved oxygen level based on requirements for adult spawning. Shad are anadromous and can tolerate a wide range of salinity levels, so we use min and max salinity levels of 0 and 60 to ensure salinity is not constraining in the model.

Table 4. Average May through September habitat volumes for larval, juvenile, and adult stages of 14 fish and shellfish species in the Chesapeake Bay. Habitat volumes are reported as a fraction of the total volume of the bay. All figures are averaged over the full ten-year simulation period using baseline and TMDL projections, based on rainfall data for the years 1991-2000. "Historic max" habitat volumes were calculated ignoring DO conditions; i.e., this is the habitat volume that would have obtained if DO levels were never constraining.

		Required				Optimal			
		Hist. max	Hist.	Base	TMDL	Hist. max	Hist.	Base	TMDL
Blue crab	larv	0.0325	0.0317	0.0324	0.0325	0.0224	0.0220	0.0223	0.0224
	juvn	0.1254	0.1178	0.1234	0.1251	0.1060	0.0916	0.0969	0.1000
	adlt	0.1254	0.1254	0.1256	0.1256	0.1060	0.0916	0.0969	0.1000
Atlantic menhaden	larv	1.0000	0.9629	0.9904	0.9980	0.1181	0.1162	0.1174	0.1179
	juvn	1.0000	0.9629	0.9904	0.9980	0.1181	0.1162	0.1174	0.1179
	adlt	1.0000	0.9629	0.9904	0.9980	0.9497	0.9128	0.9401	0.9477
Northern quahog clam	larv	0.1098	0.1056	0.1097	0.1100	0.0279	0.0267	0.0273	0.0275
	juvn	0.1098	0.1056	0.1097	0.1100	0.0279	0.0267	0.0273	0.0275
	adlt	0.1098	0.1056	0.1097	0.1100	0.0279	0.0267	0.0273	0.0275
Striped bass	larv	0.4772	0.4243	0.4401	0.4490	0.0543	0.0512	0.0525	0.0530
	juvn	1.0000	0.9546	0.9841	0.9955	0.1603	0.1428	0.1496	0.1526
	adlt	1.0000	0.9388	0.9701	0.9858	0.6556	0.6005	0.6200	0.6314
Eastern oyster	larv	0.1140	0.1140	0.1142	0.1142	0.0322	0.0292	0.0312	0.0319
	juvn	0.1074	0.1013	0.1061	0.1073	0.0736	0.0682	0.0723	0.0735
	adlt	0.1235	0.1235	0.1238	0.1238	0.0839	0.0785	0.0827	0.0838
Atlantic croaker	larv	1.0000	0.9781	0.9984	0.9998	0.7036	0.6750	0.6967	0.7023
	juvn	1.0000	0.9781	0.9984	0.9998	0.7036	0.6750	0.6967	0.7023
	adlt	1.0000	0.9389	0.9702	0.9858	0.2565	0.2072	0.2198	0.2263
Summer flounder	larv	0.1094	0.1029	0.1080	0.1093	0.0026	0.0026	0.0026	0.0026
	juvn	0.1094	0.1029	0.1080	0.1093	0.0026	0.0026	0.0026	0.0026
	adlt	0.1094	0.1029	0.1080	0.1093	0.0026	0.0026	0.0026	0.0026
Spot	larv	1.0000	0.9388	0.9701	0.9858	0.5234	0.4581	0.4743	0.4828
	juvn	1.0000	0.9388	0.9701	0.9858	0.5234	0.4581	0.4743	0.4828
	adlt	1.0000	0.9388	0.9701	0.9858	0.5234	0.4581	0.4743	0.4828
Black sea bass	larv	0.9646	0.9036	0.9348	0.9504	0.1439	0.1245	0.1303	0.1338
	juvn	0.9646	0.9036	0.9348	0.9504	0.1439	0.1245	0.1303	0.1338
	adlt	0.9646	0.9036	0.9348	0.9504	0.1439	0.1245	0.1303	0.1338
White perch	larv	0.8623	0.8335	0.8579	0.8616	0.5186	0.4869	0.5012	0.5083
	juvn	0.8623	0.8335	0.8579	0.8616	0.5186	0.4869	0.5012	0.5083
	adlt	0.8623	0.8033	0.8335	0.8487	0.1258	0.1129	0.1181	0.1216
Blue catfish	larv	0.5850	0.5579	0.5730	0.5800	0.0055	0.0053	0.0054	0.0054
	juvn	0.5850	0.5579	0.5730	0.5800	0.0055	0.0053	0.0054	0.0054
	adlt	0.5850	0.5579	0.5730	0.5800	0.0055	0.0053	0.0054	0.0054
Bluefish	larv	0.0027	0.0027	0.0027	0.0027	0.0017	0.0017	0.0017	0.0017
	juvn	0.9394	0.8782	0.9095	0.9252	0.3171	0.2662	0.2784	0.2842

	adlt	0.9394	0.7395	0.7868	0.8095	0.0504	0.0059	0.0055	0.0052
Tautog	larv	0.9646	0.9036	0.9348	0.9504	0.1439	0.1245	0.1303	0.1338
	juvn	0.9646	0.9036	0.9348	0.9504	0.1439	0.1245	0.1303	0.1338
	adlt	0.9646	0.9036	0.9348	0.9504	0.1439	0.1245	0.1303	0.1338
American shad	larv	0.8392	0.7853	0.8130	0.8265	0.2831	0.2465	0.2558	0.2608
	juvn	0.8392	0.7853	0.8130	0.8265	0.2831	0.2465	0.2558	0.2608
	adlt	0.8392	0.7853	0.8130	0.8265	0.2831	0.2465	0.2558	0.2608

Table 5. List of variables and parameters used in the Chesapeake Bay multi-species fishery model (MSFM), with definitions and measurement units. Symbols are listed in the order they first appear in the main text. Variables and parameters with a "$\{A, B\}$" subscript are patch-specific; all other parameters take the same value in both patches. All variables and parameters are species-specific; we omit species indices to streamline the notation.

Symbol	**Definition**	**Units**
$\gamma_{\{A,B\}}$	Larval retention coefficient—fraction of larvae that return to their parent estuary	fraction
$R_{\{A,B\}}$	Abundance of recruits added to the adult stock each year	$[\,\mathrm{lb} \cdot \mathrm{yr}^{-1}\,]$
$N_{\{A,B\}}$	Abundance of adults, i.e., harvestable biomass	$[\,\mathrm{lb}\,]$
M	Natural mortality rate of adults	$[\,\mathrm{yr}^{-1}\,]$
F	Fishing mortality rate	$[\,\mathrm{yr}^{-1}\,]$
S	Abundance of spawners (reproducing adults)	$[\,\mathrm{lb}\,]$
V_k	Habitat volume—fraction of water in Chesapeake Bay with DO, salinity, and temperature conditions that are suitable for survival and growth in month k for a particular species life-stage	fraction
$\alpha_{\{A,B\}}$	Eggs deposited per spawner	$[\,\mathrm{lb} \cdot \mathrm{lb}^{-1}\,]$
$\beta_{\{A,B\}}$	Instantaneous mortality rate of eggs and larvae in suitable habitat	$[\,\mathrm{yr}^{-1}\,]$
$a_{\{A,B\}}$	Eggs per spawner that survive to juvenile stage	$[\,\mathrm{lb} \cdot \mathrm{lb}^{-1}\,]$
$\tilde{V}$	Volume of habitat in the Chesapeake Bay that is always suitable for the egg and larval stage May through September	fraction
$J_{\{A,B\}}$	Abundance of juveniles	$[\,\mathrm{lb}\,]$
$b_{\{A,B\}}$	Density-dependent mortality coefficient for juveniles, i.e., increment of juvenile per capita mortality risk per unit density of juveniles	$[\,\{\text{stage duration}\}^{-1} \cdot \mathrm{lb}^{-1}\,]$
$\hat{V}$	Harmonic mean of monthly juvenile stage habitat volumes in the Chesapeake Bay during May through September	fraction
$\omega_{\{A,B\}}$	Fishery retention coefficient—fraction of harvested adults that are landed at ports in their parent estuary	fraction
B	Fishing exploitation rate, i.e., the fraction of adults just after recruitment that are harvested during the year	fraction
$H_{\{A,B\}}$	Harvest of adults from each patch	$[\,\mathrm{lb} \cdot \mathrm{yr}^{-1}\,]$
$Q_{\{A,B\}}$	Landings of fish at ports in each patch	$[\,\mathrm{lb} \cdot \mathrm{yr}^{-1}\,]$

Table 6. Fishery landings data used to calibrate the Chesapeake Bay multi-species fishery model.

	Entire Atlantic coast			Chesapeake Bay			Notes
	Commercial landings	Recreational landings	Price	Commercial landings	Recreational landings	Price	a
	[lb · yr $^{-1}$]	[lb · yr $^{-1}$]	[$ · lb^{-1}]	[lb · yr $^{-1}$]	[lb · yr $^{-1}$]	[$ · lb^{-1}]	
Blue crab	129,402,075	8,995,866	1.00	78,423,070	5,902,812	1.05	b
Atl. menhaden	464,718,322	0	0.08	400,425,892	0	0.07	
N. quahog clams	5,364,320	0	6.79	2,487,206	0	7.89	
Striped bass	7,407,144	24,746,003	2.34	4,715,332	4,266,790	2.12	
Eastern oyster	3,057,502	0	8.92	1,430,257	0	5.19	
Atl. croaker	14,842,812	4,645,820	0.69	8,768,857	3,914,068	0.77	
Summer flounder	12,275,261	6,364,520	2.22	3,098,271	935,135	1.68	
Spot	3,466,888	2,291,865	0.83	2,595,768	1,431,044	0.89	
Black sea bass	2,001,386	2,830,741	2.92	424,430	92,033	3.34	
White perch	2,045,616	1,162,399	0.79	1,657,637	692,458	0.78	
Blue catfish	1,353,456	238,845	0.35	1,353,192	238,799	0.35	c
Bluefish	6,059,030	14,788,917	0.50	548,230	626,838	0.52	
Tautog	265,793	2,926,906	2.78	11,681	331,883	2.32	
American shad	559,737	0	1.01	6,801	0	0.45	

Notes:

a. All commercial harvest estimates are averages of Chesapeake Bay landings between 2008-2012 as recorded by NOAA's commercial fisheries statistics (http://www.st.nmfs.noaa.gov/st1/commercial/). Unless otherwise noted, all recreational landing estimates are the average annual observed recreational harvest in Maryland and Virginia between 2008-2012 as recorded by NOAA's Marine Recreational Information Program (http://www.st.nmfs.noaa.gov/recreational-fisheries/index). Market prices are average of total revenues per total weight of landings between 2008-2012.

b. Blue crab recreational landings based on NOAA estimate that 93% of catch is commercial and 7% is recreational (National Oceanic and Atmospheric Administration 2010).

c. Based on assumption that 7% of total bay-wide catch is recreational (2013 Chesapeake Bay Blue Crab Advisory Report, p 5 [http://www.chesapeakebay.net/documents/Final_CBSAC_Advisory_Report_2013_.pdf]).

Table 7. Biological and fishery parameters used to calibrate the Chesapeake Bay multi-species fishery model.

	Fishing mortality rate, F	MSY fishing mortality rate, F_{msy}	Adult natural mortality rate, M	Larval retention coefficients		Harvest retention coefficients		Notes
				γ_A	γ_B	ω_A	ω_B	
	[yr^{-1}]	[yr^{-1}]	[yr^{-1}]					
Blue crab	0.435	0.675	0.900	0.90	0.75	1	1	a
Atl. menhaden	1.260	1.180	0.450	0.50	0.95	0.50	0.95	b
N. quahog clam	0.125	0.250	0.064	1	1	1	1	c
Striped bass	0.130	0.340	0.120	1	1	1	0.55	d
Eastern oyster	0.053	0.070	0.350	1	1	1	1	e
Atl. croaker	0.098	0.390	0.290	0.90	0.75	0.90	0.90	f
Summer flounder	0.241	0.310	0.250	0.90	0.75	0.90	0.75	g
Spot	1.000	1.000	0.749	0.90	0.90	0.90	0.90	h
Black sea bass	0.240	0.440	0.400	0.90	0.90	0.95	0.75	i
White perch	0.490	0.600	0.500	1	1	1	1	j
Blue catfish	0.079	0.158	0.187	1	1	1	1	k
Bluefish	0.138	0.190	0.200	0.75	0.25	0.94	0.06	l
Tautog	0.300	0.200	0.094	0.95	0.95	1	1	m
American shad	0.300	0.440	0.200	1	1	0.99	0.90	n

Notes:

a. Blue crab: M is in the middle of the range of estimates by Miller *et al.* (2011 p 4). Also based on Miller *et al.* (2011), estimated exploitation rate and max exploitation rate in 2009 were 0.24 (p v) and 0.34 (p 48), respectively. With M = 0.9, F = 0.435 gives exploitation rate of 0.24 and F = 0.675 gives exploitation rate of 0.34. Spawning occurs in estuaries, but larvae are swept into coastal waters and then return to inshore estuaries as juveniles later in the season. Larval retention coefficients chosen based loosely on Epifanio and Garvine (2001), who state “Mean circulation of surface water in the mid-shelf is characterized by wind-driven, northward flow at speeds sufficient to retain larvae within the MAB and in many cases in the vicinity of the parent estuary itself.” Fisheries are prosecuted in estuaries or adjacent coastal waters, so we expect harvest spillovers to be negligible and therefore set both harvest retention coefficients to one.

b. Atlantic menhaden: F is estimate for 2008 (Atlantic States Marine Fisheries Commission 2011a p 7). FTHRESHOLD used as proxy for FMSY (Atlantic States Marine Fisheries Commission 2011a p 3). Adult natural mortality rate from ASMFC (2012 p 4). Spawning and fishing both occur mainly in offshore waters. Epifanio and Garvine (2001) emphasize “…the role of alongshelf flow in transporting the larvae. Results of both model experiments and corroborating observational studies indicate that spawning sites may be hundreds of km distant from an estuarine nursery in the alongshelf direction....” This suggests a loose coupling between larvae and their parent estuary, so the larval retention coefficients were set as low as feasible while still allowing the calibrated model to match the observed landings data. Fishery retention coefficients were set to reflect the fact that the vast majority of the harvest is landed at Chesapeake Bay ports while still allowing some harvest spillovers.

c. Northern quahog clam: Adult natural mortality rate based on assumption that 95% of an unfished population with a stable age distribution are between 1 and 36 years of age (Bricelj 2002 p 31). No recent stock assessments have been conducted for hard clams and no quantitative estimates of fishing mortality rates in the Chesapeake Bay could be found. Based on a bio-energetic model of hard clams applied to Great South Bay, New York (Kraeuter *et al*. 2008), we assume $F_{MSY} = 0.25$. No harvest restrictions apply to hard clams, so we assume that the current fishing mortality rate is substantially below F_{MSY}; specifically we assume $F = ½ F_{MSY}$. Spawning and fishing occur in estuaries, so we set all larval and harvest retention coefficients to 1.

d. Striped bass: F is estimate of "[a]verage fishing mortality on ages 3-8, which are generally targeted in producer areas (Chesapeake Bay, Delaware Bay, and Hudson River)" (Northeast Fisheries Science center 2013 p 495). Estimate of F_{MSY} from Nelson and Sharov (2013 p 40). Adult natural mortality rate calculated based on assumption that 95% of a stable age distribution among reproductive adults, fish aged 6 years and above, are 30 years or younger. Striped bass are anadromous so we set the larval retention coefficients to 1. Fishing occurs in both inshore and offshore waters, and according to Kohlenstein (1981), migration of adult female striped bass "could support the majority of the coastal striped bass fishery." We set the harvest retention coefficient for patch B such that the share of the total landings contributed by Chesapeake Bay fish is roughly 0.5.

e. Eastern oyster: Wilberg and Miller (2010 p 30) note that "…natural mortality of market-sized oysters varied among years between 15 and 59% and average 30% per year," so we use M = 0.3. Wilberg and Miller (2010 p 33) also report "…sustainable exploitation rates for Delaware Bay eastern oysters of 7% per year… [and] the estimated exploitation rate in Maryland … averaged 25% per year [between 1980-2008]." This is far higher than the estimated max sustainable exploitation rate of 0.07. On the assumption that fishing effort will be more effectively constrained in future years, we set both F equal to $0.75 \times F_{MSY}$ (following the recommendations for precaution by Roughgarden and Smith 1996). Spawning and fishing occur in estuaries, so we set all larval and harvest retention coefficients to 1.

f. Atlantic croaker: Fishing mortality set at 25% of F_{MSY} based on visual inspection of Figure 1 in Atlantic States Marine Fisheries Commission (2011b). Note that the same report cautions that "…absolute estimates of F and SSB are unavailable because of model uncertainty, [but] the general trends in the estimates from the model are considered reliable due to support from the data" (p 3). F_{MSY} also taken from Atlantic States Marine Fisheries Commission (2011b p 2). Adult mortality rate calculated based on the assumption that 99% of a stable age distribution among reproductive adults, fish aged 2 years and above, are 17 years or younger (Atlantic States Marine Fisheries Commission 2010b p 4). Eggs are hatched in coastal waters and after hatching larvae drift into estuaries, so we set the larval retention coefficients to our default values of 0.75. Adults move in and out of estuaries seasonally and range widely along the Atlantic coast during fall and winter. We set the harvest retention coefficients to 0.9 (close to the smallest pair of values that would allow the model to calibrate with positive carrying capacities for both patches).

g. Summer flounder: F_{MSY} and F as estimated for 2011 taken from Terceiro (2012 p iii). Adult mortality rate based on mean of age and sex-specific estimated values (Terceiro 2012 p 3). Summer flounder spawn offshore and larvae and post-larvae migrate to inshore waters in the fall, so we set the larval retention coefficients for patches A and B to 0.9 and 0.75. Most recreational fishing occurs inshore, but much of the commercial fishing activity occurs in coastal waters so we set the harvest retention coefficients to 0.9 and 0.75.

h. Spot: Adult natural mortality rate based on median value of estimates from age-constant methods (Atlantic States Marine Fisheries Commission 2010a p 11). F_{MSY} based on estimate of F that maximizes utilization of cohort biomass by Piner (1999). No quantitative estimate of the recent fishing mortality rate could be found, so we use $F = F_{MSY}$. Spot migrate seasonally between estuarine and coastal waters where spawning occurs, so some potential of both larval and harvest spillovers exists. We set all retention coefficients to 0.9, close to the smallest values that would allow the model to calibrate with positive carrying capacities for both patches.

i. Black sea bass: F and F_{MSY} from Shepherd (2012 Table 3). Natural mortality rate, M, from Shepherd (2012 p 3). Black sea bass migrate seasonally between estuarine and coastal waters where spawning occurs, so some potential of both larval and harvest spillovers exists. We set the retention coefficients close to the lowest values that allow the model to calibrate with positive carrying capacities in both patches.

j. White perch: "Biological reference points (BRP) have not been formally established although an F_{LIMIT} was suggested as F = 0.60. In the nine years between 2000 and 2010 for which sufficient data was available to estimate F, F did not exceed 0.49 and was well under the F_{LIMIT} of 0.60" (http://dnr2.maryland.gov/fisheries/Documents/Section_21_White_Perch.pdf). Adult mortality rate based on assumed average age of 4 years (http://www.dnr.state.md.us/irc/docs/00000260_12.pdf), and a stable age distribution corresponding to a constant mortality rate, M, which gives M = sqrt(1/avg age). White perch spend their entire lives in bays and tidal tributaries, so we set all retention coefficients to 1.

k. Blue catfish: Natural mortality rate based on assumed age of maturity equal to 5 years and 95 percent of adults under a stable age distribution are 20 years old or less (Graham 1999). Total mortality rate assumed to equal midpoint of range among several tributaries in Virginia reported by Greenlee and Lim (2011 p 11), which was 20.8% to 32.3% per year. Fishing mortality rate calculated as difference between total mortality and natural mortality rates. Current fishing mortality rate is thought to be well below a level needed to control the growth of the population, so we set F_{MSY} equal to two times F. Blue catfish live primarily in fresh waters and do not migrate into more saline coastal waters, so we set all retention coefficients to 1.

l. Bluefish: F and F_{MSY} from Wood (2013 p v); F set equal to average estimated F since 2000. Adult mortality rate from Wood (2013 p 4). Bluefish live predominantly in coastal waters, but some juveniles enter the Chesapeake Bay and other estuaries for 2-3 months before returning to the coastal stock. This type of offshore spawning species does not fit the standard two-patch model used for the other species, so we applied a modified version of the model to bluefish as described in Appendix D. In this case the larval retention coefficients indicate the fraction of juveniles spawned offshore that disperse to each patch, and the harvest retention coefficients indicate the fraction of the harvest of the single pool of offshore adults that are landed at ports in each patch. The larval retention coefficients are based on proportions of the coastwide biomass of early life-stage bluefish by region and season, as estimated by an ASMFC multi-species modeling panel (Howard Townsend pers. comm.). The harvest retention coefficients are based on the ratio of average Chesapeake Bay landings relative to average Atlantic coast landings, i.e., $\omega_A = Q_A/(Q_A + Q_B)$ and $\omega_B = Q_B/(Q_A + Q_B)$.

m. Tautog: The tautog fishery management plan F_{TARGET} is used as a proxy for F_{MSY} (Atlantic States Marine Fisheries Commission 2011c p 1). The fishery management plan indicates that F = 0.38, but this level is not sustainable in the model so we set F = 0.3 to indicate overfishing but still allow the model to calibrate and maintain a steady state. Adult natural mortality rate calculated based on the assumption that 95% of a stable age distribution among reproductive adults, fish aged 3 years and above, are 34 years or younger (http://www.chesapeakebay.net/fieldguide/critter/tautog). Tautog spawn in lower bays and offshore and exhibit strong site fidelity. We set the larval retention coefficient for patch A to 0.95 and for patch B to 0.3, and we set the harvest retention coefficients to 1.

n. American shad: We set F_{MSY} = 0.44, which is the mid-point of the range of values for the overfishing definition ($F_{30\%}$) reported by Haas-Castro (2006 p 3). It also is within the range of estimates for multiple sub-stocks of American shad reported by the Atlantic States Marine Fisheries Commission (2007). We set F = 0.3, an intermediate value consistent with the conclusion that shad are not experiencing overfishing (Haas-Castro 2006 p 3) and a harvest moratorium in several Chesapeake Bay states. Adult natural mortality rate based on assumed average adult lifespan of 5 years. Shad live in coastal waters and spawn in freshwater rivers and streams. Limited commercial fishing occurs in tidal waters where stocks may be mixed, so we set the harvest retention coefficients for patch B to 0.9. Landings in patch A ports are two orders of magnitude larger than landings in patch B, so we set the harvest retention coefficient for patch A to 1.

Table 8. Estimated habitat elasticities for each stock, calculated as the fractional changes in simulated average annual landings assuming a 1% increase in habitat volumes at all times for each species, $\left(\tilde{V}' - \tilde{V}\right)/\tilde{V} = \left(\hat{V}' - \hat{V}\right)/\hat{V} = 1.01$. Species marked with an "*" do not migrate between the Chesapeake Bay and other Atlantic estuaries, so there are no differences between the without and with spillovers cases.

	without spillovers	with spillovers	
	$\frac{Q_B' - Q_B}{Q_B}$	$\frac{Q_A' - Q_A}{Q_A}$	$\frac{Q_B' - Q_B}{Q_B}$
Blue crab	0.019	0.011	0.020
Atl. menhaden	0.014	0.005	0.009
N. quahog clam*	0.011	NA	0.011
Striped bass	0.012	0.004	0.012
Eastern oyster*	0.054	NA	0.048
Atl. croaker	0.013	0.011	0.013
Summer flounder	0.017	0.001	0.004
Spot	0.018	0.010	0.017
Black sea bass	0.016	0.000	0.003
White perch*	0.017	NA	0.017
Blue catfish*	0.017	NA	0.017
Bluefish	0.018	0.003	0.003
Tautog	0.017	0.000	0.007
American shad	0.013	0.000	0.002

Table 9. Fractional changes in simulated average harvest levels for each species under the TMDL scenario relative to the baseline scenario.

