MARINE
RESOURCE
ECONOMICS

When is it Optimal to Delay Harvesting? The Role of Ecological Services in the Northern Chesapeake Bay Oyster Fishery

STEPHEN KASPERSKI
U.S. National Marine Fisheries Service
Alaska Fisheries Science Center

ROBERT WIELAND
Main Street Economics

Published by the MRE Foundation, Inc. in association with the
North American Association of Fisheries Economists (NAAFE)
and the International Institute of Fisheries Economics & Trade (IIFET)

Marine Resource Economics, Volume 24, pp. 361–385
Printed in the U.S.A. All rights reserved.

0738-1360/00 $3.00 + .00

When is it Optimal to Delay Harvesting? The Role of Ecological Services in the Northern Chesapeake Bay Oyster Fishery

STEPHEN KASPERSKI
U.S. National Marine Fisheries Service
Alaska Fisheries Science Center

ROBERT WIELAND
Main Street Economics

Abstract *Despite decades of rebuilding efforts, the population of oysters in the Chesapeake Bay has fallen to historically low levels. We develop a novel bioeconomic model which includes the value of ecological services provided by oysters in situ to determine the optimal length of a harvest moratorium and a subsequent harvest rate that will maximize the net present value of the oyster resource. Not surprisingly, steady-state stocks and optimal harvest rates are increasing and decreasing in ecological service values, respectively. The results also suggest that instituting a harvest moratorium and limiting harvest effort in the fishery can increase the net present value of the resource more than effort limitation alone.*

Key words Fishing moratorium, stock rebuilding, fisheries management, bioeconomic modeling, ecological value, Chesapeake Bay oysters.

JEL Classification Codes Q22, Q57, H41.

Introduction

Over two hundred years of lightly managed, open-access harvesting (Kennedy and Breisch 1983), consequent habitat loss, and more recently, disease mortality (Rothschild *et al*. 1994) have decimated oyster stocks in the Northern Chesapeake Bay (Maryland portion).[1] In signing on to the Chesapeake 2000 Agreement, federal and state management agencies agreed to pursue the goal of a ten-fold increase in the oyster population by 2010, using 1994 as the base year (more rigorous stock assessments began in 1994).

Stephen Kasperski is an economist with the U.S. National Marine Fisheries Service, Alaska Fisheries Science Center, 7600 Sand Point Way NE, Seattle, WA 98012 USA (email: Stephen.Kasperski@noaa.gov). Robert Wieland is an economist with Main Street Economics, P.O. Box 11, Trappe, MD 21673 USA (email: Robert@mainstreeteconomics.com).

The authors thank Jamie King, Chris Dungan, Doug Lipton, John Horowitz, and two anonymous referees for their helpful comments and support. Much of the research on which this paper is based was supported by the NOAA Chesapeake Bay Program Office under their Non-Native Oyster Research Program. A portion of this work was completed while Stephen was a National Marine Fisheries Service/Sea Grant Fellow, and he acknowledges their support. The opinions expressed here represent those of the authors and do not necessarily represent the views of the National Marine Fisheries Service. The authors are responsible for any errors or omissions.

[1] Similar factors have also affected oyster stocks in the more saline southern portion of the Chesapeake Bay. However, these two parts of the Bay are controlled by two different states (Maryland in the north and Virginia in the south) under two very different management regimes. This combination of two different sets of environmental and management factors complicates the discussion of an already complex topic, and for that reason this paper focuses on the northern, Maryland portion of the fishery.

In spite of this agreement, oyster stocks in the Northern Chesapeake Bay have declined almost 30% since 2000 (Barker, Greenhawk, and O'Connell 2008), and the fishery is still managed in large part as an open-access fishery (Wieland 2007).

