Ecological benefit spillovers from nutrient load reductions and management improvements in a multi-species coastal fishery

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ABSTRACT: Analysts face a variety of conceptual and practical challenges when attempting to quantify the private and public benefits of environmental quality improvements for harvested biological resources. In this paper we address two of these challenges in the context of a multi-species coastal fishery: 1) the spatial extent of the ecological benefits will be influenced by the species' dispersal and migration patterns and may not be confined to the area where habitat conditions are improved, and 2) the sustainable magnitude of the benefits generally will depend on the nature of the management regime in the affected fisheries. To address these challenges, it often will be necessary to explicitly account for spatial spillovers and to integrate non-market valuation approaches with dynamic models of fishery harvest management in a unified framework.

We develop and apply an integrated ecological and economic model of the effect of water quality improvements on the biological production and commercial harvest levels of 14 fish and shellfish species in the Chesapeake Bay, which is the second largest estuary in the world and the largest in North America. We use the model to estimate the benefits of the Chesapeake Bay Total Maximum Daily Load (TMDL) requirements for producers and consumers of the modeled species, which account for more than 80% of the total commercial fishery revenues in the region. We account for species' movements in and out of the Bay, which allows us to estimate the total benefits to consumers in the Bay and along the remainder of the U.S. Atlantic coast. In our benchmark model, we assume that the affected fisheries are regulated to protect the biological sustainability of the exploited species but that fishing effort is not fully controlled so harvester profits are dissipated in equilibrium. We also consider two alternative scenarios in which the affected fisheries are managed to maximize 1) producer surplus, and 2) producer + consumer surplus. Comparing these scenarios to our benchmark results allows us to examine the influence of the nature of the management regime on the spatial distribution and the magnitude of the ecological benefits from water quality improvements in the Bay.

The three main research questions we address in this paper are: How large are the economic benefits of water quality improvements in the Chesapeake Bay? What fraction of benefits are due to spatial spillovers to areas where water quality is not improved? How are the magnitude and spatial distribution of these benefits influenced by the nature of the management regime in the affected fisheries?

KEYWORDS: non-market valuation; water quality; fishery management; rent dissipation; Chesapeake Bay; spatial spillovers

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 fre-quency of hypoxiadefined as dissolved oxygen (DO) levels less than 2 mg/lwhich cre-ates "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 chang-es 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 bene-fits 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

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 in-flows 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 mi-crobes 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 mortal-ity 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 ecolog-ical 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]

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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) and Atlantis. 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

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 Chesa-peake Bay and connected coastal waters with and without the TMDL. The key model in-puts include estimates of the area or volume of suitable habitat for each species associat-ed with simulated water quality conditions under a baseline scenario, intended to repre-sent 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.

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 expert judgments

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 re-sults 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

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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.

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 forme.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. Blue catfish were introduced to Chesapeake Bay tributaries from midwestern 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. 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.

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 panelists 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 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 1011 cubic meters (about 20 trillion gallons).[2] 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 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

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[2] The precise figures for our calculations of habitat volumes are 29,950 cells and 7.9 × 1010 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.

by Schlenger for as many species life-stages as possible. For those species life-stages not included in Schlengers 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.[3] 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 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%.[4]

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

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[3] 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.

[4] 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.

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 expo-sure 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.[5] 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.

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

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[5] 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.

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 oxy-gen 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 con-straints, 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 structure

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.,

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

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 α, 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.[6] 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, JV_k, i.e., $dJ/dt = -\frac{-b/K}{V_k}J^2$ (Gurney and Nisbet 1998 p 125).[7] (Note that we are re-using the notation $k = 1, 2, ..., K$ 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 $J(t) = \left(\frac{b/K}{V_k}t - D\right)^{-1}$. 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 $J_1 = J_0\left(1 + \frac{b/K}{V_1}J_0\right)^{-1}$. By the same logic, the number of juveniles that survive to the end of the second month is $J_2 = J_1\left(1 + \frac{b/K}{V_2}J_1\right)^{-1}$. We can combine these equations to write J_2 as a function of J_0, V_1, and V_2: $J_2 = J_0\left(1 + \frac{b}{K}\left[\frac{1}{V_1} + \frac{1}{V_2}\right]\right)^{-1}$. Repeating this for all months of the juvenile stage and

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[6] 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.