	Habitat volume changes		Landings changes				
			without spillovers	with spillovers			
	$\frac{\tilde{V}' - \tilde{V}}{\tilde{V}}$	$\frac{\hat{V}' - \hat{V}}{\hat{V}}$	$\frac{Q'_B - Q_B}{Q_B}$	$\frac{Q'_A - Q_A}{Q_A}$	$\frac{Q'_B - Q_B}{Q_B}$	$\frac{Q_B}{Q}$	$\frac{H_B}{H}$
Blue crab	0.0017	0.0136	0.015	0.009	0.015	0.61	0.61
Atl. menhaden	0.0197	0.0078	0.015	0.005	0.009	0.86	0.81
N. quahog clam*	0.0098	0.0023	0.004	NA	0.004	0.47	0.47
Striped bass	0.0000	0.0118	0.012	0.004	0.012	0.30	0.54
Eastern oyster*	0.0000	0.0058	0.006	NA	0.006	0.45	0.45
Atl. croaker	0.0056	0.0015	0.003	0.003	0.003	0.65	0.69
Sum. Flounder	0.0344	0.0108	0.028	0.002	0.006	0.22	0.19
Spot	0.0422	0.0167	0.045	0.026	0.044	0.71	0.76
Black sea bass	0.0455	0.0175	0.037	0.000	0.006	0.11	0.09
White perch*	0.0158	0.0046	0.014	NA	0.014	0.75	0.75
Blue catfish*	0.0312	0.0122	0.033	NA	0.033	1.00	1.00
Bluefish	0.0175	0.0175	0.031	0.005	0.005	0.06	0.06
Tautog	0.0455	0.0175	0.039	0.000	0.016	0.12	0.12
American shad	0.0534	0.0238	0.027	0.000	0.009	0.02	0.01

Table 10. Ballpark estimates of changes in consumer and producer surplus for commercial fisheries in the Chesapeake Bay under full implementation of the TMDL.

	Baseline prices and quantities			Expert judgments		MSFM w/o spillovers		MSFM w/ spillovers		
	p_0	Q_0^B	Q_0^A	$\Delta Q_0^B / Q_0^B$	$p_0 \Delta Q_0^B$	$\Delta\, Q_0^B / Q_0^B$	$p_0 \Delta Q_0^B$	$\Delta\, Q_0^B / Q_0^B$	$\Delta\, Q_0^A / Q_0^A$	$p_0 \Delta Q_0$
Species	[$\$\cdot\text{lb}^{-1}$]	[$\text{lb}\cdot\text{yr}^{-1}$]	[$\text{lb}\cdot\text{yr}^{-1}$]	[ ]	[$\$\cdot\text{yr}^{-1}$]	[ ]	[$\$\cdot\text{yr}^{-1}$]	[ ]	[ ]	[$\$\cdot\text{yr}^{-1}$]
Blue crab	1.05	78,423,070	50,979,005	0.164	13,502,584	0.015	1,234,992	0.008	0.015	1,643,120
Atl. menhaden	0.07	400,425,892	64,292,431	-0.071	-2,090,538	0.015	441,663	0.005	0.009	289,434
N. quahog clams	7.89	2,487,206	2,877,114	0.079	1,550,323	0.004	78,497	0	0.004	78,497
Striped bass	2.12	4,715,332	2,691,811	0.043	428,926	0.012	119,700	0.004	0.012	144,852
Eastern oyster	5.19	1,430,257	1,627,245	0.164	1,217,554	0.006	44,545	0	0.006	44,545
Atl. croaker	0.77	8,768,857	6,073,955	0.029	197,074	0.003	20,387	0.003	0.003	32,912
S. flounder	1.68	3,098,271	9,176,990	0.043	223,960	0.028	145,834	0.002	0.007	77,192
Spot	0.89	2,595,768	871,120	0.057	132,209	0.044	102,056	0.026	0.044	120,926
Black sea bass	3.34	424,430	1,576,956	0	0	0.036	51,031	0	0.006	8,505
White perch	0.78	1,657,637	387,979	0.029	37,356	0.014	18,034	0	0.014	18,034
Blue catfish	0.35	1,353,192	264	-0.021	-9,938	0.033	15,617	0	0.033	15,617
Bluefish	0.52	548,230	5,510,800	0	0	0.031	8,921	0.005	0.005	15,254
Tautog	2.32	11,681	254,112	0	0	0.041	1,113	0	0.016	434
American shad	0.45	6,801	552,936	0.029	89	0.027	82	0	0.01	31
Sum:					15,189,600		2,282,474			2,489,352

Table 11. Harvest Groupings for inverse demand model. Species marked with a "*" are not a significant component of Chesapeake harvest.

High Value Fish	**Shellfish**	**Low Value Fish**
Black Sea Bass	American Lobster*	Croaker
Cod*	Blue Crab	Herring*
Eels	Clam*	Mackerel*
Flounder	Oyster	Menhaden
Grouper*	Scallop*	Perch
Striped Bass	Shrimp*	Spot

Table 12. Inverse demand model data summary.

	Average Annual Harvest (Tons)	Average Price (2010 Dollars)	Average Annual Value (Millions of 2010 Dollars)
American Lobster	39,353	5.01	394.63
Black Sea Bass	1,648	2.24	7.40
Blue Crab	95,400	0.91	172.87
Clam	26,641	0.70	37.22
Cod	15,782	1.51	47.59
Croaker	9,230	0.54	9.89
Eels	478	3.90	3.73
Flounder	6,374	2.48	31.65
Grouper	3,043	2.53	15.42
Herring	76,200	0.14	20.89
Mackerel	19,172	0.38	14.52
Menhaden	846,000	0.08	139.83
Oyster	11,591	3.45	79.97
Perch	874	0.95	1.66
Scallop	20,238	7.39	299.10
Shrimp	114,000	2.58	589.11
Spot	2,925	0.61	3.60
Striped Bass	2,597	2.37	12.29

Table 13. Own-price flexibilities for high value finfish.

	Own-Quantity Flexibilities			**Standard Deviations**		
	First Stage	Second Stage	Total	First Stage	Second Stage	Total
Flounder	-0.612			0.011		
Chesapeake		-0.4720	-0.3775		0.0209	0.0212
New England		-0.6452	-0.5472		0.0134	0.0135
Mid-Atlantic		-0.6143	-0.5167		0.0144	0.0147
South Atlantic		-0.4949	-0.3970		0.0129	0.0133
Striped Bass	-0.3842			0.0184		
Chesapeake		-0.8186	-0.3683		0.0371	0.0386
New England		-0.4479	-0.2735		0.1726	0.1727
Mid-Atlantic		-0.7519	-0.6620		0.0468	0.0469
South Atlantic		-0.6291	-0.5372		0.0759	0.0759
Black Sea Bass	-0.3351			0.0205		
Chesapeake		-0.6840	-0.5262		0.0155	0.0164
New England		-0.2334	-0.1243		0.0184	0.0188
Mid-Atlantic		-0.5675	-0.3873		0.0191	0.0199
South Atlantic		-0.5976	-0.4269		0.0150	0.0161
Gulf		-0.4068	-0.3546		0.0597	0.0597
Eel	-0.7295			0.0552		
Chesapeake		-0.7141	-0.5306		0.0200	0.0425
Mid-Atlantic		-0.6381	-0.5698		0.0284	0.0321
South Atlantic		-0.5968	-0.5639		0.0309	0.0318
Cod	-0.5880			0.0162		
Grouper	-0.6224			0.0143		

Table 14. Own-price flexibilities for low value finfish.

	Own Quantity Flexibilities			**Standard Errors**		
	First Stage	Second Stage	Total	First Stage	Second Stage	Total
Spot	-0.4124			0.0482		
Chesapeake		-0.7335	-0.3434		0.0083	0.0332
Mid-Atlantic		-0.6017	-0.5881		0.0468	0.0468
South Atlantic		-0.6135	-0.3719		0.0191	0.0283
Gulf		-0.0626	-0.0501		0.1188	0.1167
Perch	-0.7806			0.0759		
Chesapeake		-0.7980	-0.6199		0.0092	0.0175
Mid-Atlantic		-0.2601	-0.2473		0.0229	0.0229
South Atlantic		-0.2063	-0.1784		0.0437	0.0437
Croaker	-0.5963			0.0283		
Chesapeake		-0.9024	-0.6936		0.0092	0.0175
Mid-Atlantic		-0.7064	-0.7064		0.0229	0.0229
Gulf		-0.4376	-0.4017		0.0437	0.0437
Menhaden	-0.7524			0.0081		
Chesapeake		-0.7520	-0.6821		0.0117	0.0119
Mid-Atlantic		-0.1915	-0.1859		0.0930	0.0930
South Atlantic		-0.7055	-0.6991		0.0245	0.0245
Gulf		-0.7941	-0.6278		0.0073	0.0094
Herring	-0.7755			0.225		
Mackerel	-0.7187			0.202		

Table 15 Own-price flexibilities for shellfish

	Own Quantity Flexibilities			**Standard Errors**		
	First Stage	Second Stage	Total	First Stage	Second Stage	Total
Blue crab	-0.4022			0.0402		
Chesapeake		-0.5186	-0.2962		0.0123	0.0191
Mid-Atlantic		-0.2611	-0.2345		0.0364	0.0366
South Atlantic		-0.4831	-0.3171		0.0177	0.0211
Gulf		-0.5945	-0.4072		0.0162	0.0211
Oyster	-0.6780			0.0447		
Chesapeake		-0.2802	-0.2570		0.0236	0.0239
Mid-Atlantic		-0.5734	-0.5645		0.0261	0.0262
South Atlantic		-0.5515	-0.5409		0.0256	0.0256
Gulf		-0.9772	-0.6967		0.0040	0.0392
Clam	-0.1051			0.0833		
Scallop	-0.3461			0.0281		
Shrimp	-0.5741			0.0217		

Lobster	-0.3863	0.0219

Table 16. Expenditure allocation.

	Baseline expenditure shares	Change in expenditure shares based on alternative quantity forecasts		
High Value		Expert panel	MSFM w/o spillovers	MSFM w/ spillovers
Flounder	0.2597	+0.0040	+0.0042	+0.0041
Striped Bass	0.1675	+0.0330	+0.0315	+0.0316
Seabass	0.0565	+0.0062	+0.0066	+0.0065
Eel	0.1146	-0.0583	-0.0585	-0.0584
Grouper*	0.1357	+0.0182	+0.0185	+0.0185
Cod*	0.2660	-0.0032	-0.0023	-0.0023
Low Value				
Spot	0.0192	+0.0006	+0.0004	+0.0003
Perch	0.0107	+0.0183	+0.0176	+0.0176
Croaker	0.0710	-0.0533	-0.0556	-0.0555
Menhaden	0.6997	-0.0713	-0.0670	-0.0672
Herring*	0.1651	-0.0303	-0.0313	-0.0312
Mackerel*	0.0342	+0.1361	+0.1358	+0.1359
Shellfish				
Crab	0.1143	-0.0025	-0.0074	-0.0073
Oyster	0.0591	-0.0182	-0.0180	-0.0179
Clam*	0.0177	-0.0118	-0.0112	-0.0112
Scallop*	0.3088	-0.0702	-0.0685	-0.0685
Shrimp*	0.2572	+0.0439	+0.0451	+0.0451
Lobster*	0.2429	+0.0587	+0.0600	+0.0600

* Species not harvested in the Chesapeake Bay consistently or by a significant amount

Table 17. Annual compensating variation in steady state (millions of 2010 dollars).

	Expert Panel	MSFM w/o spillovers	MSFM w/ spillovers
High Value Fish	0.795	0.317	0.310
Low Value Fish	-2.126	0.651	0.319
Shellfish	14.070	0.128	1.597
Total	12.739	1.096	2.226

Table 18. Summary of Chesapeake Bay TMDL commercial fishery benefit estimates for three potential impact scenarios, based on expert judgments and the multi-species fishery model with and without spillovers, all assuming fixed fishing effort.

	Expert judgments			MSFM w/o spillovers			MSFM w/ spillovers			
	CV	p_1^B	$\Delta(pQ)$	CV	p_1^B	$\Delta(pQ)$	CV	p_1^A	p_1^B	$\Delta(pQ)$
	$[\$\ \mathrm{yr}^{-1}]$	$[\$\ \mathrm{lb}^{-1}]$	$[\$\ \mathrm{yr}^{-1}]$	$[\$\ \mathrm{yr}^{-1}]$	$[\$\ \mathrm{lb}^{-1}]$	$[\$\ \mathrm{yr}^{-1}]$	$[\$\ \mathrm{yr}^{-1}]$	$[\$\ \mathrm{lb}^{-1}]$	$[\$\ \mathrm{lb}^{-1}]$	$[\$\ \mathrm{yr}^{-1}]$
Blue crab	12,403,366	1.019	10,722,542	112,341	1.047	983,879	1,358,672	0.999	1.047	1,297,233
Atl. menhaden	-2,486,786	0.077	-790,109	516,713	0.073	296,085	208,504	0.076	0.073	135,803
N. quahog clams	0	7.890	1,550,283	0	7.890	78,460	0	6.790	7.890	78,585
Striped bass	428,075	2.106	383,454	119,958	2.113	105,642	126,875	2.335	2.113	127,696
Eastern oyster	1,665,258	4.937	795,102	15,151	5.188	39,865	239,364	8.933	5.188	58,856
Atl. croaker	212,719	0.772	168,038	22,061	0.776	26,473	20,655	0.688	0.776	40,509
Summer flounder	237,056	1.670	188,550	149,231	1.682	149,791	159,469	2.220	1.682	90,271
Spot	119,816	0.872	74,393	94,627	0.884	77,146	85,199	0.830	0.884	93,040
Black sea bass	0	3.347	2,943	44,575	3.337	49,771	25,273	2.920	3.337	7,563
White perch	33,829	0.766	19,034	16,337	0.772	9,123	5,780	0.788	0.772	8,310
Blue catfish	0	0.350	-9,939	0	0.350	15,616	0	0.350	0.350	15,616
Bluefish	0	0.525	0	0	0.525	8,920	0	0.501	0.525	15,232
Tautog	0	2.325	1	0	2.325	1,114	0	2.784	2.325	432
American shad	0	0.449	89	0	0.449	82	0	1.006	0.449	29
Sums:	12,613,333		13,104,378	1,090,994		1,841,967	2,229,791			1,969,175
			25,717,711			2,932,961				4,198,966

Table 19. Intercept data summary statistics.

YEAR	2010	2009	2008
Intercepts			
Total	14455	14580	16602
Chesapeake Bay	7074	7395	7937
Per Site: #			
1	783	636	609
2	1736	1412	1418
3	581	593	533
4	285	143	277
5	35	24	25
6	405	235	242
7	542	504	496
8	71	80	94
9	237	128	110
10	61	24	34
11	327	245	305
12	15	9	23
13	24	7	6
14	13	7	25
15	76	34	27
16	35	26	42
17	204	191	172
18	398	430	375
19	27	43	26
20	489	955	541
21	135	102	98
22	209	428	317
23	75	69	37
24	114	93	68
25	40	19	2
26	103	149	236
27	69	29	145
28	15	6	44
29	84	75	80
30	95	59	151
31	65	118	55
32	68	269	240
33	33	27	36
34	469	226	185
35	338	377	257
36	44	59	22
37	1300	1235	1649
38	1538	1278	1039
39	3748	2896	3222
40	437	241	436
41	1279	1099	756

Table 20. Estimated per-person travel costs by year.

Year	2010	2009	2008
TRAVEL COSTS			
Mean	$233.53	$221.02	$218.33
Min	$1.62	$1.57	$1.57
Max	$827.35	$764.74	$800.70

Table 21. Average number of days fished per year.

SOURCE	2010	2009	2008
Days Fished in the last 12 Months Question	19.22	20.60	23.42
Zip Code Intercept Counts			
Using Only Intercepted Zip Codes	5.78	5.82	6.63
Using All Zip Codes Within 150 Miles of the Bay	2.78	2.89	3.11
NOAA Mid-Atlantic Estimated Recreational ‘Total Trips’/’Total Anglers’	5.96	6.49	6.86

Table 22. Catch rate specifications. (Note that the SPECIES INCLUDED lists are not exhaustive for the groupings and only contain the major species.)

SPECIFICATION	SPECIES INCLUDED
1. TOTAL CATCH RATE	All fish caught
2. TOTAL CATCH RATE BY BOAT AND SHORE	All fish caught fishing separated into those caught by boat from shore
3. TOTAL CATCH RATE AND SPECIED SPECIFIC CATCH RATES	All fish caught plus individual catch rates for Atlantic Croaker, Spotted Sea Trout, Black Sea Bass, Striped Bass, Spot, Bluefish, Summer Flounder, White Perch, Tautog
4. TOTAL CATCH RATE BY BOAT AND SHORE AND SPECIED SPECIFIC CATCH RATES	All fish caught seperated by boat and shore catch plus individual catch rates for Atlantic Croaker, Spotted Sea Trout, Black Sea Bass, Striped Bass, Spot, Bluefish, Summer Flounder, White Perch, Tautog
5. FISH RELATED GROUPS	
Small Game	Spotted Seatrout, Striped Bass, Bluefish, Red Drum, …
Bottom Fish	Atlantic Croaker, Black Sea Bass, Spot, White Perch, Tautog, …
Flat Fish	Summer Flounder, other Flounder Species…
6. CHESAPEAKE BAY SPECIFIC GROUPS	
Bay Game Fish	Black Drum, Bluefish, Cobia, Mackerel, Red Drum, Spotted Sea Trout, Striped Bass, Weakfish, …
Bay Bottom Fish	Atlantic Croaker, Black Sea Bass, Kingfish, Sheepshead, Spot, Summer Flounder, Tautog, …
Brackish Water Fish	Shad, Herring, Perch, Sturgeon, Catfish, Bluegill, Large Mouth Bass, Small Mouth Bass, Snakeheads, …

Table 23. Percent of targeted trips in a year targeting a specific species.

	Atlantic Croaker	Spotted Sea Trout	Black Sea Bass	Striped Bass	Spot	Bluefish	Summer Flounder	White Perch	Tautog
2010	16.1%	5.4%	1.0%	33.8%	12.6%	5.9%	18.9%	3.7%	2.5%
2009	14.4%	6.5%	1.4%	34. 5%	8.5%	5.2%	19.9%	4.1%	3.1%
2008	17.0%	6.4%	0.9%	32.1%	12.3%	4.9%	18.1%	4.1%	2.3%
2007	6.2%	2.6%	0.9%	33.9%	6.0%	4.8%	8.4%	4.1%	2.5%
2006	11.2%	1.8%	1.9%	31.0%	7.8%	3.5%	12.3%	5.1%	3.3%

Table 24. Average annual catch rate per hour and per trip for each expected catch metric. (Per trip estimates assume a sample average 4.3 hour trip.)

	2010		**2009**		**2008**	
	CATCH PER HOUR	CATCH PER TRIP	CATCH PER HOUR	CATCH PER TRIP	CATCH PER HOUR	CATCH PER TRIP
TOTAL CATCH						
CATCH RATE	0.892	3.84	0.526	2.26	0.561	2.41
BOAT RATE	0.756	3.25	0.490	2.11	0.489	2.10
SHORE RATE	0.136	0.59	0.036	0.16	0.072	0.31
SPECIES SPECIFIC						
ATLANTIC CROAKER	3.111	13.38	3.294	14.16	2.110	9.07
SPOTTED SEA TROUT	0.190	0.82	0.058	0.25	0.110	0.47
BLACK SEA BASS	0.261	1.12	0.337	1.45	0.164	0.71
STRIPED BASS	0.156	0.67	0.147	0.63	0.226	0.97
SPOT	0.833	3.58	1.076	4.62	1.604	6.90
BLUEFISH	0.236	1.01	0.383	1.65	0.297	1.28
SUMMER FLOUNDER	0.158	0.68	0.198	0.85	0.164	0.70
WHITE PERCH	1.209	5.20	0.834	3.59	1.383	5.95
TAUTOG	0.009	0.04	0.025	0.11	0.055	0.23
MID ATLANTIC GROUPS						
SMALL GAME	0.335	1.44	0.241	1.04	0.486	2.09
BOTTOM FISH	3.411	14.67	3.028	13.02	5.061	21.76
FLATFISH	0.157	0.68	0.128	0.55	0.199	0.85
BAY SPECIFIC FISH GROUPS						
BAY GAME FISH	0.584	2.51	1.116	4.80	0.347	1.49
BAY BOTTOM FISH	1.627	6.99	2.022	8.69	2.400	10.32
BRACKISH FISH	1.290	5.55	1.923	8.27	1.916	8.24

Table 25. Average annual water quality values and correlation among Chesapeake Bay sites. The top number in each cell is the mean value and numbers in parenthesis are the minimum and maximum.

	KD [ug/L]	CL [ug/L]	DO [inverse meters]
2008*	2.2022 (0.9657 to 4.3474)	18.9594 (7.2948 to 71.5200)	7.6115 (4.9433 to 9.6714)
2009	2.1697 (0.9148 to 4.8001)	16.8903 (6.4584 to 61.1540)	7.4292 (6.0233 to 9.0770)
2010	2.5516 (1.0632 to 5.5567)	19.7677 (7.1571 to 51.8600)	7.5275 (5.4086 to 10.0340)
Correlation between Water Quality Measures over Sites and Years			
CL – DO	0.4051		
CL – KD	0.5863		
DO – KD	-0.2704		

Table 26. RUM site choice model results. The first value in each row of the table is the estimated coefficient; values in parenthesis are t-statistics.