Somewhat coincident with the decline in oyster stocks, there has been increasing concern over water quality in the Bay as a result of excessive nutrient loads. In estuarine systems, while additional nutrients can result in increases in productivity relative to oceanic systems, excessive amounts cause seasonal oxygen depletion, changes in the community structure, and energy flows through these systems (Lee and Jones 1991; Caddy 1993; Houde and Rutherford 1993; Breitburg *et al.* 1997). Degradation of estuarine systems can result in substantial losses to commercial fisheries (Lipton and Kasperski 2008) as well as recreational users (Freeman 1995). Low levels of dissolved oxygen have also been linked to lower harvests of blue crab in the Chesapeake Bay (Mistiaen, Strand, and Lipton 2003). In response to this problem, the states whose waters drain into the Chesapeake have formed a partnership led by the Environmental Protection Agency and agreed to several protocols aimed at reducing nutrient loads and improving water quality. Achieving these goals was estimated to require expenditure on the order of $1 billion per year for the next ten years (Chesapeake Bay Commission 2003). However, after decades of effort and $6 billion in spending (Fahrenthold 2009), the Chesapeake Executive Council has declared that the current goals will not be met (Wilson 2009).

A restored oyster stock has the potential to substantially improve the Bay's water quality (Cerco and Noel 2007). Oysters remove algae from the water column, producing biomass and waste. With help from other benthic organisms, a portion of the waste produced by oysters is denitrified or buried in bottom sediment, removing substantial amounts of nitrogen and phosphorus from the water column. Given the mandate to improve water quality in the Bay, the ecological services provided by oysters appear to have value.

In this article we present a dynamic stochastic bioeconomic model of the Northern Chesapeake Bay oyster fishery which is used to evaluate the potential value of the oyster resource. The model that is developed is unique in two aspects: it is the first model to incorporate ecological service values of the *in situ* stock in a bioeconomic model of a fishery; and it is also the first forward-looking empirical bioeconomic model of a fishery to evaluate the effects of a moratorium on harvests with the goal of maximizing the net present value of the resource.

The inclusion of non-use benefits in the bioeconomics literature has a long history with respect to the optimal rotation of forest stands (Hartman 1976; Calish, Fight, and Teeguarden 1978; Bowes and Krutilla 1989; Tahvonen and Salo 1999). Additional ecological service values, such as carbon sequestration, are included in bioeconomic models in van Kooten, Binkley, and Delcourt (1995) and Sohngen and Mendelsohn (2003) to determine how the optimal rotation age changes with these values. In contrast to the forest rotation problem, rather than fell an entire stand of trees, a fishery manager must choose the optimal level of partial harvesting (Valderrama and Anderson 2007). The optimal harvest rate is likely dependent on the value of ecological services, which have been largely ignored in the bioeconomic fisheries literature.[2]

Using several approaches for valuing the ecological services benefits from oysters, we find that the optimal harvest rate in the Northern Chesapeake Bay oyster fishery, when accounting for the ecological service value, is always substantially lower than the optimal harvest rate excluding these benefits. We also find that the optimal harvest rate incorporating ecological service values is always lower than the mean harvest rate between 1994–2007.

Clark, Clarke, and Munro (1979) and Herrera (2007) find that instituting a harvest moratorium can be optimal if the stock is in a state of collapse. Valderrama and Anderson (2007) find that a rotational harvest is optimal in the Atlantic sea scallop fishery. The implementation of moratoriums has been analyzed in Grafton, Sandal, and Steinshamn

---

[2] See Knowler (2002) for a survey of bioeconomic models with environmental influences.

(2000) for the Northern cod fisheries, who find the optimal moratorium should have been implemented three years prior to actual implementation, while Nøstbakken (2008) analyzes the North Sea herring moratorium and finds it was lifted too soon to maximize the value of the fishery. Partial closures of fisheries, such as marine protected areas, have been found to be effective in some settings (Holland and Brazee 1996; Brown and Roughgarden 1997; Armstrong 2007), while Hannesson (1998), Smith and Wilen (2003) and Smith, Zhang, and Coleman (2008) find no benefit from partial closures after accounting for responses from the harvesting sector. While these studies indirectly model the effects of a moratorium on harvests, the model developed here will examine the potential of a moratorium to accelerate the transition towards a steady-state stock, consistent with a most rapid approach path (Spence and Starrett 1975), to maximize the net present value of the oyster resource.

Given historically low current oyster abundance, ecological value provided by oysters, and the goal of oyster stock rebuilding in the Chesapeake Bay, the model also predicts that a relatively short harvest moratorium can increase the value of the oyster resource. The optimal moratorium length is sensitive to different parameter assumptions, but a positive length of moratorium (ranging from 4 to 13 years) is found to be optimal in all sensitivity tests.