[7] Note that bK 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.

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 operate in 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

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

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.[8] It is also easy to show that as V decreases from one (all available habitat is

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[8] This result is analogous to that of Gurney and Nisbet (1998 p 142-143), who used a Beverton-Holt model

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 harvested 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:

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

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

$$Q_A + Q_B = \frac{F}{M+F}\left(1 - e^{-M-F}\right)(N_A + N_B), \tag{8}$$

where $\frac{F}{M+F}\left(1 - e^{-M-F}\right)$ 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

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to examine the impact of cooling water intake withdrawals by a power plant on a coastal fishery.

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:
   $N_A = \frac{1}{B}\left[\frac{(1-\omega_B)\,Q_B - \omega_B Q_A}{1-\omega_A-\omega_B}\right]$ 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 juvenlie-per-spawner coefficients to:
   $a_A = \left(F_{MSY} + M^2\right)/M$ and $a_B = \left(F_{MSY}+M\right)^2/\big(M\hat{V}\big)$.
   These expressions correspond to the a's that would obtain in the model without spillovers.

3. Calculate the density-dependent juvenile moretality coefficients using:
   $$b_A = \frac{M+F}{1-e^{-M-F}}\left\{\frac{1-\gamma_A-\gamma_B}{\left[N_B-\gamma_B\left(N_A+N_B\right)\right]\left(1-e^{-M-F}\right)} - \frac{1}{a_A S_A}\right\}$$
   and
   $$b_B = \frac{M+F}{1-e^{-M-F}}\left\{\frac{1-\gamma_A-\gamma_B}{\left[N_A-\gamma_A\left(N_A+N_B\right)\right]\left(1-e^{-M-F}\right)} - \frac{1}{a_B S_B \hat{V}}\right\}.$$
   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 $SSE = \left(\frac{\hat{F}_{MSY} - F_{MSY}}{F_{MSY}}\right)^2 + \left(\frac{\hat{Q}_A - Q_A}{Q_A}\right)^2 + \left(\frac{\hat{Q_B} - Q_B}{Q_B}\right)^2$.
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).

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 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 Chesapeake 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 progress-es, 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 fisheryanyone who qualifies to obtain a license can participate, and there are no binding limits on the number of licenses that can be awardedthen 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.[9] 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.[10]

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 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:

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

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

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[9] 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).

[10] 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.

substitute for any other species in the representative consumers 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.[11]

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.[12] 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

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[11] 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, the Appendix 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.

[12] This ballpark estimate is 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 analysis we do not yet know if all three of the stated conditions will hold.

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 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 fin-fish 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 $D(U, X)$ is defined as

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

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 $D(U, X)$ evaluated at different quantity vectors, say X^0 and X^1, given a reference utility level U^0 or U^1. Specifically, the normalized compensating and equivalent surplus from exogenous quantity changes are

$$CS = D\left(U^0, X^1\right) - D\left(U^0, X_0\right), \tag{12}$$

and

$$ES = D\left(U^1, X^1\right) - D\left(U^1, X^0\right). \tag{13}$$

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),

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

where $a(X) = \alpha_0 + \sum_i \alpha_i \ln X_i + \frac{1}{2} \sum_i \sum_j \ln X_i \ln X_j$, $b(X) = \beta_0 \prod_i X_i^{\beta_i}$ and X_i are elements of the quantity vector X. Because $D(U, X)$ is homogeneous of degreee 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

$$\hat{P} = \frac{\partial D\left(U, X\right)}{\partial X} = a\left(U, X\right), \tag{15}$$

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

$$W_i = P_i X_i/Y = \alpha_i + \sum\nolimits_j \gamma_{ij} \ln X_j - U\beta_j b\left(X\right). \tag{16}$$