	2008		2009		2010	
TRAVEL COST	-3.396	(-40.182)	-3.466	(-33.753)	-3.143	(-40.236)
ASC1	2.249	(12.671)	2.050	(12.055)	1.780	(7.762)
ASC2	1.846	(10.844)	1.572	(9.171)	1.537	(6.175)
ASC3	1.240	(6.476)	1.029	(6.446)	1.071	(3.925)
ASC4	0.674	(2.144)	0.449	(0.303)	0.465	(-0.275)
ASC5	0.085	(0.923)	-0.031	(-0.830)	0.125	(-1.197)
ASC6	1.302	(6.375)	0.847	(3.629)	0.903	(2.549)
ASC7	1.191	(6.648)	1.173	(7.012)	1.216	(3.084)
ASC8	0.587	(1.079)	0.357	(0.956)	0.434	(-1.538)
ASC9	0.252	(0.418)	0.021	(-1.191)	0.130	(-2.082)
ASC10	0.005	(-2.228)	-0.250	(-0.895)	0.055	(-1.584)
ASC11	1.680	(7.448)	1.512	(5.267)	1.734	(7.667)
ASC12	-0.112	(-0.825)	-0.301	(-1.410)	-0.227	(-1.626)
ASC13	0.633	(0.198)	0.107	(-0.521)	0.355	(-0.293)
ASC14	-0.571	(-1.888)	-0.718	(-1.317)	-0.392	(-2.868)
ASC15	-1.206	(-6.784)	-1.383	(-1.265)	-1.105	(-1.913)
ASC16	-2.062	(-6.509)	-2.326	(-6.438)	-2.045	(-6.929)
ASC17	-0.291	(-5.964)	-0.443	(-6.616)	-0.306	(-5.913)
ASC18	-1.064	(-12.429)	-1.275	(-15.050)	-1.054	(-12.254)
ASC19	-3.361	(-8.337)	-3.601	(-9.868)	-3.315	(-8.552)
ASC20	-2.198	(-21.612)	-2.401	(-23.772)	-2.121	(-19.658)
ASC21	-3.375	(-17.772)	-3.522	(-17.516)	-3.236	(-13.949)
ASC22	-2.952	(-19.178)	-3.150	(-27.346)	-2.941	(-22.952)
ASC23	-3.687	(-14.222)	-3.836	(-13.843)	-3.581	(-9.187)
ASC24	-4.321	(-21.666)	-4.397	(-21.475)	-4.154	(-18.273)
ASC25	-4.799	(-14.012)	-4.952	(-12.233)	-4.713	(-2.146)
ASC26	-3.605	(-21.253)	-3.655	(-22.642)	-3.411	(-25.590)
ASC27	-3.673	(-18.007)	-3.749	(-13.944)	-3.574	(-24.786)
ASC28	-2.023	(-2.934)	-2.087	(-1.561)	-1.973	(-8.384)
ASC29	-2.530	(-14.331)	-2.813	(-12.412)	-2.571	(-10.926)
ASC30	-0.787	(-5.222)	-0.868	(-4.611)	-0.786	(-6.846)
ASC31	-0.372	(-3.287)	-0.518	(-4.614)	-0.413	(-2.864)
ASC32	1.636	(5.992)	1.585	(11.645)	1.616	(8.923)
ASC33	1.910	(5.630)	1.836	(4.402)	1.794	(3.136)
ASC34	1.863	(10.546)	1.777	(9.651)	1.728	(4.771)
ASC35	2.360	(14.891)	2.284	(13.832)	2.238	(9.143)
ASC36	1.781	(6.840)	1.670	(5.649)	1.687	(2.223)
ASC37	1.452	(2.936)	1.539	(10.489)	1.591	(11.522)
ASC38	0.528	(-1.418)	0.585	(-2.214)	0.703	(4.777)
ASC39	-0.223	(-8.634)	-0.253	(-8.161)	-0.080	(-4.067)
ASC40	-2.858	(-29.529)	-2.819	(-25.013)	-2.636	(-26.568)
ASC41	0.000	-----	0.000	-----	0.000	-----
Log-Likelihood	33040.40		29262.29		29564.07	

Table 27. Expected total catch models. Shaded cells indicate parameters with the anticipated signs and at least 95% significance levels; values in parenthesis are t-statistics.

	Catch Rate Records Used					
	Current Year		Lagged Year		3 Year Average	
1. Total Catch						
CONSTANT	-1.193	(-1.687)	-1.108	(-1.578)	-1.109	(-1.499)
2009	-0.068	(-0.154)	-0.381	(-0.887)	-0.349	(-0.796)
2008	0.102	(0.234)	-0.423	(-0.973)	-0.282	(-0.643)
NSITES	0.084	(3.083)	0.086	(3.140)	0.090	(3.225)
TOTAL CATCH	0.831	(3.194)	0.911	(3.086)	0.835	(2.376)
KD	-0.591	(-3.527)	-0.530	(-3.085)	-0.579	(-3.340)
ADJUSTED r^2	0.260		0.255		0.227	
2. Total Catch Rate Separated Out by Boat and Shore Catch						
CONSTANT	-1.220	(-1.730)	-1.597	(-2.235)	-1.947	(-2.681)
2009	-0.203	(-0.449)	-0.276	(-0.656)	-0.247	(-0.602)
2008	0.022	(0.050)	-0.230	(-0.533)	0.087	(0.206)
NSITES	0.094	(3.306)	0.105	(3.767)	0.100	(3.817)
BOAT CATCH	0.864	(3.310)	1.061	(3.602)	1.352	(3.800)
SHORE CATCH	-0.740	(-0.563)	-1.060	(-1.226)	-1.833	(-2.371)
KD	-0.557	(-3.286)	-0.442	(-2.573)	-0.337	(-1.938)
ADJUSTED R^2	0.264		0.291		0.322	

Table 28. Expected total catch combined with specific targeted species models.

	Catch Rate Records Used					
	CURRENT YEAR		LAGGED YEAR		3 YEAR AVERAGE	
3. TOTAL CATCH PLUS TARGETED SPECIES						
CONSTANT	-0.800	(-1.481)	-0.612	(-1.096)	-0.468	(-0.870)
2009	-0.259	(-0.802)	-0.419	(-1.322)	-0.388	(-1.359)
2008	-0.113	(-0.352)	-0.332	(-1.006)	0.044	(0.143)
NSITES	0.067	(3.372)	0.070	(3.454)	0.070	(3.771)
TOTALCATCH	0.462	(2.421)	0.453	(1.846)	0.278	(1.072)
ATL. CROAKER	2.933	(3.668)	2.144	(2.629)	5.054	(4.100)
SP. SEA TROUT	4.128	(0.918)	11.862	(2.302)	23.688	(3.362)
BLACK SEA BASS	1.345	(1.881)	1.110	(1.515)	-0.188	(-0.239)
STRIPED BASS	-16.516	(-0.690)	-15.090	(-0.720)	-64.687	(-2.431)
SPOT	0.375	(0.321)	0.423	(0.386)	1.167	(0.974)
BLUEFISH	1.146	(0.377)	2.491	(0.983)	4.317	(1.209)
SUM.FLOUNDER	19.265	(2.484)	8.533	(0.936)	-7.344	(-0.572)
WHITE PERCH	-1.710	(-2.515)	-2.344	(-3.037)	-2.136	(-3.138)
TAUTOG	3.985	(0.427)	10.639	(1.554)	12.212	(1.521)
KD	-0.726	(-5.917)	-0.721	(-5.353)	-0.815	(-6.564)
ADJUSTED r^2	0.642		0.610		0.680	
4. TOTAL CATCH SPLIT BETWEEN BOAT AND SHORE PLUS TARGETED SPECIES						
CONSTANT	-0.877	(-1.676)	-0.962	(-1.673)	-1.021	(-1.931)
2009	-0.461	(-1.436)	-0.353	(-1.127)	-0.331	(-1.230)
2008	-0.221	(-0.708)	-0.221	(-0.673)	0.238	(0.812)
NSITES	0.081	(4.073)	0.083	(3.958)	0.077	(4.417)
BOAT	0.507	(2.739)	0.584	(2.340)	0.661	(2.483)
SHORE	-1.897	(-2.097)	-0.758	(-1.178)	-1.316	(-2.587)
ATL. CROAKER	2.962	(3.831)	1.900	(2.344)	4.471	(3.826)
SP. SEA TROUT	4.498	(1.035)	11.100	(2.187)	21.386	(3.219)
BLACK SEA BASS	1.469	(2.120)	1.189	(1.649)	-0.009	(-0.013)
STRIPED BASS	-13.468	(-0.581)	-13.261	(-0.643)	-60.487	(-2.419)
SPOT	0.475	(0.421)	0.388	(0.361)	1.176	(1.046)
BLUEFISH	0.364	(0.123)	1.811	(0.721)	3.814	(1.137)
SUM. FLOUNDER	19.795	(2.639)	10.041	(1.118)	-5.117	(-0.424)
WHITE PERCH	-1.553	(-2.355)	-2.075	(-2.694)	-1.983	(-3.097)
TAUTOG	3.655	(0.405)	10.982	(1.632)	12.272	(1.629)
KD	-0.675	(-5.612)	-0.658	(-4.840)	-0.656	(-5.262)
ADJUSTED r^2	0.665		0.623		0.718	

Table 29. Expected fish group catch models.

	Catch Rate Records Used					
	CURRENT YEAR		LAGGED YEAR		3 YEAR AVERAGE	
5. MID ATLANTIC GROUPS						
CONSTANT	-1.863	(-3.247)	-1.389	(-2.376)	-1.967	(-3.398)
2009	-0.179	(-0.525)	-0.684	(-1.959)	-0.440	(-1.453)
2008	-0.388	(-1.142)	-0.585	(-1.678)	-0.126	(-0.416)
NSITES	0.108	(4.898)	0.081	(3.680)	0.085	(4.347)
SMALL GAME	0.107	(0.347)	0.114	(0.389)	0.020	(0.046)
BOTTOM FISH	0.024	(0.878)	0.004	(0.141)	-0.001	(-0.035)
FLATFISH	4.713	(8.415)	4.676	(8.187)	6.478	(10.543)
KD	-0.498	(-3.648)	-0.455	(-3.214)	-0.411	(-3.110)
ADJUSTED r^2	0.538		0.523		0.630	
6. BAY SPECIFIC FISH GROUPS						
CONSTANT	0.085	0.130	0.248	0.365	0.429	0.630
2009	-0.264	-0.649	-0.465	-1.136	-0.469	-1.224
2008	-0.132	-0.326	-0.458	-1.114	-0.475	-1.226
NSITES	0.077	2.910	0.079	2.988	0.073	2.861
GAME FISH	-0.050	-0.527	-0.023	-0.247	-0.085	-0.453
BOTTOM FISH	0.080	1.668	0.077	1.656	0.101	1.771
BRACKISH FISH	-0.234	-4.683	-0.237	-4.647	-0.371	-5.681
KD	-0.682	-4.252	-0.706	-4.316	-0.684	-4.256
ADJUSTED r^2	0.346		0.341		0.408	

Table 30. Anticipated catch rate and water quality percentage improvements due to the TMDL.

		MSFM Fractional Stock and Harvest Changes	
	Expert Panel Predictions	Chesapeake Bay	Other Atlantic
TOTAL CATCH MODEL (EC1 & 2)			
CATCH RATE	2.69%	1.46%	0.60%
SPECIES SPECIFIC MODEL (EC3 &4)			
ATLANTIC CROAKER	2.90%	0.30%	0.20%
SPOTTED SEA TROUT	0.00%	0.00%	0.00%
BLACK SEA BASS	0.00%	0.60%	0.00%
STRIPED BASS	4.30%	1.20%	0.40%
SPOT	5.70%	4.40%	2.60%
BLUEFISH	0.00%	1.50%	1.50%
SUMMER FLOUNDER	4.30%	0.70%	0.20%
WHITE PERCH	2.90%	1.40%	0.00%
TAUTOG	0.00%	1.60%	0.00%
MID ATLANTIC GROUPS MODEL (EC5)			
SMALL GAME	2.50%	1.23%	0.60%
BOTTOM FISH	2.60%	1.66%	0.56%
FLATFISH	4.30%	0.70%	0.20%
BAY SPECIFIC FISH GROUPS MODEL (EC6)			
BAY GAME FISH	2.20%	1.35%	0.95%
BAY BOTTOM FISH	2.60%	1.52%	0.60%
BRACKISH FISH	2.90%	1.90%	0.00%
CHANGE IN WATER QUALITY FROM CBPO WATER QUALITY MODEL			
KD	11.0%		

Table 31. Per person per trip WTP for expert panel and MSFM predicted changes in catch rates and the CBEM predicted changes in water quality inside Chesapeake Bay. Shaded rows contains average of the three model specifications for a given year and the minimum and maximum values across all specifications in that year.

	2008		2009		2010	
	MEAN	95% CI	MEAN	95% CI	MEAN	95% CI
EXPERT PANEL PREDICTED CHANGES IN CATCH RATES AND CBEM PREDICTED KD CHANGES						
TOTAL CATCH RATE (EC1)						
CURRENT YEAR	$1.83	$0.96 to $2.51	$1.57	$0.87 to $2.08	$2.19	$1.24 to $2.95
LAGGED YEAR	$1.70	$0.83 to $2.34	$1.37	$0.65 to $1.82	$1.97	$1.00 to $2.69
THREE YEAR AVG	$1.82	$0.95 to $2.48	$1.50	$0.78 to $1.99	$2.10	$1.14 to $2.83
MEAN*	$1.78	$0.83 to $2.51	$1.48	$.065 to $2.08	$2.09	$1.00 to $2.95
TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT (EC2)						
CURRENT YEAR	$1.69	$0.84 to $2.47	$1.47	$0.80 to $2.05	$1.95	$1.03 to $2.85
LAGGED YEAR	$1.42	$0.55 to $2.07	$1.13	$0.40 to $1.60	$1.67	$0.68 to $2.39
THREE YEAR AVG	$1.12	$0.21 to $1.78	$0.97	$0.24 to $1.48	$1.29	$0.22 to $2.02
MEAN	$1.41	$0.21 to $2.47	$1.19	$0.24 to $2.05	$1.64	$0.22 to $2.85
TOTAL CATCH RATE AND SPECIES SPECIFIC CATCH RATES (EC3)						
CURRENT YEAR	$2.77	$2.11 to $3.75	$2.27	$1.79 to $3.06	$3.16	$2.47 to $4.26
LAGGED YEAR	$2.36	$1.60 to $3.27	$1.96	$1.44 to $2.67	$2.76	$1.79 to $3.70
THREE YEAR AVG	$2.35	$1.61 to $3.20	$2.02	$1.44 to $2.68	$2.98	$2.09 to $3.99
MEAN	$2.49	$1.60 to $3.75	$2.08	$1.44 to $3.06	$2.97	$1.79 to $4.26
TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT AND SPECIES SPECIFIC CATCH RATES (EC4)						
CURRENT YEAR	$2.59	$1.92 to $3.60	$2.16	$1.67 to $2.93	$2.84	$2.13 to $4.03
LAGGED YEAR	$2.17	$1.41 to $3.08	$1.82	$1.32 to $2.52	$2.56	$1.60 to $3.50
THREE YEAR AVG	$1.88	$1.12 to $2.78	$1.66	$1.12 to $2.38	$2.41	$1.52 to $3.50
MEAN	$2.21	$1.12 to $3.60	$1.88	$1.12 to $2.93	$2.60	$1.52 to $4.03
MSFM PREDICTED CHANGES IN CATCH RATES AND CBEM PREDICTED KD CHANGES						
TOTAL CATCH RATE (EC1)						
CURRENT YEAR	$1.76	$0.87 to $2.42	$1.47	$0.74 to $1.95	$2.06	$1.09 to $2.80
LAGGED YEAR	$1.60	$0.72 to $2.24	$1.29	$0.57 to $1.74	$1.85	$0.86 to $2.54
THREE YEAR AVG	$1.74	$0.85 to $2.39	$1.42	$0.68 to $1.88	$2.00	$1.02 to $2.70
MEAN	$1.70	$0.72 to $2.42	$1.39	$0.57 to $1.95	$1.97	$0.86 to $2.80
TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT (EC2)						
CURRENT YEAR	$1.63	$0.75 to $2.38	$1.37	$0.65 to $1.93	$1.88	$0.92 to $2.72
LAGGED YEAR	$1.33	$0.44 to $1.96	$1.06	$0.33 to $1.52	$1.55	$0.54 to $2.25
THREE YEAR AVG	$1.03	$0.10 to $1.66	$0.87	$0.12 to $1.36	$1.18	$0.10 to $1.89

MEAN	$1.33	$0.10 to $2.38	$1.10	$0.12 to $1.93	$1.54	$0.10 to $2.72
TOTAL CATCH RATE AND SPECIES SPECIFIC CATCH RATES (EC3)						
CURRENT YEAR	$2.30	$1.29 to $3.07	$1.87	$1.07 to $2.42	$2.56	$1.47 to $3.38
LAGGED YEAR	$2.28	$1.18 to $3.13	$1.85	$0.98 to $2.42	$2.56	$1.30 to $3.42
THREE YEAR AVG	$2.51	$1.35 to $3.39	$2.08	$1.15 to $2.68	$2.85	$1.56 to $3.77
MEAN	$2.36	$1.18 to $3.39	$1.93	$0.98 to $2.68	$2.66	$1.30 to $3.77
TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT AND SPECIES SPECIFIC CATCH RATES (EC4)						
CURRENT YEAR	$2.11	$1.09 to $2.88	$1.74	$0.92 to $2.35	$2.29	$1.16 to $3.18
LAGGED YEAR	$2.08	$0.95 to $2.95	$1.69	$0.79 to $2.28	$2.35	$1.09 to $3.19
THREE YEAR AVG	$2.03	$0.82 to $2.95	$1.70	$0.74 to $2.37	$2.29	$0.94 to $3.28
MEAN	$2.07	$0.82 to $2.95	$1.71	$0.74 to $2.37	$2.31	$0.94 to $3.28

Table 32. Per person per trip WTP for the MSFM predicted changes in catch rates outside Chesapeake Bay. Shaded rows contains average of the three model specifications for a given year and the minimum and maximum values across all specifications in that year.

	2008		2009		2010	
	MEAN	95% CI	MEAN	95% CI	MEAN	95% CI
TOTAL CATCH RATE (1)						
CURRENT YEAR	$0.04	$0.04 to $0.05	$0.05	$0.05 to $0.07	$0.06	$0.06 to $.09
LAGGED YEAR	$0.05	$0.04 to $0.06	$0.04	$0.03 to $0.05	$0.06	$0.04 to $0.08
THREE YEAR AVG	$0.04	$0.04 to $0.06	$0.04	$0.04 to $0.06	$0.05	$0.04 to $.07
MEAN	$0.04	$0.04 to $0.06	$0.04	$0.03 to $0.07	$0.06	$0.04 to $0.09
TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT (2)						
CURRENT YEAR	$0.03	$0.03 to $0.05	$0.04	$0.04 to $0.07	$0.03	$0.03 to $0.08
LAGGED YEAR	$0.04	$0.03 to $0.06	$0.03	$0.02 to $0.04	$0.06	$0.04 to $0.08
THREE YEAR AVG	$0.04	$0.04 to $0.07	$0.05	$0.05 to $0.07	$0.05	$0.04 to $0.07
MEAN	$0.04	$0.03 to $0.07	$0.04	$0.02 to $0.07	$0.05	$0.03 to $0.08
TOTAL CATCH RATE AND SPECIES SPECIFIC CATCH RATES (3)						
CURRENT YEAR	$0.08	$0.05 to $0.15	$0.09	$0.05 to $0.16	$0.10	$0.07 to $0.15
LAGGED YEAR	$0.08	$0.05 to $0.11	$0.09	$0.04 to $0.12	$0.08	$0.04 to $0.11
THREE YEAR AVG	$0.10	$0.05 to $0.17	$0.13	$0.08 to $0.20	$0.12	$0.08 to $0.18
MEAN	$0.09	$0.05 to $0.17	$0.10	$0.04 to $0.20	$0.10	$0.04 to $0.18
TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT AND SPECIES SPECIFIC CATCH RATES (4)						
CURRENT YEAR	$0.04	$0.04 to $0.13	$0.05	$0.05 to $0.16	$0.06	$0.03 to $0.12
LAGGED YEAR	$0.05	$0.04 to $0.10	$0.04	$0.03 to $0.10	$0.06	$0.04 to $0.11
THREE YEAR AVG	$0.04	$0.05 to $0.17	$0.04	$0.08 to $0.20	$0.05	$0.08 to $0.18
MEAN	$0.04	$0.04 to $0.17	$0.04	$0.03 to $0.20	$0.06	$0.03 to $0.18

Table 33. Negative binomial trip model results.

DAYS FISHED IN THE LAST 12 MONTHS	2008		2009		2010	
CONSTANT	2.7867	(51.897)	3.0914	(71.243)	2.9232	(60.571)
INCOME	0.0099	(5.404)	0.0000	(0.002)	0.0031	(2.051)
RUM IV INDEX	0.8456	(49.449)	0.7989	(44.651)	0.7090	(32.662)
ALPHA	1.7555	(65.017)	1.9920	(63.642)	2.0698	(60.684)
Log Likelihood	-30740.45		-27014.91		-25875.41	
N	7663		7098		6834	
ZIP CODE TRIPS	2008		2009		2010	
CONSTANT	1.9544	(17.960)	1.8096	(9.875)	1.3236	(7.464)
INCOME	0.0155	(4.384)	0.0205	(2.978)	0.0241	(3.922)
POPULATION	0.0453	(13.558)	0.0315	(10.248)	0.0385	(11.853)
RUM IV INDEX	2.9318	(44.303)	2.3599	(25.476)	2.1669	(30.554)
ALPHA	4.2525	(21.294)	3.8920	(21.686)	3.3646	(20.892)
Log Likelihood	-3632.26		-3449.41		-3490.33	
N	2435		2435		2435	

Table 34. Predicted percent change in trips using the expert panel predicted changes in catch rates and the CBEM predicted changes in water quality inside Chesapeake Bay.

	2008		2009		2010	
TRIP COUNT TYPE	DAYS FISHED	ZIP CODE	DAYS FISHED	ZIP CODE	DAYS FISHED	ZIP CODE
EXPERT PANEL PREDICTED CHANGES IN CATCH RATES AND CBEM PREDICTED KD CHANGES						
1. TOTAL CATCH RATE						
CURRENT YEAR	2.23%	3.32%	1.69%	3.54%	2.20%	4.11%
LAGGED YEAR	2.08%	3.06%	1.47%	3.10%	1.99%	3.66%
THREE YEAR AVG	2.22%	3.31%	1.61%	3.41%	2.11%	3.94%
MEAN	2.18%	3.23%	1.59%	3.35%	2.10%	3.90%
2. TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT						
CURRENT YEAR	2.06%	3.06%	1.58%	3.31%	1.94%	3.67%
LAGGED YEAR	1.76%	2.53%	1.21%	2.55%	1.71%	3.09%
THREE YEAR AVG	1.40%	1.99%	1.07%	2.18%	1.31%	2.39%
MEAN	1.74%	2.53%	1.29%	2.68%	1.65%	3.05%
3. TOTAL CATCH RATE AND SPECIES SPECIFIC CATCH RATES						
CURRENT YEAR	3.57%	4.77%	2.59%	5.01%	3.41%	5.70%
LAGGED YEAR	2.98%	4.20%	2.20%	4.38%	3.41%	5.10%
THREE YEAR AVG	2.89%	4.28%	2.22%	4.62%	3.41%	5.56%
MEAN	3.15%	4.42%	2.34%	4.67%	3.41%	5.45%
4. TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT AND SPECIES SPECIFIC CATCH RATES						
CURRENT YEAR	3.37%	4.43%	2.47%	4.73%	3.05%	5.07%
LAGGED YEAR	2.76%	3.83%	2.05%	4.02%	2.67%	4.69%
THREE YEAR AVG	2.34%	3.36%	1.86%	3.75%	2.52%	4.44%
MEAN	2.82%	3.87%	2.13%	4.17%	2.75%	4.73%
MSFM PREDICTED CHANGES IN CATCH RATES AND CBEM PREDICTED KD CHANGES						
1. TOTAL CATCH RATE						
CURRENT YEAR	2.13%	3.18%	1.57%	3.32%	2.06%	3.88%
LAGGED YEAR	1.95%	2.88%	1.38%	2.93%	1.85%	3.44%
THREE YEAR AVG	2.11%	3.14%	1.51%	3.22%	1.99%	3.75%
MEAN	2.06%	3.07%	1.49%	3.16%	1.97%	3.69%
2. TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT						
CURRENT YEAR	1.98%	2.95%	1.47%	3.10%	1.86%	3.53%
LAGGED YEAR	1.63%	2.37%	1.14%	2.40%	1.56%	2.87%
THREE YEAR AVG	1.27%	1.82%	0.94%	1.94%	1.19%	2.19%

MEAN	1.63%	2.38%	1.18%	2.48%	1.54%	2.86%
3. TOTAL CATCH RATE AND SPECIES SPECIFIC CATCH RATES						
CURRENT YEAR	2.82%	4.11%	2.02%	4.22%	2.59%	4.78%
LAGGED YEAR	2.82%	4.07%	2.01%	4.17%	2.58%	4.80%
THREE YEAR AVG	3.06%	4.57%	2.24%	4.75%	2.83%	5.39%
MEAN	2.90%	4.25%	2.09%	4.38%	2.67%	4.99%
4. TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT AND SPECIES SPECIFIC CATCH RATES						
CURRENT YEAR	2.60%	3.76%	1.88%	3.92%	2.30%	4.26%
LAGGED YEAR	2.59%	3.68%	1.84%	3.78%	2.37%	4.37%
THREE YEAR AVG	2.50%	3.62%	1.86%	3.84%	2.30%	4.28%
MEAN	2.56%	3.69%	1.86%	3.85%	2.32%	4.30%

Table 35. Predicted percent change in trips due to the MSFM predicted changes in catch rates outside Chesapeake Bay.