The Fishery

Beginning in the late 1980s, oyster stocks, harvests, and harvesters all decreased by large percentages. As a result of the decline in oyster stocks, harvests have averaged only .19 million bushels per year between 1994 and 2007 despite sustained annual harvests of approximately 2 million bushels during most of the 1960s and 1970s.

Over the past 25 years, stocks have declined primarily as a result of two parasitic diseases—Dermo and MSX. Both diseases are thought to be of fairly recent origin to the Chesapeake Bay and both can, under certain environmental conditions, cause either baywide or localized epizootics. While some oyster populations appear to have developed some resistance to MSX, both diseases remain a threat to oyster stocks and stock growth in the Chesapeake Bay. In addition to disease effects on stocks, both oyster habitat and water quality have continued to decline, and harvest pressure on remaining stocks is largely unabated. Through much of the history of the Northern Chesapeake Bay oyster fishery, oysters have been managed as an open-access resource, with some licensing, gear, and harvest restrictions (Wieland 2007). While some of those restrictions likely slowed the rate at which rents were extracted from the fishery, they were not sufficient to maintain stocks in the presence of severe disease epizootics.[3]

The Northern Chesapeake Bay oyster fishery is technically a limited-entry fishery, but one with far more licenses than active harvesters. Oysters can be harvested with either an Oyster Harvester License ($50/yr.) or an Unlimited Tidal Fish License ($300/yr.). While both license types limit new entrants to the fishery, in 2006 there were 661 Oyster Harvester Licenses and 2,023 Unlimited Tidal Fish Licenses, but only 317 watermen with a recorded catch above 100 bushels, which at $25 per bushel, is worth approximately $2,500.[4] Oysters are currently harvested with four technologies: hand tongs, patent tongs, diving, and power dredges, all of which require only one to three individuals, a small vessel, and a modest amount of capital.

---

[3] Maryland also had an active management program over part of this period under which oyster spat were moved from more productive areas to grow out on public bars elsewhere in the bay, and this is thought to have boosted stocks over that period.

[4] In 2006, there were no harvesters with a catch greater than 1,300 bushels, which at $25 per bushel means that no harvesters had revenue from oysters over $32,500.

While there are multiple restrictions on harvest in the Northern Chesapeake Bay oyster fishery, none of the restrictions are fully binding, and the resulting fishery has behaved as an open-access fishery. The restrictions include daily harvest limits as well as gear and area restrictions. However, over the period 1989–2006, the annual average daily catch rates were below 50% of the daily catch limits. See Wieland (2007) for an overview of oyster harvesting methods and regulations in the Maryland portion of the Chesapeake Bay. Therefore, throughout the rest of the article, we shall refer to a continuation of the current policy as the "Open Access" policy (*OA*).[5] The value of ecological services provided by oysters and the inability of stocks to recover under the current management plan suggests that the *OA* policy should be evaluated in the context of maximizing the value of the oyster resource with respect to ecological services in addition to harvest value.

Bioeconomic Model

The problem that we analyze is choosing the optimal season to begin harvesting and the subsequent optimal harvest rate to maximize the value (profits from harvest plus the value of ecological services provided by oysters) of the Northern Chesapeake Bay oyster fishery. We consider this over a period of 100 years, after which there is a salvage value given to the remaining stock.[6] For each season with no harvests, the fishery experiences capital, depreciation, and maintenance costs according to the number of active vessels in the fishery during the previous year. For each potential moratorium length, the optimal harvest rate is determined to maximize the value of the oyster resource, subject to the stochastic growth of the oysters in the Northern Chesapeake Bay.

Harvesting Revenue

Let the harvest (h) in any season be equal to the product of the percentage of the stock that is taken (q) and the stock (x) available at the beginning of the season t such that:

$$h_t = q\,x_t, \tag{1}$$

where both h_t and x_t are measured in millions of oysters. We assume that the proportion of the market-sized oyster population that is harvested is the same in all periods. It is this fixed harvest rate that we solve for in the Social Planner (*SP*) model.[7] This fixed harvest rate will be chosen in the bioeconomic model to maximize the value of the oyster resource. The advantage of this formulation of the model is that it allows for simple comparisons of optimal harvest rates across moratoriums of different lengths.