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

$$W_i = \alpha_i + \sum\nolimits_j \gamma_{ij} \ln X_j - \beta_i \ln Q, \tag{17}$$

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

3.3.2 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 $i = 1, 2, ..., n$ will provide estimates of expenditure shares

$$W_i = \alpha_i + \sum_j \gamma_{ij} \ln X_j - \beta_i \ln Q, \tag{18}$$

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 system

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 $r = 1, 2, ..., R$ will be estimated with the same linear inverse demand system described for the second stage

$$W_{ir} = \alpha_{ir} + \sum_j \gamma_{irs} \ln X_{rs} - \beta_{ir} \ln Q_r. \tag{19}$$

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

$$CV = \sum_i Y_i^0 \left[D\left(U^0, X_i^1\right) - D\left(U^0, X_i^0\right) \right] - \left(Y_i^1 - Y_i^0\right) \tag{20}$$

and

$$EV = \sum_i Y_i^1 \left[D\left(U^1, X_i^1\right) - D\left(U^1, X_i^0\right)\right] - \left(Y_i^1 - Y_i^0\right), \tag{21}$$

where Y_i^0 (Y_i^1) is forecasted expenditures on species i assuming baseline (TMDL) supply, X_i^0 (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

$$W_i^0 = \alpha_i + \sum_j \gamma_{ij} \ln X_i^0 - \beta_i \ln Q^0, \tag{22}$$

and

$$W_i^1 = \alpha_i + \sum_j \gamma_{ij} \ln X_i^1 - \beta_i \ln Q^1, \tag{23}$$

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

$$E_{ij} = \frac{\partial \ln P_i}{\partial \ln X_j} = \frac{\gamma_{ij} + \beta_i \left(W_j - \beta_j \ln Q\right)}{W_j} - \delta_{ij}, \tag{24}$$

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

$$\varepsilon_{irs} = \frac{\partial \ln p_{ir}}{\partial \ln x_{is}} = \frac{\gamma_{irs} + \beta_{ir}\left(w_{is} - \beta_i \ln Q_i\right)}{w_{ir}} - \delta_{rs}. \tag{25}$$

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 in-direct 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 species i in region r and species j in region s is

$$\begin{aligned}\varepsilon_{irjs} = \delta_{ij} &\left[-\delta_{rs} + \left\{\, \gamma_{irs} - \beta_{ir}\left[w_{is} + \beta_{is} \ln Q_i\right]\right\} \left(\frac{1}{w_{ir}}\right)\right] \\ &+ \left[\left\{\gamma_{ij} - \beta_i \left(W_j + \beta_j \ln Q\right) \left(\frac{1}{W_i}\right)\right\}\right] \left(\frac{X_{js}}{X_j}\right)\end{aligned} \tag{26}$$

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 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.

Appendix

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:

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

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

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

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, $Q\left(p\right)$) between the final and initial prices:

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

Plugging the inverse price function, $Q = \left(p/\alpha\right)^{-1/\varepsilon}$, 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

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

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:

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

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

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

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

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

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

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

In equation (13) we have derived an expression for σ_{OA} in terms of two quantities: Fr and ε. Note that if $\varepsilon = 1$ then $\sigma_{OA} = 1$ no matter the value of Fr, and if $Fr = 0.5$ 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 Fr 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.[13] A bare bones sketch of the model is as follows. Under open access (and assuming $0 \leq \varepsilon < 1$), consumer surplus is:

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

Based on the Schaefer-logistic fishery model we have:

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

Combining the iso-elastic demand function in (1) with the condition for rent dissipation, $p\left(Q_{OA}\right) Q_{OA} = cE_{OA}$, 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 modek which can be easily solved numerically for E_{OA}:

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

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[13] 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.

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

Under a sustainable surplus regime (again assuming $0 \leq \varepsilon < 1$), total (consumer + producer) surplus is:

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

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

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

The first-order condition for a maximum is

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

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

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

As in the open access case, with E_{MSS} known, all other endogenous variables in the maximum sustainable surplus modelcan 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 en-tire 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.

References

Figures and Tables