	2008		2009		2010	
TRIP COUNT TYPE	DAYS FISHED	ZIP CODE	DAYS FISHED	ZIP CODE	DAYS FISHED	ZIP CODE
1. TOTAL CATCH RATE						
CURRENT YEAR	0.05%	0.06%	0.06%	0.10%	0.07%	0.10%
LAGGED YEAR	0.06%	0.08%	0.04%	0.08%	0.07%	0.10%
THREE YEAR AVG	0.05%	0.07%	0.05%	0.09%	0.06%	0.09%
MEAN	0.05%	0.07%	0.05%	0.09%	0.07%	0.10%
2. TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT						
CURRENT YEAR	0.04%	0.05%	0.05%	0.09%	0.04%	0.06%
LAGGED YEAR	0.06%	0.07%	0.04%	0.07%	0.07%	0.10%
THREE YEAR AVG	0.06%	0.08%	0.06%	0.11%	0.06%	0.09%
MEAN	0.05%	0.07%	0.05%	0.09%	0.06%	0.08%
3. TOTAL CATCH RATE AND SPECIES SPECIFIC CATCH RATES						
CURRENT YEAR	0.12%	0.13%	0.10%	0.17%	0.12%	0.15%
LAGGED YEAR	0.11%	0.14%	0.11%	0.18%	0.09%	0.14%
THREE YEAR AVG	0.13%	0.18%	0.15%	0.28%	0.14%	0.21%
MEAN	0.12%	0.15%	0.12%	0.21%	0.12%	0.17%
4. TOTAL CATCH RATE WITH BOAT AND SHORE SPLIT AND SPECIES SPECIFIC CATCH RATES						
CURRENT YEAR	0.10%	0.11%	0.10%	0.17%	0.07%	0.08%
LAGGED YEAR	0.10%	0.12%	0.09%	0.16%	0.09%	0.14%
THREE YEAR AVG	0.13%	0.18%	0.16%	0.29%	0.13%	0.21%
MEAN	0.11%	0.14%	0.12%	0.21%	0.10%	0.14%

Table 36. Number of trips taken within and outside the Chesapeake Bay.

	2008		2009		2010	
Annual Trips						
Chesapeake Bay	6,098,868		5,511,035		5,061,204	
Outside the Chesapeake Bay	29,467,084		22,668,030		23,455,354	
	2008		2009		2010	
Predicted change in trips	Days Fished	Zip Code	Days Fished	Zip Code	Days Fished	Zip Code
Expert Panel, In Ch. Bay	150,795	214,223	101,265	204,873	125,391	216,746
MSFM - In Ch. Bay	139,512	204,160	91,208	191,095	107,551	200,424
MSFM - outside of Ch. Bay	24,310	31,677	19,268	34,002	20,523	28,733

Table 37. Populations WTP from existing trips.

EXPERT PANEL CATCH PREDICTIONS

MODEL	2008	2009	2010
1	$10,855,985 $5,062,060 - $15,308,159	$8,156,332 $3,582,173 - $11,462,953	$10,577,916 $5,061,204 - $14,930,552
2	$8,599,404 $1,280,762 - $15,064,204	$6,558,132 $1,322,648 - $11,297,622	$8,300,375 $1,113,465 - $14,424,431
3	$15,186,181 $9,758,189 - $22,870,755	$11,462,953 $7,935,890 - $16,863,767	$15,031,776 $9,059,555 - $21,560,729
4	$13,478,498 $6,830,732 - $21,955,925	$10,360,746 $6,172,359 - $16,147,333	$13,159,130 $7,693,030 - $20,396,652

MSFM MODEL INSIDE THE CHESAPEAKE BAY CATCH PREDICTIONS

MODEL	2008	2009	2010
1	$10,368,075.6000 $4,391,184.9600 - $14,759,260.5600	$7,660,338.6500 $3,141,289.9500 - $10,746,518.2500	$9,970,571.8800 $4,352,635.4400 - $14,171,371.2000
2	$8,111,494.4400 $609,886.8000 - $14,515,305.8400	$6,062,138.5000 $661,324.2000 - $10,636,297.5500	$7,794,254.1600 $506,120.4000 - $13,766,474.8800
3	$14,393,328.4800 $7,196,664.2400 - $20,675,162.5200	$10,636,297.5500 $5,400,814.3000 - $14,769,573.8000	$13,462,802.6400 $6,579,565.2000 - $19,080,739.0800
4	$12,624,656.7600 $5,001,071.7600 - $17,991,660.6000	$9,423,869.8500 $4,078,165.9000 - $13,061,152.9500	$11,691,381.2400 $4,757,531.7600$16,600,749.1200

MSFM MODEL OUTSIDE THE CHESAPEAKE BAY CATCH PREDICTIONS

MODEL	2008	2009	2010
1	$1,178,683 $1,178,683 - $1,768,025	$906,721 $680,041 - $1,586,762	$1,407,321 $938,214 - $2,110,982
2	$1,178,683 $884,013 - $2,062,696	$906,721 $453,361 - $1,586,762	$1,172,768 $703,661 - $1,876,428
3	$2,652,038	$2,266,803	$2,345,535

4	$1,473,354 - $5,009,404	$906,721 - $4,533,606	$938,214 - $4,221,964
	$1,178,683	$906,721	$1,407,321
	$1,178,683 - $5,009,404	$680,041 - $4,533,606	$703,661 - $4,221,964

Table 38. Population WTP from new trips.

	DAYS FISHED TRIP COUNTS			ZIP CODE TRIP COUNTS		
EXPERT PANEL INSIDE THE CHESAPEAKE BAY CATCH PREDICTIONS						
MODEL	2008	2009	2010	2008	2009	2010
1	$268,414	$149,873	$262,068	$381,316	$303,212	$452,999
	$125,159 - $378,494	$65822 - $210,632	$125,391 - $369,904	$177,805 - $537,699	$133,167 - $426,135	$216,746 - $639,401
2	$212,620	$120,506	$205,642	$302,054	$243,799	$355,464
	$31,667 - $372,462	$24304 - $207,594	$27,586 - $357,365	$44,987 - $529,130	$49,169 - $419,989	$47,684 - $617,726
3	$375,478	$210,632	$372,412	$533,415	$426,135	$643,736
	$241,271 - $565,479	$145822 - $309,872	$224,450 - $534,167	$342,756 - $803,335	$295,017 - $626,911	$387,975 - $923,338
4	$333,256	$190,379	$326,017	$473,432	$385,161	$563,540
	$168,890 - $542,860	$113,417 - $296,707	$190,595 - $505,327	$239,929 - $771,202	$229,457 - $600,277	$329,454 - $873,487
MSFM MODEL INSIDE THE CHESAPEAKE BAY CATCH PREDICTIONS						
MODEL	2008	2009	2010	2008	2009	2010
1	$237,170	$126,779	$211,875	$347,071	$265,622	$394,835
	$100,448 - $337,618	$51,988 - $177,855	$92,494 - $301,142	$146,995 - $494,066	$108,924 - $372,636	$172,364 - $606,889
2	$185,550	$100,328	$165,628	$271,532	$210,205	$308,652
	$13,951 - $332,038	$10,945 - $176,031	$10,755 - $292,538	$20,416 - $485,900	$22,931 - $368,814	$20,042 - $589,549
3	$329,247	$176,031	$286,085	$481,817	$368,814	$533,127
	$164,624 - $472,944	$89,383 - $244,436	$139,816 - $405,466	$240,908 - $692,101	$187,273 - $512,135	$260,551 - $817,133
4	$288,789	$155,965	$248,442	$422,610	$326,773	$462,979
	$114,400 - $411,559	$67,494 - $216,162	$101,098 - $352,766	$167,411 - $602,271	$141,410 - $452,895	$188,398 - $710,927
MSFM MODEL OUTSIDE THE CHESAPEAKE BAY CATCH PREDICTIONS						
MODEL	2008	2009	2010	2008	2009	2010
1	$972	$771	$1,231	$1,267	$1,360	$1,724
	$972 - $1,459	$578 - $1,349	$821 - $1,847	$1,267 - $1,901	$1,020 - $2,380	$1,149 - $2,586
2	$972	$771	$1,026	$1,267	$1,360	$1,437
	$729 - $1,702	$385 - $1,349	$616 - $1,642	$950 - $2,217	$680 - $2,380	$862 - $,2299

3	$2,188	$1,927	$2,052	$2,851	$3,400	$2,873
	$1,216 - $4,133	$771 - $3,854	$821 - $3,694	$1,584 - $5,385	$1,360 - $6,800	$1,149 - $5,172
4	$972	$771	$1,231	$1,267	$1,360	$1,724
	$972 - $4,133	$578 - $3,854	$616 - $3,694	$1,267 - $5,385	$1,020 - $6,800	$862 - $5,172

Table 39. Total population WTP.

	LOW PARTICIPATION RATE ESTIMATES			HIGH PARTICIPATION RATE ESTIMATES		
EXPERT PANEL CATCH PREDICTIONS						
MODEL	2008	2009	2010	2008	2009	2010
1	$12,304,055	$9,213,696	$12,248,537	$12,417,252	$9,367,625	$12,439,961
	$6,366,876 - $17,456,137	$4,328,614 - $13,261,695	$6,125,630 - $17,413,285	$6,419,816 - $1,7615,783	$4,396,401 - $13,478,230	$6,217,314 - $17,683,520
2	$9,991,680	$7,586,129	$9,679,810	$10,081,408	$7,710,011	$9,830,042
	$2,197,171 - $17,501,064	$1800698 - $13093326	$1,845,327 - $16,659,867	$2,210,712 - $17,658,247	$1,825,858 - $13,306,753	$1,865,672 - $16,920,885
3	$18,215,885	$13,942,314	$17,751,776	$18,374,484	$14,159,291	$18,023,920
	$11,474,030 - $28,449,771	$8,989,204 - $21,711,098	$10,223,041 - $26,320,554	$11,575,883 - $28,688,880	$9,138,988 - $22,031,084	$10,386,894 - $26,711,203
4	$14,991,410	$11,458,616	$14,893,701	$15,131,881	$11,653,988	$15,131,715
	$8,179,278 - $27,512,322	$6,966,395 - $20,981,499	$8,587,901 - $25,127,637	$8,250,612 - $27,741,916	$7,082,878 - $21,288,016	$8,727,007 - $25,497,274
MSFM CATCH PREDICTIONS						
1	$12,304,055	$9,213,696	$12,248,537	$12,417,252	$9,367,625	$12,439,961
	$5,671,289 - $16,866,362	$3,873,897 - $12,512,484	$5,384,164 - $16,585,342	$5,718,130 - $17,023,252	$3,931,275 - $12,708,296	$5,464,363 - $16,891,828
2	$9,991,680	$7,586,129	$9,679,810	$10,081,408	$7,710,011	$9,830,042
	$,1508,580 - $16,911,741	$1,126,015 - $12,400,439	$1,221,152 - $1,5937,083	$1,515,266 - $17,066,119	$1,138,296 - $12,594,253	$1,230,685 - $16,234,751
3	$18,215,885	$13,942,314	$17,751,776	$18,374,484	$14,159,291	$18,023,920
	$8,835,858 - $26,161,644	$6,397,690 - $19,551,470	$7,658,416 - $23,711,863	$8,912,511 - $26,382,053	$6,496,169 - $19,822,115	$7,779,479 - $24,125,007
4	$14,093,102	$10,487,327	$13,348,376	$14,227,218	$10,658,724	$13,563,405
	$6,295,127 - $23,416,757	$4,826,278 - $17,814,775	$5,562,906 - $21,179,173	$6,348,433 - $23,608,721	$4,900,637 - $18,054,455	$5,650,453 - $21,538,812

Table 40. Outdoor recreation site annual visits and attribute summary statistics. N(Y) is the number of years of visitation data available for the site. Mean(.) and CV(.) refer to averages and coefficients of variation of total annual visits to each site across years. The final six columns contain the water quality measures that were used to estimate the model and calculate the outdoor recreation benefits of the TMDL. "CL baseline" and "CL TMDL" are the average predicted May-September chlorophyll-a concentrations [µg/L] at each site over the 10 years simulated by the Chesapeake Bay Estuary Model (CBEM), 1991-2000. The final four columns in the table contain analogous data on dissolved oxygen (DO) [µg/L] and Secchi depth (SD) [ft].

Park name	N(Y)	mean(Y)	CV(Y)	Water access	Boat	Fish	Swim	Trail	Restroom	Parking	CL baseline	CL TMDL	DO baseline	DO TMDL	SD baseline	SD TMDL
Antietam National Battlefield, MD	10	3.17E+05	0.153	0												
Assateague Island Ntl Seashore, MD	10	2.04E+06	0.039	0												
Ches & Ohio Canal Ntl Hist Park, MD	10	3.29E+06	0.159	0												
Catoctin Mountain Park, MD	10	5.24E+05	0.178	0												
Clara Barton Ntl Historic Site, MD	10	1.43E+04	0.309	0												
Fort McHenry Ntl Monument, MD	10	6.19E+05	0.044	1	0	0	0	1	1	1	23.30	17.89	9.05	8.80	0.93	1.07
Fort Washington National Park, MD	10	3.20E+05	0.209	1	0	0	0	1	1	1	20.92	14.41	8.31	8.42	0.98	1.21
Greenbelt Park, MD	10	2.12E+05	0.299	0												
Hampton National Historic Site, MD	10	2.99E+04	0.157	0												
Monocacy National Battlefield, MD	10	2.24E+04	0.340	0												
Piscataway Park, MD	10	2.22E+05	0.204	1	1	1	0	1	1	1	15.01	14.19	8.53	8.58	0.76	0.89
Thomas Stone Ntl Historic Site, MD	10	5.32E+03	0.115	0												
Appomattox Ct House Ntl Hist Pk, VA	10	1.69E+05	0.147	0												
Arlington House Robert E. Lee Mem, VA	10	4.94E+05	0.158	0												
Blue Ridge Parkway, NC	10	1.79E+07	0.113	0												
Booker T. Washington Ntl Mnmt, VA	10	1.95E+04	0.098	0												
Colonial National Historical Park, VA	10	3.34E+06	0.014	1	0	1	0	1	1	1	10.69	9.38	7.46	7.57	0.76	0.90
Cumberland Gap Ntl Hist Park, VA	10	9.30E+05	0.057	0												
Fredericksbg & Spotsylvania Ntl Mlt Pk, VA	10	5.60E+05	0.326	1	0	0	0	0	1	1	9.29	8.85	8.09	8.16	0.81	0.89
G Washington Birthplace Ntl Mnmt, VA	10	1.07E+05	0.273	1	0	0	0	1	1	1	12.49	10.35	7.73	7.81	1.40	1.55
George Washington Mem Parkway, VA	10	7.02E+06	0.062	0												
LBJ Mem Grove on the Potomac, VA	10	4.26E+05	0.202	0												
Maggie L. Walker Ntl Historic Site, VA	10	9.82E+03	0.196	0												

Manassas National Battlefield Park, VA	10	6.85E+05	0.129	0												
Petersburg National Battlefield, VA	10	1.59E+05	0.056	0												
Prince William Forest Park, VA	10	2.74E+05	0.254	0												
Richmond National Battlefield, VA	10	1.12E+05	0.213	0												
Shenandoah National Park, VA	10	1.20E+06	0.120	0												
Theodore Roosevelt Island, DC	10	1.37E+05	0.233	1	0	1	0	1	1	1	17.74	13.55	8.02	8.14	1.10	1.29
Wolf Trap Ntl Pk for Performing Arts, VA	10	5.23E+05	0.065	0												
Bear Creek Lake State Park, VA	5	6.68E+04	0.189	0												
Belle Isle State Park, VA	5	3.80E+04	0.108	1	1	1	0	1	1	1	7.29	5.75	6.32	6.39	0.83	0.90
Caledon Natural Area State Park, VA	5	3.77E+04	0.209	1	0	0	0	1	1	1	13.24	11.52	7.86	7.95	1.13	1.33
Chippokes Plantation State Park, VA	5	8.41E+04	0.239	1	0	1	0	1	1	1	10.20	8.92	7.45	7.56	0.77	0.91
Claytor Lake State Park, VA	5	2.42E+05	0.116	0												
Douthat State Park, VA	5	2.00E+05	0.084	0												
Fairy Stone State Park, VA	5	1.13E+05	0.273	0												
False Cape State Park, VA	5	3.20E+04	0.156	0												
First Landing State Park, VA	5	1.72E+06	0.017	1	1	1	1	1	1	1	NA	NA	NA	NA	NA	NA
Grayson Highlands State Park, VA	5	1.28E+05	0.060	0												
Hemlock Haven Conference Center, VA	3	2.24E+04	0.044	0												
High Bridge Trail, VA	3	4.45E+04	0.779	0												
Holliday Lake State Park, VA	5	4.53E+04	0.108	0												
Hungry Mother State Park, VA	5	2.41E+05	0.135	0												
James River State Park, VA	5	4.90E+04	0.205	0												
Kiptopeke State Park, VA	5	3.71E+05	0.171	1	1	1	1	1	1	1	4.85	4.26	8.04	7.97	2.00	2.05
Lake Anna State Park, VA	5	2.43E+05	0.230	1	1	0	0	0	0	0	9.55	9.08	8.08	8.16	0.83	0.92
Leesylvania State Park, VA	5	3.37E+05	0.304	1	1	1	0	1	1	1	20.07	16.41	8.27	8.29	1.01	1.17
Mason Neck State Park, VA	5	8.73E+04	0.098	1	0	1	0	1	1	1	22.37	17.82	8.55	8.52	0.92	1.05
Natural Tunnel State Park, VA	5	2.03E+05	0.349	0												
New River Trail State Park, VA	5	9.66E+05	0.043	0												
Occoneechee State Park, VA	5	1.98E+05	0.020	0												
Pocahontas State Park	5	3.51E+05	0.119	0												
Sailors Creek Battlefield Hist State Pk, VA	5	2.38E+04	0.124	0												
Shenandoah River State Park, VA	5	1.26E+05	0.117	0												
Sky Meadows State Park, VA	5	1.34E+05	0.070	0												
Smith Mountain Lake State Park, VA	5	3.26E+05	0.105	0												
Southwest Virginia Museum State Pk, VA	5	3.38E+04	0.495	0												

Staunton River State Park, VA	5	1.81E+05	0.083	0												
Staunton River Battlefield State Park, VA	5	2.46E+04	0.199	0												
Twin Lakes State Park, VA	5	9.24E+04	0.100	0												
Westmoreland State Park, VA	5	1.20E+05	0.066	1	1	1	1	1	1	1	11.65	9.59	7.79	7.83	1.45	1.59
Wilderness Road State Park, VA	5	1.38E+05	0.196	0												
York River State Park, VA	5	1.58E+05	0.112	1	1	1	0	1	1	1	7.29	5.75	6.32	6.39	0.83	0.90
Breaks Interstate Park, VA	5	3.27E+05	0.027	0												
Assateague State Park, MD	10	1.16E+06	0.167	0												
Big Run	10	7.28E+03	0.168	0												
Bridgetown NRMA	9	1.29E+03	0.390	0												
Calvert Cliffs State Park, MD	10	6.53E+04	0.540	1	0	1	0	1	1	1	8.20	6.92	8.29	8.16	1.92	2.09
Casselman River Bridge State Park, MD	10	3.58E+03	0.410	0												
Cedarville State Forest, MD	10	4.79E+04	0.329	0												
Chapel Point State Park, MD	10	8.12E+03	0.655	1	1	1	0	1	1	1	14.34	12.27	7.79	7.90	1.19	1.38
Chapman State Park, MD	7	3.81E+03	0.203	0												
Choptank River Fishing Pier State Pk, MD	10	3.90E+04	0.321	1	0	1	0	0	1	1	9.86	8.29	6.99	7.17	1.51	1.58
Cuningham Falls State Park, MD	10	3.21E+05	0.308	0												
Dans Mountain State Park, MD	10	8.39E+04	0.516	0												
Deep Creek NRMA	4	8.51E+05	0.342	0												
Deep Creek	10	3.60E+05	0.552	0												
Elk Neck State Park, MD	10	4.52E+05	0.312	1	1	1	1	1	1	1	6.22	5.06	7.48	7.59	1.30	1.56
Fair Hill Natural Res Mgmt Area, MD	10	5.38E+05	0.166	0												
Fort Frederick State Park, MD	10	1.01E+05	0.280	0												
Franklin Point, Shady Side, MD	3	2.53E+03	0.486	0												
Gambrill State Park, MD	10	8.06E+04	0.273	0												
Gathland State Park, MD	10	7.24E+04	0.144	0												
Greenbrier State Park, MD	10	2.56E+05	0.210	0												
Greenwell State Park, MD	10	1.11E+05	0.723	0												
Gunpowder Falls State Park, MD	10	8.27E+05	0.421	1	0	1	1	1	1	1	21.30	13.53	8.26	8.26	0.87	1.33
Hallowing Point Park, MD	3	2.22E+04	0.029	0												
Hart Miller Island State Park, MD	7	1.18E+05	0.906	1	0	1	1	1	0	0	8.95	7.88	7.72	7.84	1.16	1.32
Herrington Manor State Park, MD	10	1.14E+05	0.088	0												
Janes Island State Park, MD	10	9.51E+04	0.436	1	1	1	1	1	1	1	5.37	4.52	7.23	7.25	2.35	2.48
Jennings Randolph	7	1.35E+04	0.518	0												
Martinak State Park, MD	10	5.07E+04	0.142	1	1	1	0	1	1	1	20.69	17.05	7.14	7.35	0.51	0.62
Merkle Wildlife Management Area, MD	10	1.22E+04	0.311	1	0	1	0	1	1	1	36.08	33.81	7.92	8.10	0.43	0.50

Monocacy Natural Resource Area, MD	10	2.51E+04	0.589	0												
Morgan Run Environmental Area, MD	10	1.90E+04	0.456	0												
New Germany State Park, MD	10	3.13E+04	0.143	0												
North Point, MD	10	8.83E+04	0.620	1	1	1	1	1	1	1	11.02	9.29	8.01	8.06	1.19	1.34
Patapsco Valley State Park, MD	10	7.90E+05	0.199	1	0	1	0	1	0	1	23.30	17.89	9.05	8.80	0.93	1.07
Patuxent Natural Res Mgmt Area, MD	10	7.13E+03	0.235	0												
Patuxent River State Park, MD	10	2.17E+05	0.139	0												
Pocomoke River Shad Landing St Pk, MD	10	1.39E+05	0.349	1	1	1	0	1	1	1	5.11	5.68	5.90	6.15	0.63	0.71
Point Lookout State Park, MD	10	4.05E+05	0.127	1	1	1	1	1	1	1	7.80	6.54	7.95	7.88	1.88	1.99
Purse State Park, MD	10	3.37E+03	0.501	1	1	1	0	1	0	1	12.89	11.53	7.94	8.00	1.07	1.32
Rocks State Park, MD	10	1.28E+05	0.451	0												
Rocky Gap State Park, MD	10	7.52E+05	0.332	0												
Rosaryville State Park, MD	10	3.03E+04	0.592	0												
Sandy Point State Park, MD	10	8.92E+05	0.119	1	1	1	1	1	1	1	15.93	12.56	8.72	8.56	1.09	1.22
Sassafras River	10	8.55E+03	0.136	0												
Seneca Creek State Park, MD	10	5.72E+05	0.156	1	1	1	0	0	1	1	17.67	13.55	8.05	8.12	1.03	1.23
Severn Run Natural Env Area, MD	10	4.88E+04	0.261	0												
Smallwood State Park, MD	10	5.36E+04	0.382	1	1	1	0	1	1	1	23.29	17.15	9.36	9.29	0.70	0.75
Soldiers Delight Natural Env. Area, MD	10	3.03E+04	0.262	0												
South Mountain State Park, MD	10	6.63E+04	0.229	0												
Saint Clements Island State Park, MD	10	9.39E+03	0.446	1	0	1	0	1	0	0	10.77	8.59	7.88	7.85	1.51	1.63
St Marys State Park, MD	10	8.98E+04	0.401	0												
Susquehanna State Park, MD	10	1.93E+05	0.528	1	1	1	0	1	1	1	9.31	9.13	7.75	7.83	0.98	1.13
Swallow Falls State Park, MD	10	2.59E+05	0.049	0												
Tawes Garden, Annapolis, MD	4	2.06E+04	0.326	0												
TCB NCRT	2	3.50E+05	0.032	0												
Tuckahoe State Park, MD	10	8.00E+04	0.112	1	0	1	0	1	1	1	20.69	17.05	7.14	7.35	0.51	0.62
Washington Monument, Washington, DC	10	7.50E+04	0.118	0												
Western Maryland Scenic Rail Road	7	7.99E+04	0.335	0												
Wye Island Natural Res Mgmt Area, MD	10	3.09E+04	0.225	1	0	1	0	1	0	1	7.03	5.56	8.04	8.05	2.16	2.41
Wye Oak State Park, MD	10	1.96E+04	1.017	0												
Youghiogheny Path, MD	6	1.07E+04	0.238	0												
Zekiah Swamp natural Env Area, MD	9	1.97E+03	0.305	0												
Bellevue State Park, DE	9	3.43E+05	0.721	0												
Brandywine Creek State Park, DE	9	1.68E+05	0.702	0												
Cape Henlopen State Park, DE	9	1.22E+06	0.350	0												
Delaware Seashore State Park	9	1.69E+06	0.494	0												

Fenwick Island State Park, DE	9	3.66E+05	0.482	0												
Fort Delaware State Park	9	3.90E+04	1.154	0												
Fort DuPont State Park, DE	7	8.19E+03	0.873	0												
Fox Point State Park, DE	8	6.63E+04	0.727	0												
Holts Landing State Park, DE	9	6.42E+04	0.591	0												
Killens Pond State Park, DE	9	2.55E+05	0.564	0												
Lums Pond State Park, DE	9	1.96E+05	0.213	0												
Trap Pond State Park, DE	9	1.16E+05	0.568	0												
White Clay Creek State Park, DE	9	1.85E+05	0.802	0												
Wilmington State Parks, DE	7	6.76E+05	0.745	0												
Alapocas Run State Park, DE	1	1.77E+04	0.000	0												

Table 41. Summary statistics for supplemental data used to help estimate the stage 1 parameters of the aggregation recreation demand and site choice model from the stated preference survey conducted by Kirkley *et al*. (2011).

	Mean	5th	50th	95th
Trips	3.15	0	0	15.7
Distance [miles]	61.8	1	30	250
Income [$/yr]	82,263	12,500	62,500	175,000
Gender [0=female, 1=male]	0.482	0	1	1
Age	50.9	24	50	78
Household size	2.73	1	2	5

Table 42. Summary statistics for supplemental data used to help estimate the stage 1 parameters of the aggregation recreation demand and site choice model from the Chesapeake Bay TMDL stated preference survey.

	Mean	5th	50th	95th
Trips	1.72	0	0	10
Income [$/yr]	85,454	12,500	62,500	225,000
Gender [1=female, 0=male]	0.467	0	0	1
Age	55.8	28.9	58.8	79
Children under 18 in household	0.423	0	0	2

Table 43. Stage 1 estimation results for δ and λ using supplemental data from the SP survey by Kirkley *et al*. (2011).

	Coeff. est.	t-statistic
Constant	-1.99	-33.30
Phone	0.0751	1.92
Internet	1.19	26.96
College	-0.0284	-0.59
Age	0.00916	7.46
Income	-2.92E-6	-10.08
Male	-0.438	-12.60
HH size	-0.0207	-1.67
Travel cost	0.0126	96.3

Table 44. Stage 1 estimation results for δ and λ using supplemental data from the Chesapeake Bay TMDL SP survey.

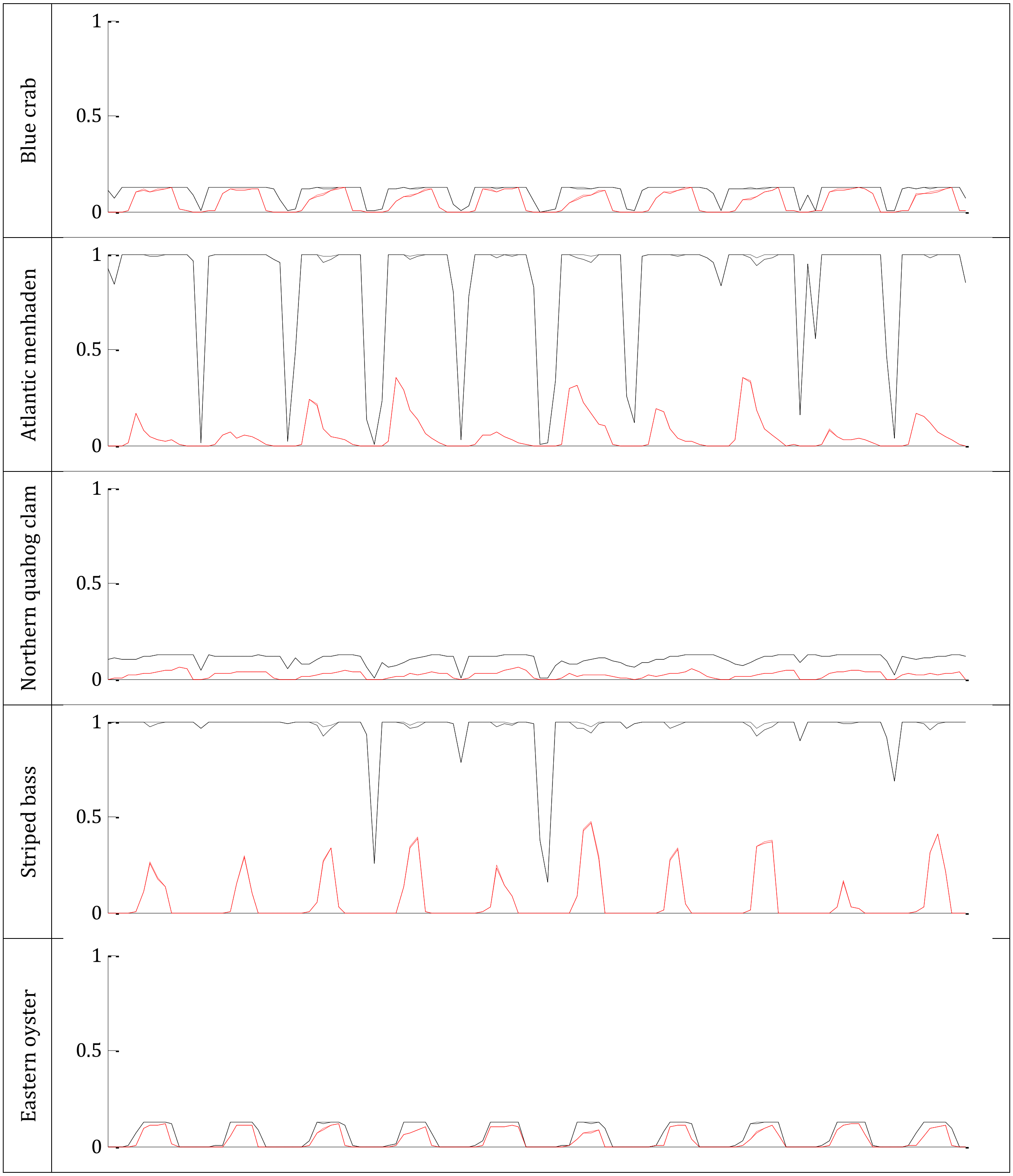
	Coeff. est.	t-statistic
Constant	-2.02	-34.54
College	0.0834	1.33
Age	0.00648	4.42
Income	-4.29E-6	-13.63
Female	0.303	7.50
Children	-0.116	-5.48
Travel cost	0.00891	158.74

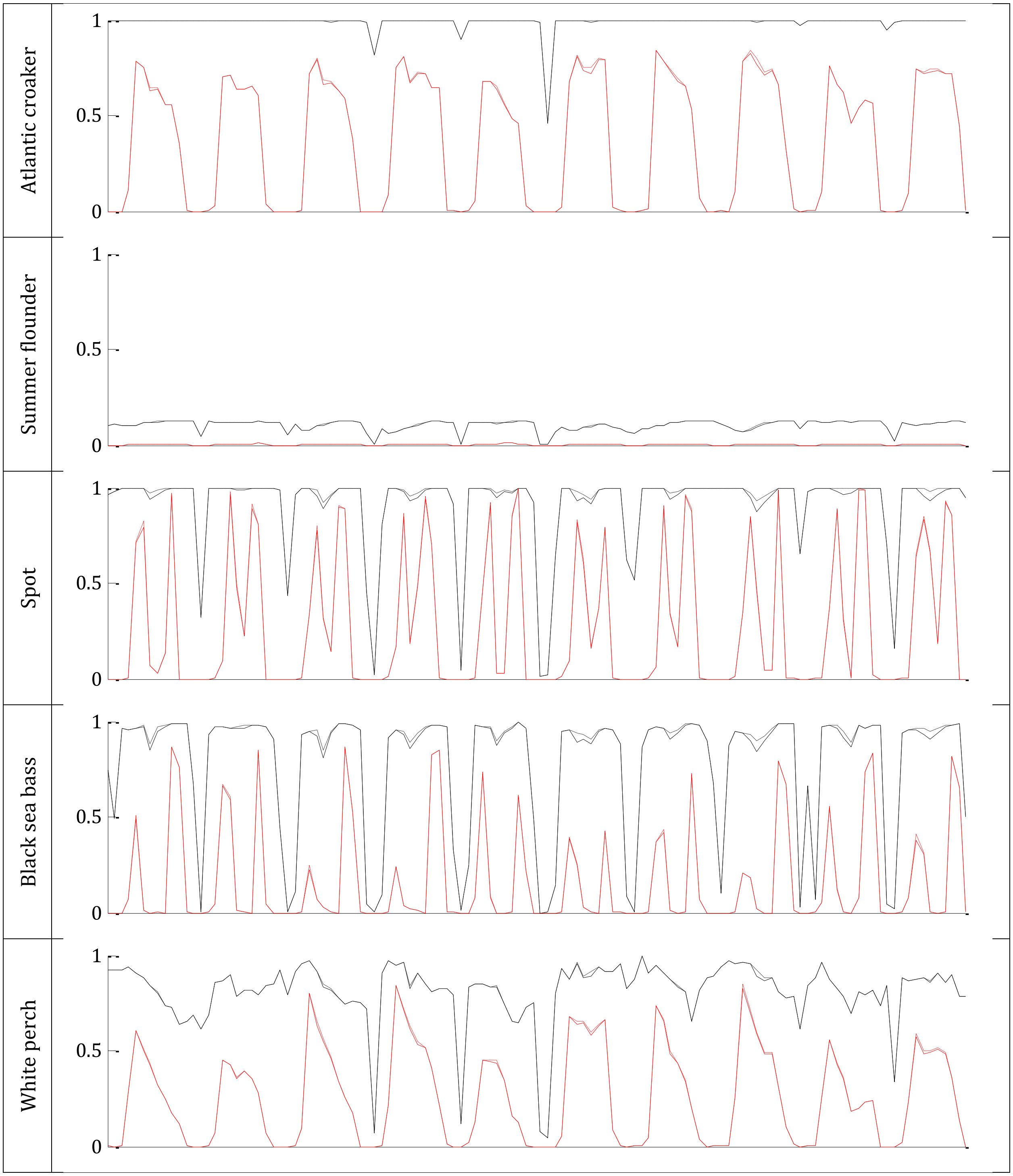
Table 45. Stage 2 regression results and associated benefit estimates for predicted improvements in water quality at all sites due to the Chesapeake Bay TMDL for a variety of estimating equation specifications, based on data from the SP survey by Kirkley *et al*. (2011). The dependent variable is the set of stage 1 estimates of the ASCs in each year. Numbers in square brackets are t-statistics. Benefit percentiles are based on Krinsky-Robb simulation. Estimates of WTP per trip can be calculated by dividing the total WTP estimates in the bottom three rows of the table by 42 million trips.

	Site fixed effects				Site amenity indicators						
Chl	-0.00496 [-1.06]	-0.0104 [-0.84]	0.0180 [0.47]		-0.0104 [-1.56]			-0.0128 [-1.24]	0.0643 [2.61]	0.164 [2.07]	0.0629 [2.57]
Chl^2		0.00013 [0.51]	-0.00019 [-0.40]						-0.00184 [-3.82]	-0.00296 [-2.94]	-0.00181 [-3.78]
ln(Chl)			-0.272 [-0.89]							-0.870 [-1.36]	
DO	0.9193 [1.42]	0.325 [0.90]	-2.480 [-1.05]			0.0448 [0.68]		0.101 [1.09]	-2.89 [-5.12]	0.0415 [0.01]	
DO^2		-0.0156 [-0.66]	0.0808 [0.94]						0.202 [5.39]	0.0946 [0.52]	0.101 [5.64]
ln(DO)			9.92 [1.25]	0.478 [1.16]						-9.46 [-0.54]	-9.977 [-5.22]
SD	0.269 [1.48]	0.619 [0.12]	-3.55 [-1.43]				0.422 [2.15]	0.200 [0.76]	-2.70 [-3.00]	-4.05 [-0.82]	-2.729 [-3.03]
SD^2		0.0955 [0.34]	1.00 [1.43]	0.173 [1.93]					1.64 [3.55]	1.96 [1.40]	1.639 [3.55]
ln(SD)			1.44 [1.49]							0.558 [0.29]	
$\bar{R}^2$	0.9190	0.9183	0.9187	0.9192	0.4007	0.3966	0.4053	0.4045	0.4817	0.4823	0.4835
$\widehat{WTP}$ for predicted water quality changes at all sites due to Chesapeake Bay TMDL [10^6 \$ yr^{-1}]											
5th	31	30	-117	28	-5	-3	42	48	98	12	97
50th	171	186	65	179	85	2	199	204	284	224	281
95th	312	339	250	330	175	7	354	359	469	430	469

Table 46. Stage 2 regression results and associated benefit estimates for predicted improvements in water quality at all sites due to the Chesapeake Bay TMDL for a variety of estimating equation specifications, based on supplemental data from the Chesapeake Bay TMDL SP survey. The dependent variable is the set of stage 1 estimates of the ASCs in each year. Numbers in square brackets are t-statistics. Benefit percentiles are based on Krinsky-Robb simulation. Estimates of WTP per trip can be calculated by dividing the total WTP estimates in the bottom three rows of the table by 42 million trips.

	Site fixed effects				Site amenity indicators						
Chl	-0.00499 [-1.07]	-0.0105 [-0.85]	0.0188 [0.49]		-0.00658 [-0.99]			-0.0151 [-1.48]	0.0618 [2.50]	0.193 [2.43]	0.200 [2.75]
Chl^2		0.00013 [0.52]	-0.00020 [-0.41]						-0.00182 [-3.76]	-0.00329 [-3.27]	-0.00338 [-3.70]
ln(Chl)			-0.280 [-0.91]	-0.132 [-1.96]						-1.143 [-1.79]	-1.198 [-2.03]
DO	0.0919 [1.42]	0.331 [0.91]	-2.511 [-1.06]			0.0985 [1.51]		0.175 [1.90]	-2.626 [-4.63]	0.832 [0.16]	
DO^2		-0.0159 [-0.68]	0.0817 [0.95]						0.189 [5.02]	0.06224 [0.34]	0.092 [5.12]
ln(DO)			-3.524 [-1.41]	0.753 [1.63]						-11.1 [-0.64]	-8.286 [-4.25]
SD	0.271 [1.49]	0.0559 [0.11]	-3.524 [-1.41]				0.353 [1.80]	0.0616 [0.24]	-2.512 [-2.77]	-3.71 [-0.76]	-2.410 [-2.68]
SD^2		0.0994 [0.36]	0.995 [1.42]						1.463 [3.15]	1.735 [1.24]	1.379 [2.98]
ln(SD)			1.426 [1.48]							0.512 [0.27]	
$\bar{R}^2$	0.9173	0.9166	0.9171	0.9176	0.3937	0.3964	0.3985	0.4020	0.4691	0.4737	0.4773
$\dot{W}TP$ for predicted water quality at all sites due to Chesapeake Bay TMDL [10^6 \$ yr^{-1}]											
5th	46	44	-164	19	-49	-1	14	9	42	-96	-54
50th	244	265	93	115	76	6	236	228	306	203	217
95th	443	482	355	206	204	13	453	446	569	495	493


Blue crab
1
0.5
0
Atlantic menhaden
1
0.5
0
Northern quahog clam
1
0.5
0
Striped bass
1
0.5
0
Eastern oyster
1
0.5
0


Atlantic croaker
Summer flounder
Spot
Black sea bass
White perch
1
0.5
0

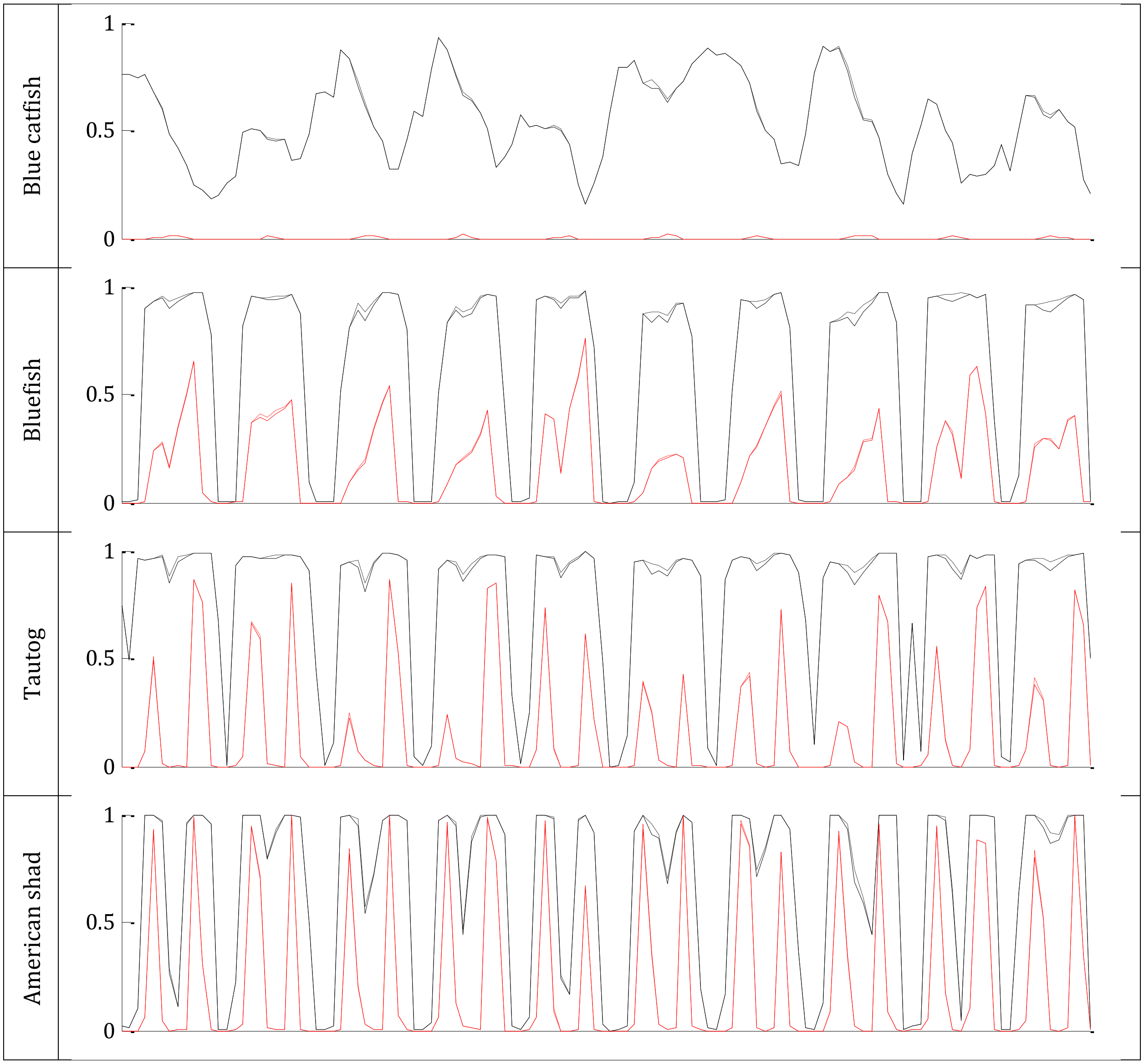


Figure 1. Habitat volume time series for juvenile stages of 14 species in the Chesapeake Bay under projected baseline and TMDL conditions, corresponding to rainfall patterns between 1991-2000. The height of each curve is the fraction of the total volume of water represented in the Chesapeake Bay Estuary Model that satisfies the “required” or “optimal” water quality conditions for each species. Black lines correspond to required water quality conditions; red lines correspond to optimal water quality conditions; solid lines correspond to the baseline scenario; broken lines correspond to the TMDL scenario. Differences between the solid and broken lines indicate the predicted effect of the TMDL on habitat volumes in each month for each species life stage. The 14 species shown here are those that appear in the multi-species fishery model described in section 2.3.

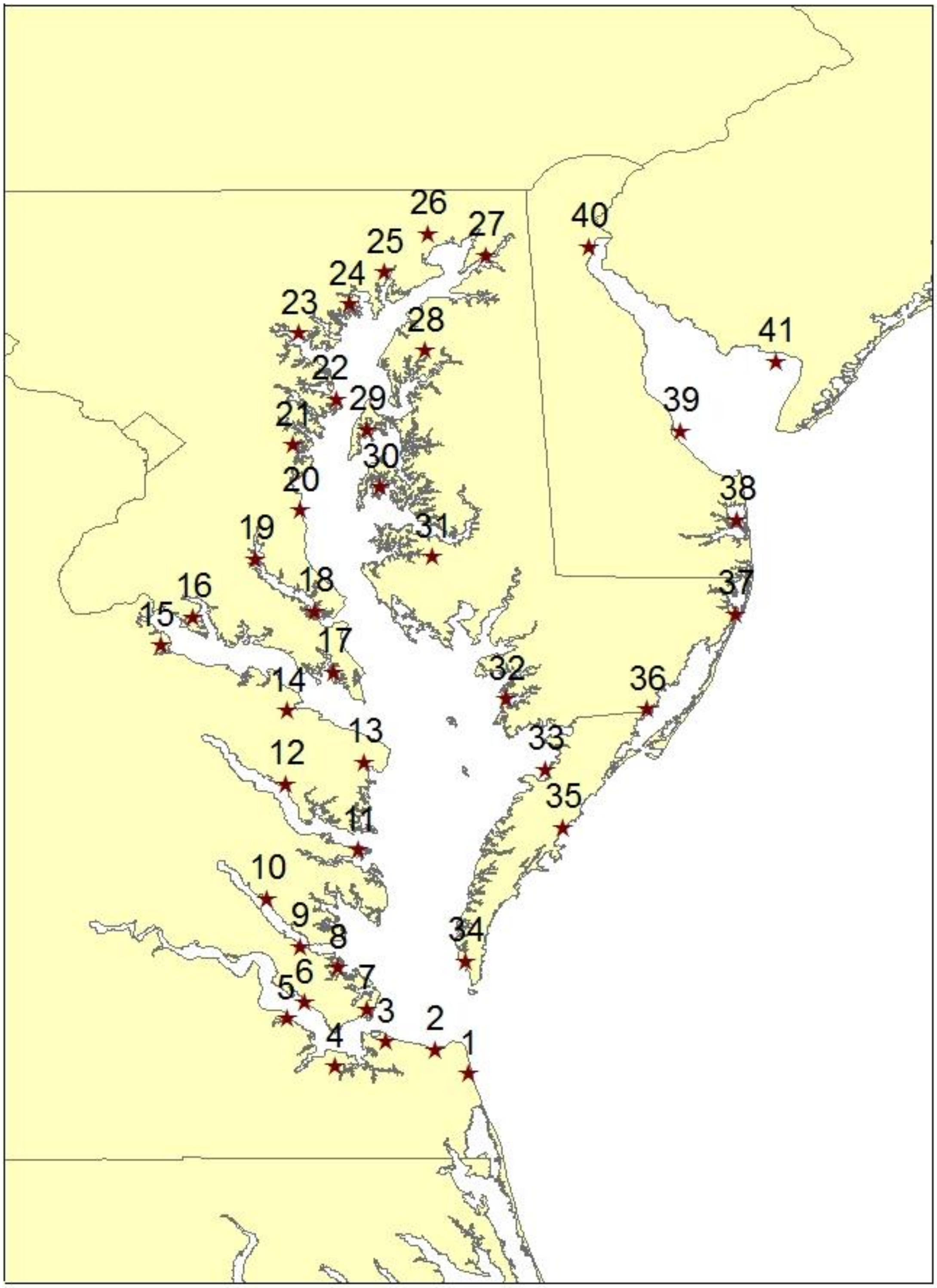


Figure 2. Aggregated access point in and around the Chesapeake Bay.

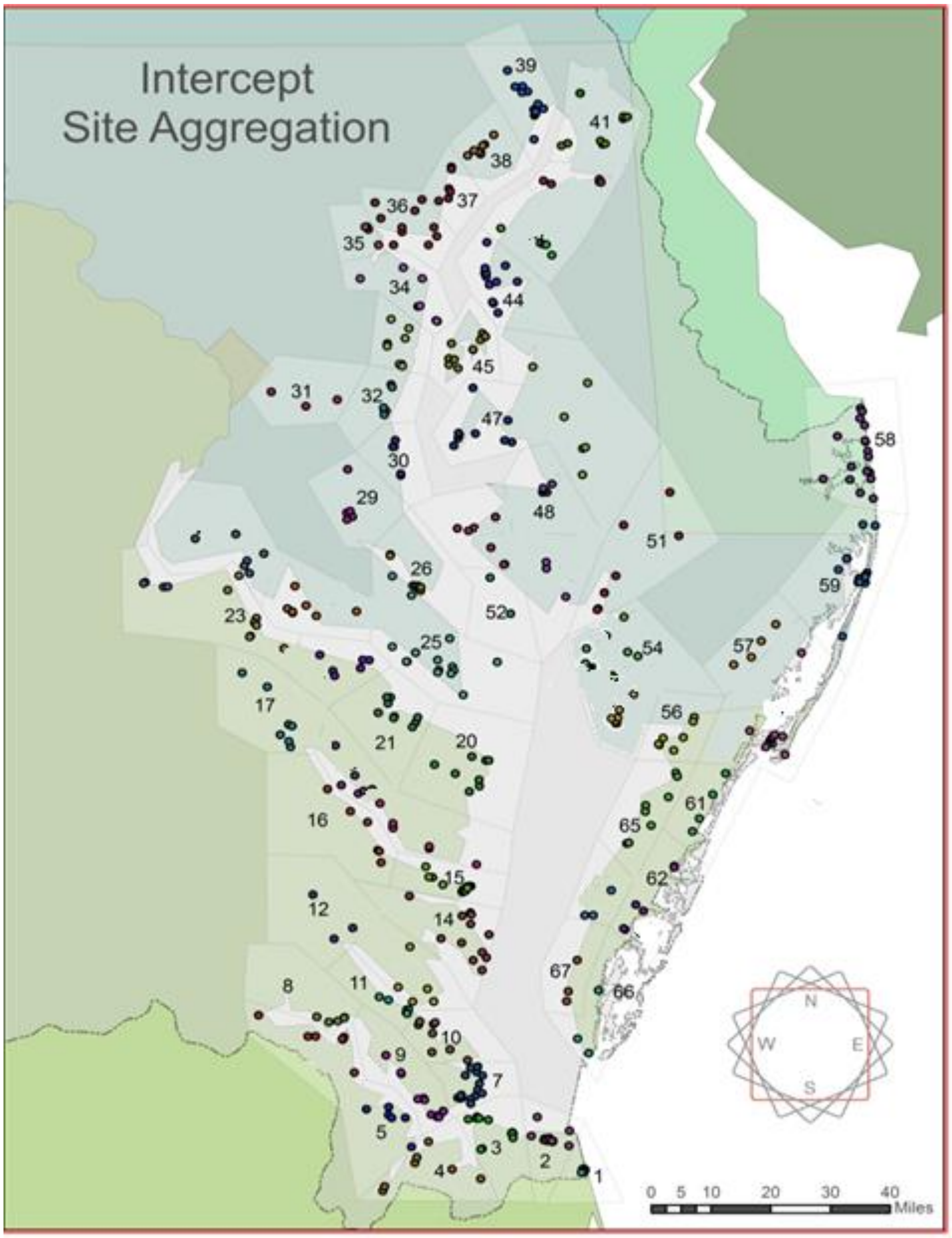


Figure 3. NOAA-defined recreational fishing access points in and around Chesapeake Bay. The site numbers reflect the original sixty-seven sites, before some groups of nearby sites were lumped and treated as single sites for estimation.

Appendix A

This appendix provides an overview of the Chesapeake Bay Fisheries Ecosystem Model (CBFEM), which was used to conduct preliminary exploratory analyses of the effects of water quality changes on aquatic living resources in the Chesapeake Bay, and was consequently modified to improve its representation of the influences of nutrient loads on harvested fish species for use in the Chesapeake Bay TMDL benefits analysis. This approach expands on previous published work (Townsend 2014) in which output from the Chesapeake Bay Water Quality Modeling suite was used to create forcing functions for the CBFEM. These forcing functions were used to explore the effects of water quality management strategies on Chesapeake Bay living resources. The previous work only explored the effects of nitrogen removal from the trophic flows of the system and did not fully consider the effects of nitrogen removal on nitrogen cycling and habitats in the system. The analysis in this appendix uses a variety of approaches to incorporate the potential negative consequences of eutrophication and its influence on dissolved oxygen in the bay waters.

The CBFEM also has been used to assess the effects of habitat change on blue crab fisheries (Ma *et al*. 2010). This application was exploratory as quantitative information on the extent to which submerged aquatic vegetation habitat influences predation on blue crab was not available. Additional research to quantify habitat effects on trophic flows would be necessary for developing a more realistic linked modeling system.

Basic model structure

The CBFEM is a fisheries-oriented trophic network model for the Chesapeake Bay based on a widely applied software package, Ecopath with Ecosim (EwE) (Christensen and Walters 2004). The Ecopath module of the CBFEM uses data on biomass, production/ biomass, consumption/biomass, and species' diet compositions for 57 trophic groups representing the major fisheries species of the Chesapeake Bay as well as significant prey, predators, and primary producers to estimate the structure of a mass-balanced aquatic ecosystem under conditions in the bay that prevailed in the 1950s. The state variables of the balanced system then serve as initial conditions for Ecosim, which is a dynamic simulation model (Christensen *et al*. 2009). The 57 trophic groups represent either single stocks or sub-stocks or species groups that occupy similar foraging niches. The major species for the model include multi-life-stage representations of commercially important species (striped bass, bluefish, weakfish, white perch, Atlantic menhaden, blue crab, and oyster) as well as single biomass pool groups of other commercially important species (American eel, Atlantic croaker, summer flounder, spot, alewife, American shad, black drum, catfish, and bivalves) (Christensen *et al*. 2009).

Ecopath module parameterization is based on satisfying two 'master' equations for each model group: one for production and the other for consumption. The Ecosim module of the CBFEM provides a 60-year (1950-2009) simulation that attempts to estimate the current status and dynamics of the bay's fish species. This module can be used to simulate various management options for the Chesapeake Bay by varying parameters over time to estimate potential ecosystem changes.

Input data for most trophic groups included in the model were based upon advice from local experts at the Chesapeake Bay Ecopath Workshop (Sellner *et al*. 2001) as well as journal articles with published local data. Other basic input parameter estimates were taken from peer-reviewed literature sources, tagging studies conducted in the Chesapeake Bay by authors of the CBFEM Technical Memorandum (Christensen *et al*. 2009), FishBase (http://www.fishbase.org), other models, and estimations made by Ecopath itself.

The Ecosim module of the CBFEM is used to simulate management policy scenarios. The Ecosim module requires extensive time series data for calibration. Time-series data that represent trends in relative biomass, fishing effort by gear type, fishing and total mortality rates, and catches for the period of 1950-2009 were also used to drive and calibrate the simulation model. In addition to time series drivers (e.g., catch and fishing mortality data) in the CBFEM, a primary production forcing function was used to drive the production of phytoplankton and benthic macroalgae.

The CBFEM focuses on modeling fisheries stocks within the Chesapeake Bay and tidal portions of its tributaries. Some groups being modeled, such as black drum, reside in the Chesapeake Bay but are considered part of larger 'stocks' usually encompassed by the eastern or northeast United States. Further complicating matters, many of the groups spend only part of the year or different parts of their life histories within the Chesapeake Bay, such as weakfish or bluefish. Therefore, in order to derive time series for EwE time simulations, it was often necessary to develop assumptions and correction factors such that stock assessments for a larger population could be applied to the Chesapeake Bay EwE model. These assumptions and correction factors are fully documented in a NOAA Technical Memorandum (Christensen *et al*. 2009).

Representing changes in nutrient loads in the CBFEM

The internal dynamics of the CBFEM is driven by production rates for each of the major species/trophic groups and the flows between the groups (trophic interactions). External drivers for the CBFEM result in changes to the biomass of the trophic groups and changes in flow rates among the groups. External drivers include top-down drivers (i.e., harvest) and bottom-up drivers (i.e., nutrient inputs that fuel primary production at the bottom of the food web). Fisheries harvest is discussed later in the methods. Modifications to the primary production forcing function were made to reflect changes in nutrient loading associated with the TMDL scenarios.

As the CBFEM was designed to allow long-term (multi-decadal scale) simulation of changes in the fisheries, a long-term primary production forcing function was necessary. We used a simple, linearized, barotropic, two-layer hydrodynamic model, Chesapeake Bay Regional Estuarine Ecology Model (CBREEM; Ma *et al*. 2009) and satellite indices of chlorophyll-a to create this function. CBREEM uses climatologic, hydrologic, and nutrient loading data to estimate historical patterns in primary productivity (chlorophyll-a) for a regional estuary on a monthly time-scale. The forcing function is depicted in Figure A1.

Nutrient loading correlates with chlorophyll-a (Figure A2), so we assumed that the relative change in nutrient loading would result in the same relative change in chlorophyll-a. For each TMDL scenario, the primary production forcing function was modified according to the relative change in nutrient loading (Table A1).

Simulating eutrophication in the CBFEM

Because of uncertainty in the potential effects of additional nutrients on the ecosystem, two different forcing functions were used to simulate two extreme hypotheses regarding potential responses to increased nutrient loads. According to the “agriculture model” of estuarine ecosystems (e.g., Nixon and Buckley 2002), increased inputs of nitrogen (N) typically results in more ecological production and production of ecological consumers (e.g., fisheries species). Alternatively, Baird *et al*. (2004) demonstrated in a model of the Neuse River estuary in North Carolina that more N leads to more hypoxia which eventually leads to energy being diverted from fish production and to microbial pathways, a "microbial shunt model" of eutrophication.

The agricultural model is straightforward to simulate in the CBFEM. In this case, we assumed that relative changes in nutrient loads results in the same relative change in primary productivity so the CBFEM primary production forcing function was extended to project the changes in nutrient loads.

To represent the microbial shunt model, we assumed that above a threshold level of primary productivity, consumers were no longer able to efficiently use the additional primary production so bacteria alone would capitalize on the surplus nutrients. We have not found documented evidence in the literature as to what level of nutrients would result in a mircrobial shunt, so we developed a provisional model based on correlations with spring chlorophyll-a levels and summer dissolved oxygen levels from the Eutrophication Model. Specifically, we assumed that when spring chlorophyll-a levels were high enough to result in DO ≤ 5 mg/l (derived from TMDL DO criteria), the microbial shunt began to result in appreciable loss of primary production to higher trophic level consumers and the loss in primary production was proportional to the decrease in DO from the threshold (5 mg/l) during the summer. The microbial pathways for using excess nitrogen were not modeled explicitly. The microbial shunt pathways for the baseline and TMDL scenarios are depicted in Figure A3.

Bounding results

Our preliminary results using CBFEM and the modified forcing functions described above suggest that under the agricultural model assumptions the TMDL will have a negative effect on fish biomass, while under the microbial shunt model the TMDL will have a negligible effect on fish biomass. Fish catches show a similar pattern (see Figure A4).

This approach provides estimates of putative upper and lower limits on the effects of TMDL on fish biomass; however, it is limited because it does not take into account the effect of changing habitat associated with improved water quality.

The results presented above illustrate some possible outcomes of the effects of water quality management plans on upper level trophic organisms of commercial and recreational interest. In the spirit of risk analysis this sort of linkage is useful for pointing out potential outcomes that are unexpected or unintended (Levin *et al*. 2009). Rodríguez *et al*. (2006) suggested that to make judgments about ecosystem trade-offs some additional economic analysis is necessary.

Other researchers have attempted to couple these modeling systems (Cerco *et al*. 2010, Ihde *et al*. 2016). Cerco *et al*. (2010) provided a number of valuable insights for model coupling, but did not allow the effects of fisheries management to be adequately captured due to the static nature of the Ecopath model used. Direct coupling of these models is feasible, but would require additional programming for a spatial CBFEM and aggregation of the water quality modeling results to fit the CBFEM spatial scale. Other modeling options to simulate the effects of nutrient load changes and eutrophication on fish stocks have been developed (e.g., the Atlantis model; Fulton 2004, Ihde *et al*. 2016). The Atlantis modeling system is likely a better alternative to the approach presented here as it more thoroughly represents nutrient cycling, sediment loading effects on primary production, as well as habitat growth and the benefits of its availability as refuge to prey groups. Moreover, the Atlantis approach also accounts for realistic physical forcing for temperature, salinity and water circulation by incorporating Regional Ocean Modeling Systems (ROMS) output to constrain the modeled system. An Atlantis model of the Chesapeake Bay has recently been developed and is now producing reliable results (see Appendix B; Ihde *et al*. 2016), but it was not available for use at the initiation of this project.

Application of the Atlantis model to predict effects of attainment of TMDL levels of nitrogen and sediment loadings results in more optimistic fish production compared to EwE (Appendix B). Atlantis predicts that 53% of harvested populations will increase under the improved nutrient and sediment loads required by the TMDL (relative to status quo conditions), while EwE predicts that none of these populations will increase under TMDL conditions.

Habitat volumes as consumption forcing functions in EwE

This section describes a proposed modification of Ecopath with Ecosim to account for the influence of water quality conditions on species' consumption rates through their predator-prey interactions in the model. Some initial model experiments using this approach have been tested; however, as of the submission deadline for this report, these model modifications have not been completed and thoroughly tested. The approach is described below for future consideration.

To simplify the presentation, we will explain the proposed modification in the context of a simple predator-prey system. Biomass of predators is B_j, total biomass of prey is B_i, biomass of foraging prey potentially vulnerable to predation is V_i. Following the logic of arena foraging theory (e.g., Walters and Christensen 2009), the equation of motion for vulnerable prey is

$$\frac{dV_i}{dt} = \nu_i(B_i - V_i) - \nu_i' V_i - a_{ij}\frac{H_{ij}}{H_i}V_i\frac{H_{ij}}{H_j}B_j\,. \tag{1}$$

where ν_i is the rate that refugia prey move into foraging habitat, ν_i' is the rate that foraging prey move into refuge habitat, $\frac{H_{ij}}{H_i}V_i$ is the biomass of foraging prey in habitat that overlaps that of the predator, and $\frac{H_{ij}}{H_j}B_j$ is the biomass of the predators in habitat that overlaps that of the prey. Note that the last term on the right hand side of equation (1) is the rate of consumption of prey *i* by predator *j*, which we will denote Q_{ij} following standard EwE notation.

Next, if the exchange rates of prey between refuge habitat and foraging habitat are rapid compared to rates of change of B_j and B_i, then the vulnerable foraging prey abundance will closely track the total prey and predator abundances, so we can write:

$$V_i = \frac{\nu_i B_i}{\nu_i + \nu_i' + a_{ij}\frac{H_{ij}}{H_i}\frac{H_{ij}}{H_j}B_j}\,. \tag{2}$$

We can substitute (2) in the consumption term in (1) to write the rate of consumption of prey *i* by predator *j* as follows:

$$Q_{ij} = a_{ij}\frac{H_{ij}}{H_i}V_i\frac{H_{ij}}{H_j}B_j = \frac{\frac{H_{ij}}{H_i}\frac{H_{ij}}{H_j}a_{ij}\nu_i B_i B_j}{\nu_i + \nu_i' + \frac{H_{ij}}{H_i}\frac{H_{ij}}{H_j}a_{ij}B_j}. \tag{3}$$

Note that if the habitat occupied by foraging prey that is accessible to the predator shrinks towards zero, $H_{ij} \to 0$, e.g., because increased hypoxic conditions exclude the predator from areas of low DO still tolerable to the prey species, then the predator consumption term also goes to zero, as it should.

The consumption equation in EwE is specified as follows:

$$Q_{ijt} = \frac{\alpha_{ijt}a_{ij}\beta_{ijt}\nu_i B_{it}B_{jt}}{2\beta_{ijt}\nu_i + \alpha_{ijt}a_{ij}\frac{B_{jt}}{\theta_{ijt}}}, \tag{4}$$

where it is assumed that $\nu_i' = \nu_i$ and where α_{ijt}, β_{ijt}, and θ_{ijt} are externally defined “forcing functions,” which allow for the representation of a variety of time-dependent exogenous factors that may influence the search rate of predators for their prey, the exchange rates of prey between refuge and foraging habitat, and the size of predator-prey foraging arenas, respectively. To match the form of the consumption function in(3), we can specify the search rate forcing function as follows:

$$\alpha_{ijt} = \frac{H_{ij}^2}{H_i H_j}. \tag{5}$$

This provides a simple recipe for developing a set of forcing functions in EwE based on the habitat volumes of each species in the model, plus the overlaps of habitat volumes between all species pairs. Equation (5) represents a (quasi-)mechanistic link between changes in water quality conditions in the Bay and species’ growth and survival rates, as influenced by the full set of predator-prey interactions represented in the model. These habitat volumes and habitat volume overlaps were estimated using the approach described in section 2.2.

Monthly average DO as production forcing functions in EwE

Within EwE, biomass growth rate is driven by an initial input value of the production-to-biomass ratio (P/B). External forcing functions representing environmental drivers can be used to induce variability in P/B and result in changes in biomass over time in the simulation model depending on a species' preference for the driver level. This allows simulation of changes in growth and reproduction under sub-optimal conditions.

Applying forcing functions to the CBFEM

The CBFEM is designed to simulate the Chesapeake fisheries ecosystem from 1950 to 2009. However, the Chesapeake Bay Eutrophication Model is designed to simulate from 1991 to 2011 for Baseline and TMDL conditions with an additional run for historical conditions. The CBFEM runs were projected to 2031 to simulate baseline and TMDL scenarios from 2011 to 2031. Model runs were divided into a burn-in phase (1950-1990), a calibration phase (1991-2010), and a projection phase (2011-2031). Fishing effort was projected past 2009 by assuming a constant fishing effort for each species, the constant effort was calculated as the average effort for 2004-2009.

To create the forcing function from 1950-2011 in the CBFEM, the volumes were replicated using historical CEM output. For the projection phase of the two scenarios, the Baseline and TMDL output from the CEM was used.

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Tables and figures

Table A1. Preliminary projections of nitrogen loading (from eutrophication model) under hypothetical TMDL scenarios.

Year	Increasing Baseline	Constant Baseline	TMDL
2010	248	248	248.0
2011	250.5	248	242.7
2012	253.0	248	237.4
2013	255.5	248	232.1
2014	258.1	248	226.7
2015	260.7	248	221.4
2016	263.3	248	216.1
2017	265.9	248	210.8
2018	268.5	248	207.7
2019	271.2	248	204.6
2020	273.9	248	201.5
2021	276.7	248	198.4
2022	279.5	248	195.3
2023	282.2	248	192.2
2024	285.1	248	189.1
2025	287.9	248	186

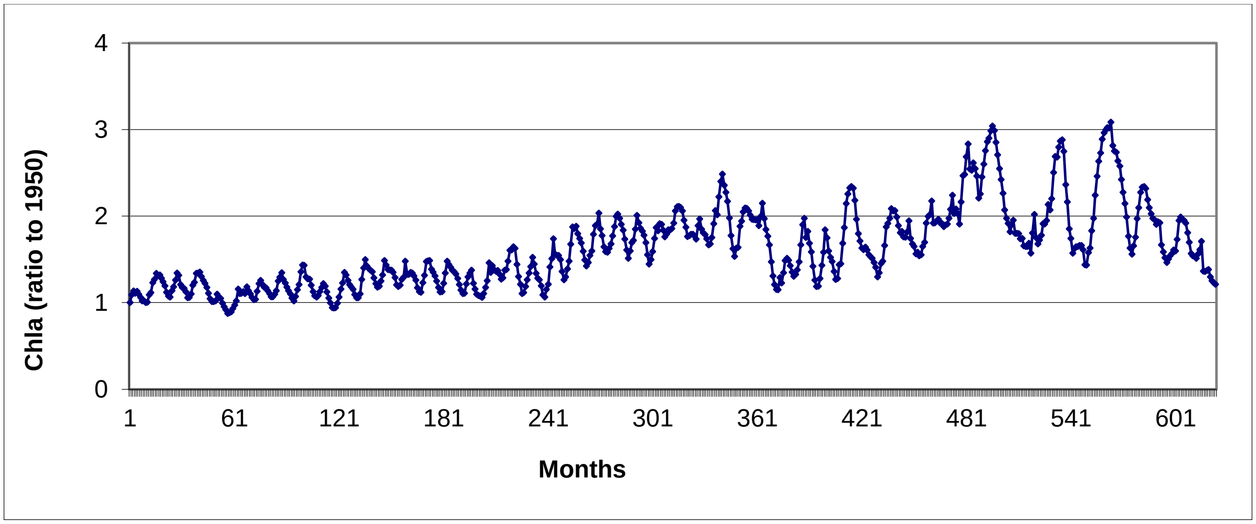


Figure A1. Primary production forcing function used for CBFEM.

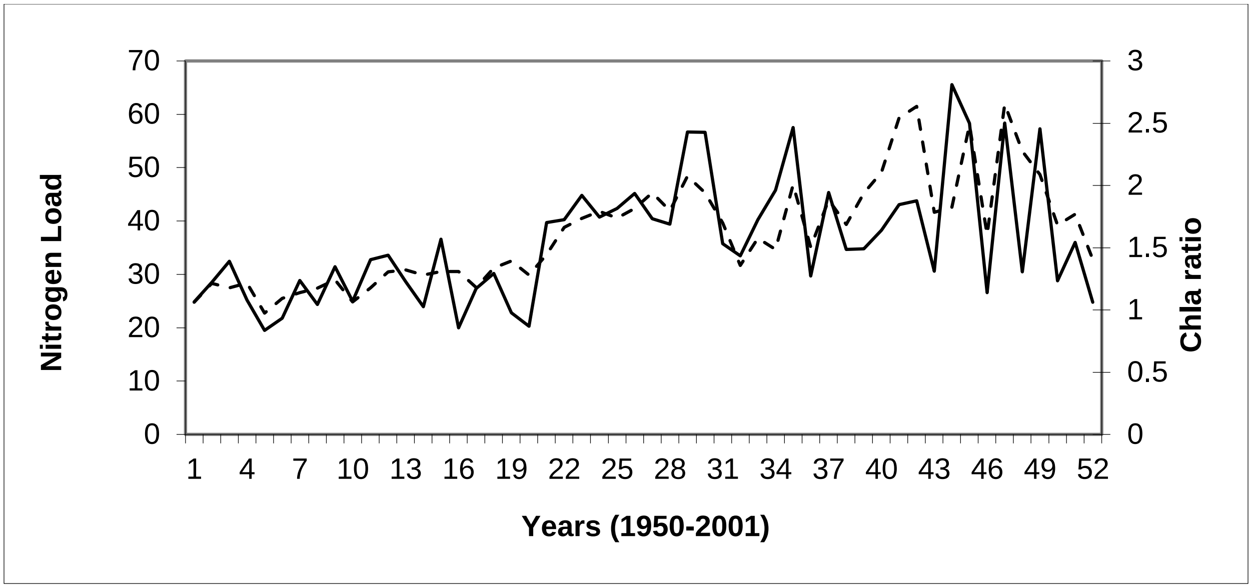


Figure A2. Chlorophyll a /primary production (dashed line) and nitrogen loads (solid line).

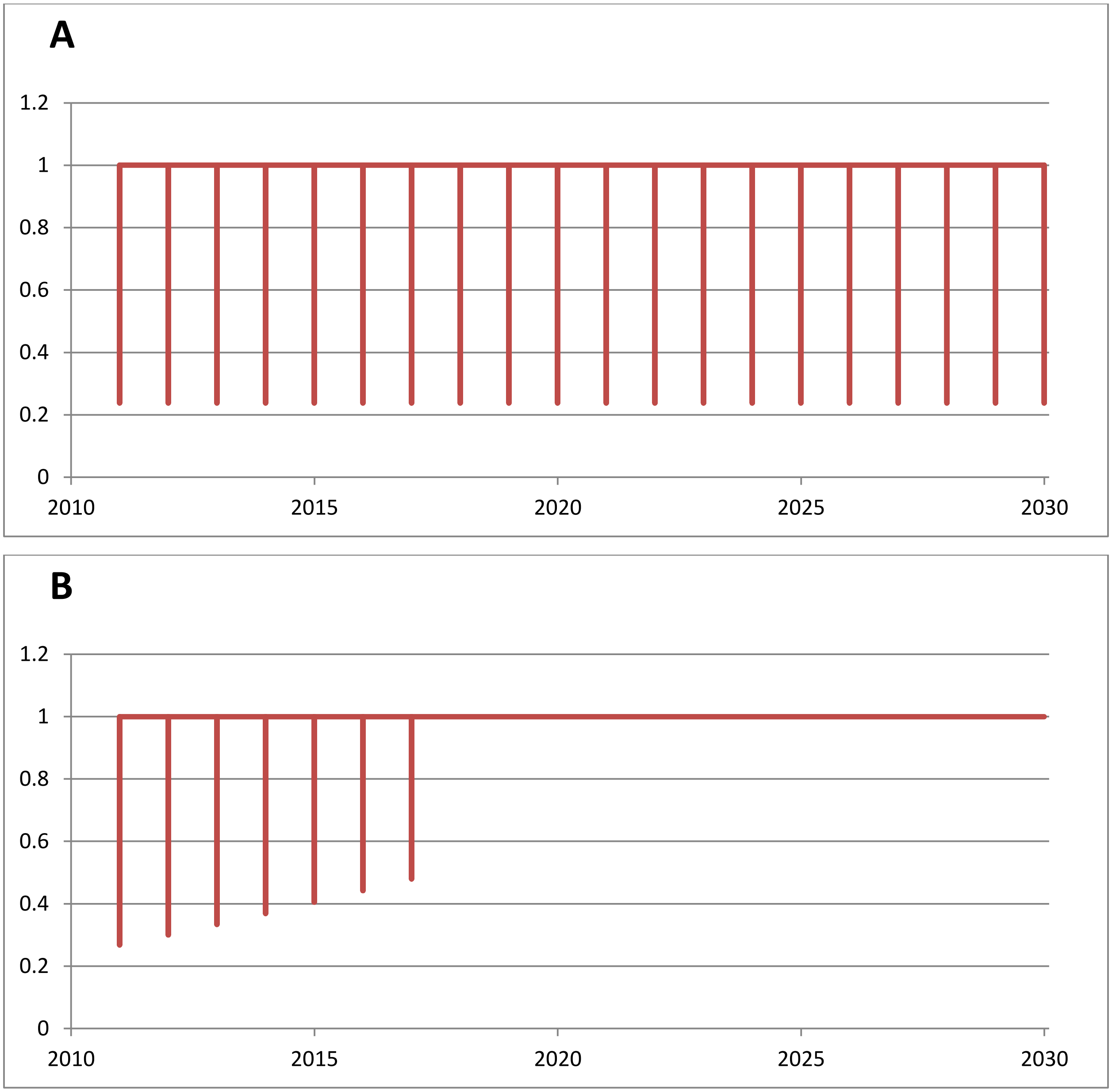


Figure A3. Eutrophication forcing functions for the microbial shunt. A) Constant Baseline, B) TMDL.

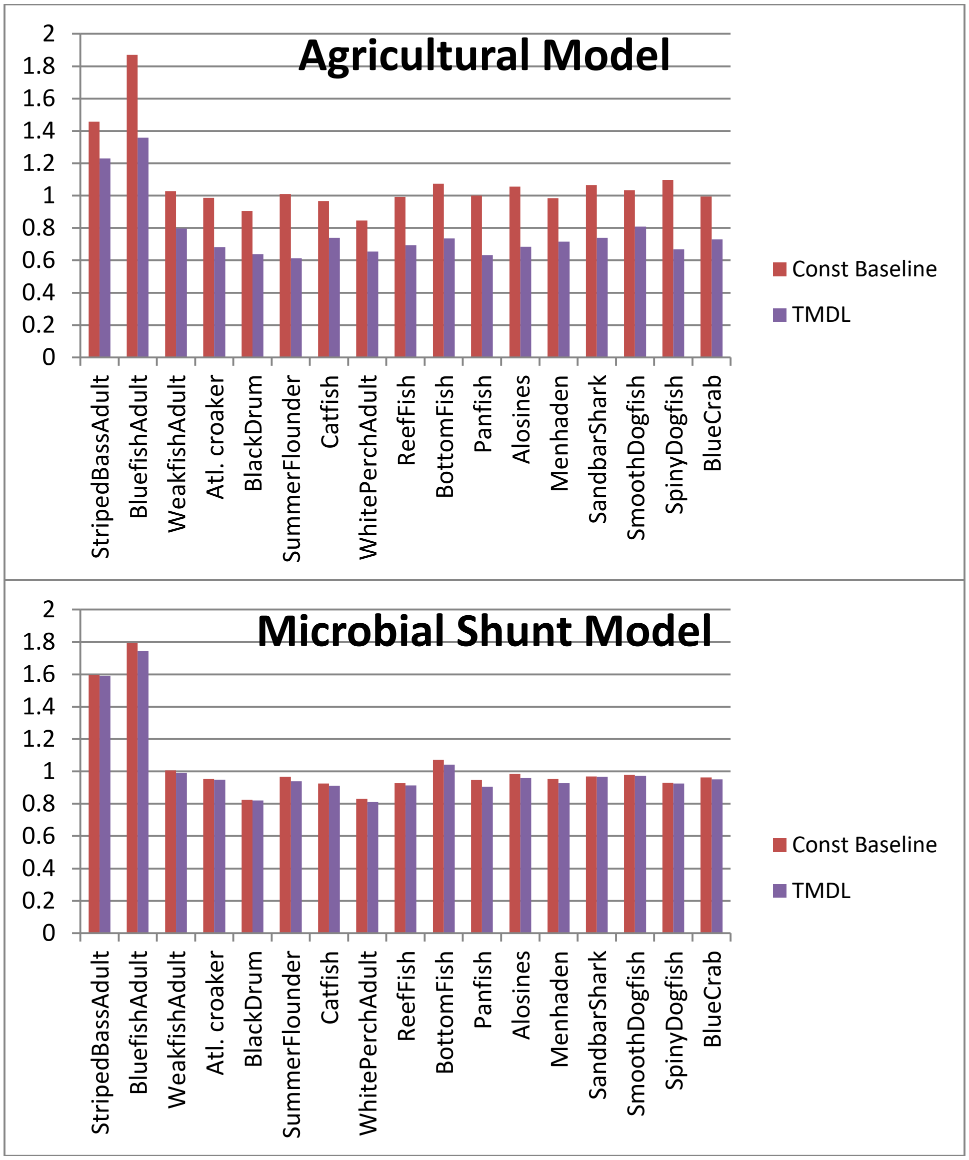


Figure A4. Relative change (over 20 yrs) in fish biomass under different water quality management scenarios (constant baseline and TMDL) and different assumptions about the influence of eutrophication on the fisheries food web (e.g., Agricultural and Microbial Shunt Models).

Appendix B

This appendix provides an overview of the Chesapeake Atlantis Model (CAM), which was used to conduct preliminary exploratory analyses of the effects of water quality changes on aquatic living resources in the Chesapeake Bay. Atlantis is a full system approach (or "end-to-end" model) that, like EwE, ultimately models productivity of the system. The Atlantis approach, however, attempts to simulate or account for all of the key features of a system that affect production, including physical, chemical, biological (age-structured life history characteristics, biological timing, and movement), ecological (trophic), habitat, as well as nutrient dynamics through bacterial cycling and plant growth effects (Ihde *et al*. 2016). In contrast, Ecopath with Ecosim is a trophic-focused approach that can represent the influence of over-enrichment on hypoxia and the subsequent effects of hypoxia on aquatic fauna through exogenously specified "forcing functions" (Appendix A). The current approach using CAM described here builds on earlier work (Ihde *et al*. 2016), where a preliminary analysis of TMDL attainment for nitrogen and sediment loads is described (a reduction of 25% and 20%, respectively, from 2010 levels [US EPA 2010]).

Atlantis is particularly well-suited to simulate system improvements related to the TMDL because the approach simultaneously accounts for multiple effects of nitrogen and sediment inputs into the system. Moreover, the model directly incorporates the actual loading outputs for both nitrogen and suspended sediment from the EPA Chesapeake Bay Program Office's Watershed Model, for both the status quo (circa 2010) and TMDL (full attainment) scenarios. Habitat changes estimated with the Atlantis approach includes TMDL effects on both the physical habitats (submerged aquatic vegetation, or "SAV," and microscopic sediment algae, called microphytobenthos or "MPB," and the chemical conditions in the water column. Physical changes accounted for include water clarity improvements (due to reduced suspended sediment loading) and multiple effects on the growth of submerged plants—both SAV and MPB. The effects on SAV and MPB are similar, and could be positive (due to improved light penetration from a reduction in both the magnitude of phytoplankton blooms and suspended sediment) or negative (due to decreased nutrient availability).

Application of Atlantis in estimating TMDL effects offers additional benefits over simpler approaches. Some of these benefits include: identification of unintended consequences, an improved understanding of scale, and identification of cumulative effects of the TMDL in combination with other simultaneous system changes. Accounting for multiple environmental factors when estimating productivity of multiple populations, as done with both EwE and Atlantis, commonly allows for non-intuitive and non-linear affects on groups to be realized, because such approaches allow modeled groups to interact in an unanticipated but reasonable manner. Such unanticipated interactions are commonly driven by, or related to, secondary, tertiary, or even higher level effects propagated through trophic connections. Atlantis, however, allows such interactions in dimensions beyond the trophic (e.g., physical, chemical, spatial, etc.), and thus should also be expected to predict unanticipated consequences of system change that EwE cannot. Atlantis integrates data from a diverse array of disciplines at a wide variety of scales. Once data are entered, they can be visualized together on a shared scale, and thus, dominant drivers for a system can be more easily recognized. Similarly, when system stressor effects (e.g., increases in water temperature, or losses of marsh habitats due to subsidence, sea level rise, or shoreline hardening) are simulated along with the TMDL improvements, the scale of the physical and chemical improvements of the TMDL are put into context with other such system-changing factors, and the dominant drivers of productivity change can be identified.

5.1 Basic model structure

CAM is a deterministic, forward projecting, spatially-explicit, production model (Ihde *et al*. 2016). The model domain encompasses 8,896 sq. km (3,434 sq. mi.) and includes the brackish waters and sediments of the Chesapeake Bay and eight of its largest tributaries: in Virginia, the James R., York R. (including large portions of the Mattaponi and Pamunkey Rivers), and Rappahannock R.; the Potomac R.; and in Maryland, the Patuxent R., the Susquahanna R., and on the eastern shore of Maryland, the Choptank R., and Nanticoke R. The model area is divided into 97 irregular polygons, or “boxes,” which are contiguous and relatively homogenous areas defined by salinity and depth, and also by bottom type in the main trunk of the system.

The model includes physical inputs for temperature, salinity, water movement, dissolved inorganic nitrogen, dissolved organic nitrogen, labile and refractory particulate matter, and silica. Water movements are driven by the Navy Coastal Ocean Model (NCOM) Relocatable Model (http://ecowatch.ncddc.noaa.gov/amseas/). NCOM has a horizontal resolution of 1/30 degrees, roughly 3 km as input into the CAM domain. The Hybrid Coordinate Ocean Model (HYCOM) provides boundary conditions, and atmospheric forcing is from Coupled Ocean-Atmosphere Mesoscale Prediction System (COAMPS). Tidal forcing is included. Since our main goal at this point was to capture the strong seasonal patterns of this shallow estuary, rather than inter-annual variability, we repeat the oceanographic conditions from 2012 for each year simulated in CAM.

CAM includes 55 functional groups to model biological processes: 26 invertebrate (including the primary producers) and 29 vertebrate groups. Most invertebrates are modeled as single biomass pools (in each box), but 2 invertebrate groups (Blue crab and Squid) are modeled instead as linked juvenile and adult biomass pools. Each functional group, if not sessile, can “move” at each time step (12 hours) to search and optimize its spatial distribution in relation to food and habitat (refuge) availability. All vertebrate groups are divided into 10 age classes, each tracked by abundance and weight at age. Weights are measured through both structural and reserve nitrogen, with structural nitrogen representing bones and other hard parts and reserve nitrogen representing muscle, fat, reproductive parts, and other soft tissue. The separation of age classes for vertebrates allows for ontogenetic shifts in the parameterization. The trophic structure of CAM is as similar to that of the EwE model described in Appendix A as the structure of the two models allow, but predictions of CAM are expected to be more similar to the EwE-microbial shunt model, since the microbial forcing of this model mimics bacterial dynamics explicitly modeled in Atlantis, but not included in the trophic structure of the EwE model.

Habitat types in Atlantis include both physical and biogenic habitats. We defined four physical habitats: mud (includes all substrates except clean sand useful for oyster recruitment), sand (does not include clayey-sand or silty-sand) (Shepard 1954), rock (or man-made structure, e.g., artificial reefs, hardened shoreline, etc.), and woody debris. Biogenic habitat types are Marsh, SAV, and Oyster reef. Biogenic habitats convey refuge to juveniles or adults of groups as specified by the modeler. The proportion of these seven habitat types sums to one for each CAM box. Fish and other animal groups can be assigned a "dependence" to one or more of the seven habitat types, and at least one of those habitats must be available in a CAM box to allow that group to move into that box.

5.1.1 Representing changes in nutrient loads in CAM

Nutrient and sediment loads to the model were derived from the Chesapeake Bay Phase 5.3 Community Watershed Model (US EPA 2010). This watershed model predicts nitrogen, phosphorous, and sediment transport to the Bay system, and how these respond to changes in watershed management. The model includes non-point nutrient inputs, such as atmospheric deposition, fertilizer, and manure. We use two simulations of the Watershed Model, the first being a calibration scenario, and the other a scenario that assumes reduced loads under restrictions associated with a Total Maximum Daily Load (TMDL). The TMDL scenario includes future attainment of 25% reductions in nitrogen and 20% reduction in sediment inputs as required under the federal Clean Water Act (http://www.epa.gov/reg3wapd/tmdl/ ChesapeakeBay/tmdlexec.html).

We estimated the biological effects of fully achieving the goals of the EPA Total Maximum Daily Load (TMDL) requirements for the jurisdictions of the Chesapeake Bay watershed, by comparing base model production to that of a model modified with TMDL-level reductions in nutrient and sediment loads. Nutrient and sediment inputs for the TMDL and base simulations varied and are described above, but all other aspects of the scenarios were identical. Though the majority of groups (62%) were predicted to have increased production under the TMDL when compared to the status quo scenario predictions for the same 70 year time span, the changes were generally minor. Realized change for most groups (40 of 52 groups) was less than 3%. Moderate increases were relatively common, and the productivity increases of five fish groups (Alosines, Catfish, Panfish, Other flatfish, White perch) ranged between five and seven percent. Only one group had an increase in production greater than 7% (Dinoflagellates; 12%). Similarly, few groups realized moderate or large decreases in productivity in the TMDL simulation compared to the base case.

Atlantis predictions were more variable and more optimistic than were EwE estimates for the same time frame. When Atlantis predictions for the status quo and TMDL scenarios were compared to starting biomass estimates for 20 year scenarios (as described for EwE runs, Appendix A), Atlantis predictions for fished groups of management importance were more variable than those of EwE, by factors that ranged between less than 0.2x to more than 8x (Figure B1). The Atlantis simulation also predicted an increase in productivity of seven of the 17 commercially- or recreationally-important groups shown in Figure B1, while none of these groups were predicted to have an increase in the more comparable EwE model (Figure B1-B).

Since the trophic structure (i.e., predator-prey relationships) is similar in both modeling approaches, the differences in model predictions are attributable to the greater complexity of the Atlantis model structure. The biogeophysical structure that allows for estimation of cumulative effects of change in the system with Atlantis is consequently thought to be a more complete representation of the benefits of the TMDL to living resources (i.e., fished species) of management interest than is EwE.

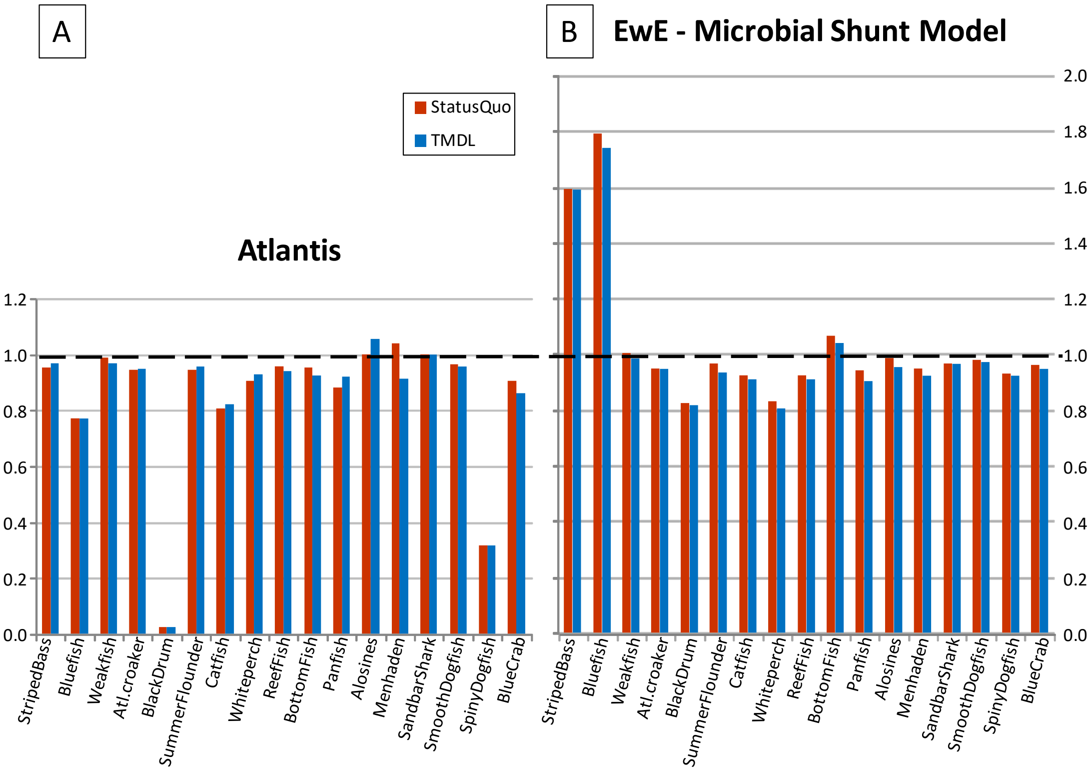


Figure B1. Productivity estimates of fished stocks simulated in Atlantis (panel A) and EwE (panel B), for 20 year simulations, relative to starting biomass estimates. Estimates from Atlantis were somewhat more variable than those of EwE, and also suggest most harvested populations will continue to decline over the next 20 years whether under status quo or TMDL conditions. Atlantis projections predict that TMDL conditions will benefit more than half (53%) of the harvested populations, whereas none of the populations simulated with EwE benefit from improved water quality conditions. Estimates for the EwE Microbial Shunt Model (more comparable to Atlantis estimates because the microbial forcing of this model mimics bacterial dynamics explicitly modeled in Atlantis) shown are the same as those presented in FigureA4 (lower panel), but are redrawn to be on the same scale as Atlantis estimates. Dashed lines are starting biomass.

References

Ihde, T.F., I.C. Kaplan, E.A. Fulton, I.A. Gray, M. Hasan, D. Bruce, W. Slacum, and H.M. Townsend. 2016. Design and parameterization of the Chesapeake Bay Atlantis Model: A spatially explicit end-to-end ecosystem model. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-F/SPO-166, 145 p.

US EPA. 2010. Chesapeake Bay Phase 5.3 Community Watershed Model. U.S. Environmental Protection Agency, EPA 903S10002 - CBP/TRS-303-10, http://ches.community modeling.org/models/CBPhase5/documentation.php#scenario Annapolis, 558 p.

Shepard, F.P. 1954. Nomenclature based on sand-silt-clay ratios. Journal of Sedimentary Research 24(3):151-158.

Appendix C

This appendix presents a minimal model of hypoxia hysteresis, closely following Sheffer *et al*. (2001). We begin by assuming that nitrogen (in phytoplankton) in the water column, N, is controlled by a fixed loading rate, a, burial rate, b, and flux from recycling of buried nitrogen, $r(D)B$ (which depends on the dissolved oxygen concentration, D, as described below):

$$\frac{dN}{dt} = a - bN + r(D)B.$$

Nitrogen buried in bottom sediment, B, is controlled by input from nitrogen in the water column, bN return flux to the water column, $r(D)B$, and export from the system (or deep burial), xB:

$$\frac{dB}{dt} = bN - r(D)B - xB.$$

Dissolved oxygen in water column, D, is controlled by a fixed aeration rate, c, and consumption of oxygen by decomposition of phytoplankton:

$$\frac{dD}{dt} = c - dbND.$$

Next, we assume that the rate of release of sediment nitrogen is low at high oxygen concentrations but switches in a sigmoid fashion to a high rate as D falls below the threshold h:

$$r(D) = g - \frac{D^p}{D^p + h^p}.$$

Solving the system for the steady-state, we find that the equilibrium oxygen concentration is:

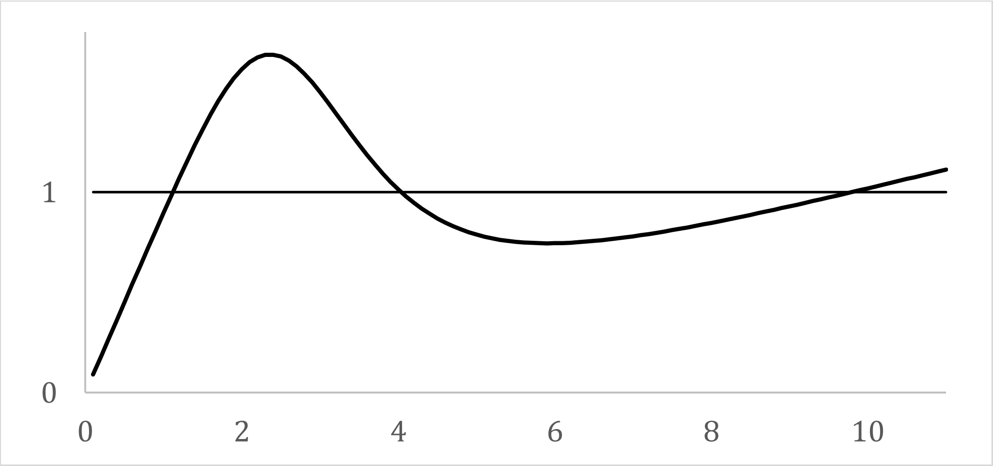
$$\frac{c}{dD} = a\left[1 + \frac{1}{x}\left(g - \frac{D^p}{D^p + h^p}\right)\right].$$

We can re-write the above expression using reduced-form parameters as follows:

$$\frac{1}{D} = \alpha - \beta\frac{D^p}{D^p + h^p}.$$

The left-hand side is a hyperbola, so it declines monotonically as D increases along the x-axis. The right-hand side is a reversed S-curve. It declines slowly at first, then rapidly around $D = h$, then slowly again. Therefore, these curves can intersect at three values of D. These points of intersection can be seen by plotting the product of D and the right-hand side of the equilibrium equation on D itself and noting where this curve crosses the y-axis value of 1. The example shown in the graph below uses $p = 5$, $h = 3$, $\alpha = 0.9$ and $\beta = 0.8$.

In this example, there is a high equilibrium around $D = 10$, a low equilibrium around $D = 1$, and a threshold around $D = 4$. Suppose the dissolved oxygen concentration is initially high, around $D = 10$. If the environment is stochastic, then external perturbations such as variations in nutrient loading rates and weather variables will cause D to bounce around its central value of 10. Small perturbations are "corrected," since when $D > 4$ the tendency of the system is to converge back to the high-D equilibrium around 10. However, if a large negative perturbation causes D to drop below 4, then the tendency of the system will be to converge to the low-D equilibrium around 1. The system has experienced a "regime-shift" to a new equilibrium, or "basin of attraction."



Appendix D

Adult bluefish reside and spawn almost entirely in offshore coastal waters and so cannot be neatly fit into the basic framework that is used for the other species in the multi-species fishery model. This appendix describes a simplified version of the model tailored for offshore spawners such as bluefish. The adult stock will be denoted by the single state variable, N. In equilibrium, the number of recruits from patches A and B equals the number of adults that die each year:

$$R_A + R_B = N(1 - e^{-M-F}) . \tag{1}$$

As in the basic model, the number of spawners is assumed to be equal to the average adult abundance over the year:

$$S = \left(\frac{1 - e^{-M-F}}{M + F}\right) N . \tag{2}$$

Bluefish enter coastal bays and estuaries as juveniles. We assume that a fixed fraction of juveniles, γ, enter the Chesapeake Bay and the remaining fraction, $1 - \gamma$, enter other bays and estuaries. Only individuals in the juvenile stage can be affected by water quality conditions in the Chesapeake Bay, and, as in the basic model, we assume that juveniles are active swimmers and are able to effectively avoid lethal hypoxic conditions. We also will assume that $b_A/b_B = (1-\gamma)/\gamma$ i.e., the relative juvenile carrying capacities in the two patches is proportional to the relative juvenile abundances. This leads to the following recruitment functions for the two patches:

$$R_A = \frac{(1-\gamma)aS}{1 + \frac{(1-\gamma)^2}{\gamma} ab_B S} \text{ and } R_B = \frac{\gamma S}{1 + \gamma a b_B S/\hat{V}} . \tag{3}$$

Next, we can combine(1), (2), and (3) to get

$$\frac{(1-\gamma)a\left(\frac{1 - e^{-M-F}}{M + F}\right)}{1 + \frac{(1-\gamma)^2}{\gamma} ab_B \left(\frac{1 - e^{-M-F}}{M + F}\right)} + \frac{\gamma a \left(\frac{1 - e^{-M-F}}{M + F}\right)}{1 + \gamma a b_B \left(\frac{1 - e^{-M-F}}{M + F}\right) N/\hat{V}} = 1 - e^{-M-F} , \tag{4}$$

and then use the quadratic equation to solve (4) for b_B :

$$b_B = \frac{(AD + BC - BE - DE) \pm \sqrt{(AD + BC - BE - DE)^2 + 4(BDE)(A + C - E)}}{2BDE}, \qquad (5)$$

where $A = (1-\gamma)a\left(\frac{1-e^{-M-F}}{M+F}\right)$, $B = \frac{(1-\gamma)^2}{\gamma}a\left(\frac{1-e^{-M-F}}{M+F}\right)N$, $C = \gamma a\left(\frac{1-e^{-M-F}}{M+F}\right)$, $D = \gamma a\left(\frac{1-e^{-M-F}}{M+F}\right)N/\hat{V}$, and $E = 1 - e^{-M-F}$.

Text box D1. Calibration algorithm for offshore spawners

1. Calculate the stock abundance using $N = (Q_A + Q_B)/B$.
2. Set the initial value for the juvenile-per-spawner coefficient to $a = \left(F_{msy} + M\right)^2/M$. (This is the value of a that would obtain in a single patch system with $c = 0$.)
3. Calculate b_B using (5).
4. Use numerical simulation to calculate $\hat{Q}$ and $\hat{F}_{msy}$ conditional on the current values of a and b_B.
5. Calculate $SSE = \left(\frac{\hat{Q}-Q}{Q}\right)^2 + \left(\frac{\hat{F}_{msy}-F_{msy}}{F_{msy}}\right)^2$.
6. Adjust a and repeat steps 3-6 to minimize SSE.

Appendix E

A key simplifying assumption underlying the commercial fishing benefit estimates presented in the main text is that the fishing effort in each fishery will not change as a result of the water quality improvements associated with the TMDL. To examine the implications of this assumption, we used a simplified fisheries model to compare the commercial fishery benefits under three alternative assumptions about the management regime in each fishery. The framework we used for this analysis is comprised of a Shafer-logistic model (e.g., Clark 1990 p 9-16) with a constant elasticity demand curve applied to each fishery but no cross-price effects among species.

First, we compared the change in total surplus under the fixed effort (FE) assumption, which was used to derive our primary estimates of commercial fishery benefits in section 3, to an alternative model that assumes open access (OA). To be concrete, we denote the iso-elastic demand curve as follows:

$$p(Q) = \alpha Q^{-\varepsilon}\,. \tag{1}$$

In the FE model, the total surplus change is the area under the demand curve between the initial and final quantities:

$$\Delta S_{FE} = \int_{Q_0}^{Q_{1,FE}} p(Q)dQ. \tag{2}$$

Plugging the price function from (1) into (2) and then integrating and simplifying gives:

$$\Delta S_{FE} = \alpha Q_0^{1-\varepsilon}\left[\frac{1}{1-\varepsilon}\left(1+\frac{\Delta K}{K}\right)^{1-\varepsilon} - \frac{1}{1-\varepsilon}\right], \text{where } \alpha = p_0 Q_0^{\varepsilon}\,. \tag{3}$$

Note that in (3) we have used the fact that in the FE model the new harvest will increase by the same proportion as the stock size, which in turn is proportional to the carrying capacity, i.e., $Q_{1,FE} = Q_0\left(1+\frac{\Delta K}{K}\right)$.

In the OA model, fishery rents will be dissipated both before and after the water quality improvement, so the total (consumer) surplus change is the area behind the price function (i.e., under the inverse price function, $Q(p)$) between the final and initial prices:

$$\Delta S_{OA} = \int_{p_{1,OA}}^{p_0} Q(p)dp. \tag{4}$$

Plugging the inverse price function, $Q = (p/\alpha)^{-1/\varepsilon}$, into (4) and then integrating and simplifying gives:

$$\Delta S_{OA} = \alpha Q_0^{1-\varepsilon}\left[\frac{\varepsilon}{1-\varepsilon}\left(1+\sigma_{OA}\frac{\Delta K}{K}\right)^{1-\varepsilon} - \frac{\varepsilon}{1-\varepsilon}\right]. \tag{5}$$

Note that in (5) we have assumed the change in carrying capacity is small enough to use a first-order approximation for the new quantity, $Q_{1,OA} = Q_0\left(1+\sigma_{OA}\frac{\Delta K}{K}\right)$, where $\sigma_{OA} = \frac{\partial Q}{\partial K}\frac{K}{Q}$.

To derive an expression for σ_{OA}, first note that under open access, and assuming a traditional Shafer-logistic fishery model, the following conditions will hold:

$$pQ = cE. \tag{6}$$

where E is fishing effort and c is the variable cost of effort;

$$F = qE, \tag{7}$$

where F is the fishing mortality rate and q is the catchability coefficient; and

$$Q = FK(1 - F/r), \tag{8}$$

where K and r are the carrying capacity and intrinsic growth rate, respectively, in the logistic growth model. Combining(1), (6), (7), and (8) and simplifying gives:

$$1 = \frac{\alpha q K}{c} Q^{-\varepsilon}\left(1 - \frac{\alpha q}{cr} Q^{1-\varepsilon}\right). \tag{9}$$

We cannot solve (9) explicitly for Q, but we can solve for $\partial Q/\partial K$ using implicit differentiation. To do so, we take the derivative of both sides of (9) with respect to K, which gives:

$$0 = \frac{\alpha q K}{c}\left[-\varepsilon Q^{-\varepsilon-1}\frac{\partial Q}{\partial K} - \frac{\alpha q}{cr}(1-2\varepsilon)Q^{-2\varepsilon}\frac{\partial Q}{\partial K}\right] + \frac{\alpha q}{c}\left(Q^{-\varepsilon} - \frac{\alpha q}{cr}Q^{1-2\varepsilon}\right). \tag{10}$$

Solving (10) for $\partial Q/\partial K$ and then multiplying by K/Q gives:

$$\sigma_{OA} \equiv \frac{\partial Q}{\partial K}\frac{K}{Q} = \frac{1 - \frac{\alpha q}{cr}Q^{1-\varepsilon}}{\varepsilon + \frac{\alpha q}{cr}(1-2\varepsilon)Q^{1-\varepsilon}}. \tag{11}$$

Next, we combine (6) and (7) to get

$$\frac{q}{c} = \frac{F}{pQ}. \tag{12}$$

Finally, substituting (1) and (12) into (11) gives:

$$\sigma_{OA} = \frac{1 - \frac{F}{r}}{\varepsilon + \frac{F}{r}(1 - 2\varepsilon)}. \tag{13}$$

In equation (13) we have derived an expression for σ_{OA} in terms of two quantities: F/r and ε. Note that if $\varepsilon = 1$ then $\sigma_{OA} = 1$ no matter the value of F/r, and if $F/r = 0.5$ then $\sigma_{OA} = 1$ no matter the value of ε. Assuming $r/2 = F_{msy}$, as in the standard Schaefer-logistic fishery model used here, then according to the estimates of F and F_{msy} used in the MSFM (Table 7), most species examined in this study have F/r values between 0.2 and 0.5. Also, simple regressions of ln(price) on ln(quantity) for each species at the level of Atlantic coast harvests indicate that most of these species have ε values between 0.1 and 0.9 (see Table E1). The levels of σ_{OA} and the relative size of the surplus gain under the OA and FE models within these broad ranges are shown in Table E2, with all cases assuming $\Delta K/K = 0.1$. These results suggest that the fixed effort assumption will tend to give larger estimates of the change in total surplus than the open access assumption.

Using this simplified framework, we also compared commercial fishery benefits under a fixed effort regime, an open access regime, and a regime in which the total economic surplus (consumer + producer surplus) is maximized.[45] A bare bones sketch of the model is as follows. Under open access (and assuming $0 \leq \varepsilon < 1$), consumer surplus is:

$$S_{OA} = \int_0^{Q_{OA}} p(x)dx - p(Q_{OA})Q_{OA} = \frac{\alpha}{1 - \varepsilon} Q_{OA}^{1-\varepsilon} - p(Q_{OA})Q_{OA}. \tag{14}$$

Based on the Schaefer-logistic fishery model we have:

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[45] An important caveat here is that we ignore the recreational fisheries in this analysis. A comprehensive management approach would maximize the sum of consumer and producer surplus in the commercial fishing sector plus the economic surplus of the recreational anglers that also exploit these stocks. Here we assume for simplicity that fishing mortality from recreational angling will not change.

$$Q_{OA} = qE_{OA}K\left(1-\frac{q}{r}E_{OA}\right). \tag{15}$$

Combining the iso-elastic demand function in (1) with the condition for rent dissipation, $p(Q_{OA})Q_{OA} = cE_{OA}$, we get:

$$Q_{OA} = \left(\frac{c}{\alpha}E_{OA}\right)^{\frac{1}{1-\varepsilon}}. \tag{16}$$

Combining (15) and (16) gives an equation in E_{OA} and the exogenous parameters of the model, which can be easily solved numerically for E_{OA}:

$$E_{OA}^{-\varepsilon} = \left(1-\frac{q}{r}E_{OA}\right)^{1-\varepsilon} = \frac{c}{\alpha(qK)^{1-\varepsilon}}. \tag{17}$$

With E_{OA} known, all other endogenous variables in the open access model can then be determined.

Under a sustainable surplus regime (again assuming $0 \leq \varepsilon < 0$), total (consumer + producer) surplus is:

$$S = \int_0^Q p(x)dx - cE = \frac{\alpha}{1-\varepsilon}Q^{1-\varepsilon} - cE. \tag{18}$$

Combining (18) with the Schaefer-logistic equation for the steady-state harvest level, $Q = qEK(1-qE/r)$, gives:

$$S = \frac{\alpha}{1-\varepsilon}[qEK(1-qE/r)]^{1-\varepsilon} - cE. \tag{19}$$

The first-order condition for a maximum is

$$\frac{\partial S}{\partial E} = \alpha[qEK(1-qE/r)]^{-\varepsilon}qK\left[1-\frac{2q}{r}E\right] - c = 0, \tag{20}$$

which gives an equation in E_{MSS} and the exogenous parameters of the model that can be easily solved numerically for E_{MSS}:

$$E_{MSS}^{-\varepsilon}\left(1-\frac{q}{r}E_{MSS}\right)^{-\varepsilon}\left(1-\frac{2q}{r}E_{MSS}\right) = \frac{c}{\alpha(qK)^{1-\varepsilon}}. \tag{21}$$

As in the open access case, with E_{MSS} known, all other endogenous variables in the maximum sustainable surplus model can then be determined.

Results of this analysis based on the summary of expert judgments and the multi-species fishery model without and with spillovers are shown in Table E3, Table E4, and Table E5, respectively. The estimated value of fully rationalizing the fisheries—i.e., the value of transitioning from an open access regime to a regime in which the sustainable economic surplus is maximized—in the Chesapeake Bay alone is around $20 million per year (Table E3 and Table E4). The estimated value of rationalizing these fisheries along the entire Atlantic coast is $31 million per year (Table E5). The estimated commercial fishery benefits of the TMDL using the summarized expert judgments are between $11 and $15 million per year (Table E3). The corresponding estimates using the multi-species fishery model without spillovers is between $1.4 and $2.3 million per year (Table E4) and with spillovers is between $1.6 and $2.5 million per year (Table E5). These results provide another set of comparisons between the potential benefits of the economic rationalization of fishery management institutions and the potential benefits of water quality improvements for fisheries (Smith and Crowder 2011). These results also suggest that, as a practical matter in light of the benefits estimated in other categories and the estimated costs of the TMDL, the estimated commercial fishery benefits will be sufficiently robust to alternative assumptions about the nature of the management regime in each fishery for the purposes of this study.

Table E1. Price flexibilities, ε, for each fishery estimated by regressing $ln(p)$ on $ln(Q)$, where p and Q are the Atlantic coast prices and quantities harvested between 1992 and 2011. Estimates shown for Atlantic croaker, blue catfish, and American shad are based on averages of all other estimated flexibilities.

Species	Price flexibility, ε
Blue crab	0.548
Atl. menhaden	0.247
N. quahog clams	0.191
Striped bass	0.135
Eastern oyster	0.521
Atl. croaker	0.557
Summer flounder	0.360
Spot	0.679
Black sea bass	0.888
White perch	0.648
Blue catfish	0.557
Bluefish	0.943
Tautog	0.966
American shad	0.557

Table E2. Open access harvest elasticities, σ_{OA}, and ratios of surplus changes under open access relative to a fixed effort management regime, $\Delta S_{OA}/\Delta S_{FE}$, over ranges of F/r and ε that cover most of the species examined in this study. All cases assume $\Delta K/K = 0.1$.

F/r	ε	σ_{OA},	$\Delta S_{OA}/\Delta S_{FE}$
0.20	0.1	3.08	0.30
0.20	0.6	1.43	0.85
0.20	0.9	1.08	0.97
0.50	0.1	1.00	0.10
0.50	0.6	1.00	0.60
0.50	0.9	1.00	0.90

Table E3. Commercial fishery benefits of fishery rationalization and of the Chesapeake Bay TMDL, based on a Shafer-logistic model and iso-elastic demand curves for each fishery. Estimated changes in carrying capacities were based on the summary of judgments by a panel of experts described in section 2.1.

	Value of TMDL under fixed effort	Value of TMDL under open access	Value of rationalization	Value of TMDL after rationalization
	[$ / yr]	[$ / yr]	[$ / yr]	[$ / yr]
Blue crab	12,336,289	9,043,601	6,130,802	11,008,979
Atl. menhaden	-2,180,658	-507,147	8,548,246	-1,840,336
N. quahog clams	1,324,298	538,691	1,160,461	1,007,827
Striped bass	472,275	281,709	194,087	373,262
Eastern oyster	2,009,958	1,008,618	3,006,056	1,819,797
Atl. croaker	173,404	161,826	39,006	164,355
Summer flounder	293,420	166,201	578,166	241,086
Spot	120,956	98,661	184,766	111,582
Black sea bass	0	0	43,534	0
White perch	37,629	30,851	89,059	34,444
Blue catfish	-9,997	-8,225	18,165	-8,967
Bluefish	0	0	5,063	0
Tautog	0	0	11	0
American shad	197	139	689	174
Total:	14,577,772	10,814,924	19,998,109	12,912,204

Table E4. Commercial fishery benefits of fishery rationalization and of the Chesapeake Bay TMDL, based on a Schaefer-logistic model and iso-elastic demand curves for each fishery. Estimated changes in carrying capacities were based on the multi-species fishery model without spillovers described in section 2.3.

	Value of TMDL under fixed effort	Value of TMDL under open access	Value of rationalization	Value of TMDL after rationalization
	[$ / yr]	[$ / yr]	[$ / yr]	[$ / yr]
Blue crab	1,172,393	864,760	6,130,802	1,036,272
Atl. menhaden	455,735	100,552	8,548,246	388,913
N. quahog clams	67,521	27,961	1,160,461	49,773
Striped bass	132,069	77,594	194,087	100,732
Eastern oyster	76,438	39,744	3,006,056	68,763
Atl. croaker	18,067	16,821	39,006	17,067
Summer flounder	184,759	104,874	578,166	151,391
Spot	95,866	78,216	184,766	88,407
Black sea bass	43,910	42,365	43,534	42,972
White perch	18,253	14,967	89,059	16,698
Blue catfish	15,476	12,755	18,165	13,948
Bluefish	8,666	8,561	5,063	8,591
Tautog	1,276	1,275	11	1,275
American shad	190	134	689	168
Total:	2,290,621	1,390,580	19,998,109	1,984,969

Table E5. Commercial fishery benefits of fishery rationalization and of the Chesapeake Bay TMDL, based on a Schaefer-logistic model and iso-elastic demand curves for each fishery. Estimated changes in carrying capacities were based on the multi-species fishery model with spillovers described in section 2.3.

	Value of TMDL under fixed effort	Value of TMDL under open access	Value of rationalization	Value of TMDL after rationalization
	[$ / yr]	[$ / yr]	[$ / yr]	[$ / yr]
Blue crab	1,583,601	1,168,194	10,116,136	1,399,469
Atl. menhaden	297,838	65,973	9,920,753	253,966
N. quahog clams	69,915	28,966	2,502,842	51,489
Striped bass	110,688	86,297	80,622	93,359
Eastern oyster	75,229	39,145	6,426,132	67,665
Atl. croaker	26,915	24,844	79,914	25,305
Summer flounder	84,405	52,405	1,716,828	69,192
Spot	110,567	91,841	212,928	102,073
Black sea bass	42,283	41,614	46,995	41,746
White perch	16,913	14,202	88,890	15,519
Blue catfish	15,479	12,757	18,169	13,951
Bluefish	45,250	44,932	20,256	44,988
Tautog	1,419	1,416	1,612	1,416
American shad	113	80	56,674	100
Total:	2,480,614	1,672,667	31,288,750	2,180,238