Annual oyster prices are determined through an inverse demand model, such that the annual price per million oysters (p_t) is defined as:

---

[5] After stock rebuilding, it is possible that the daily catch limits would bind for some individuals. However, as there is a surplus of license holders who could enter the fishery, the fishery is likely to continue to behave as an open-access fishery and could continue to harvest at the *OA* rate despite the increased population.

[6] We chose 100 years arbitrarily, but because our non-stochastic model converged to a steady state after approximately 50 years, the additional years are included to allow for multiple high mortality events.

[7] The social planner's choice of optimal harvest rate is equivalent to setting a hard total allowable catch (TAC) on the harvests each period, which is conditional on the current stock abundance. Given the fixed catch per vessel per year assumption in equation (4), each year of the model the social planner can auction off (or give away) the appropriate number of permits such that all of the permitted vessels operating at full capacity will approximately reach the annual TAC and optimal harvest rate for the fishery.

$$p_t = \hat{p}_t - \eta q x_t, \tag{2}$$

where $\hat{p}_t \sim N(\overline{p}, \sigma^2_{\overline{p}})$ is a normally distributed random number with mean $\overline{p} = 73{,}486$ (which is equivalent to \$25.72 per bushel in 2007 dollars) and standard deviation $\sigma^2_{\overline{p}} = 16{,}171$, taken from the period 1994–2007. The value of η is taken from Lipton (2008) and is equal to .011, while qx_t is the total harvest in millions of oysters in year t. This represents a fairly inelastic portion of the demand curve, as production would have to increase to 490 million oysters (or over 70% of the current stock) for the average price per bushel of oysters to drop \$5 to a level below \$20.

The reason that demand is so inelastic with respect to Chesapeake Bay production is that regional production is small compared to the national average, and Maryland and Virginia processed oyster production comes from oysters harvested in other states. In 2003, U.S. production of oysters was 2,800 million bushels (Lipton, Kirkley, and Murray 2006), while Northern Chesapeake Bay production was only .52 million bushels. The majority of oysters coming into the Chesapeake Bay region are from the Gulf of Mexico, which are a near perfect substitute for oysters from the Chesapeake (Murray 2002; Lipton 2008). Given the relatively small production from the Chesapeake Bay and the abundance of imported oysters into the region, it is not surprising that the demand for oysters is so inelastic. Thus, the harvest revenue function is equal to the price times harvest:

$$R(p_t, q, x_t) = p_t q\, x_t. \tag{3}$$

Harvesting Cost

To determine harvesting costs, we assume that all vessels are homogeneous and use the most efficient technology (dredge),[8] and we take the high annual cost estimate (μ = 37,624) for these vessels from Wieland (2006).[9] This annual cost accounts for boat, motor, and gear maintenance, docking and license fees, in addition to capital and depreciation costs, all of which are prorated over a 100-day annual oyster season.

We determine the number of boats active in the fishery each season given harvest levels, to be equal to the total harvest in bushels divided by each boat's expected seasonal catch:

$$b_t = \frac{\alpha}{\beta} q x_t, \tag{4}$$

where α = (1,000,000/350) is a conversion factor from millions of oysters to bushels of oysters, assuming 350 oysters per bushel, and β = (50*100) is the expected seasonal catch

---

[8] We ignore any habitat effects from widespread use of dredges on the oyster reef height and other benthic habitat. An interesting area for further research would be to include these habitat externalities on the efficiency of each harvesting technology.

[9] While the homogeneity assumption is likely not valid in this fishery, harvesting costs are likely to be substantially higher for the other harvesting technologies. This will cause the optimal harvest rates to be artificially lower and more likely optimal to institute a moratorium than if the fishery were being harvested with the most efficient technology. For this reason, we chose to focus on the efficiency of the two policies under the best possible circumstances for the *OA* policy.

in bushels which is equal to the expected daily harvest (50 bushels/day) times the season length (100 days).[10]

The total direct cost of any level of harvest (c_t) is equal to the number of boats active in the fishery times the annual cost of maintaining a boat over the 100-day oyster season:

$$c_t = \mu b_t = \frac{\alpha\mu}{\beta} q x_t. \tag{5}$$

In addition to the direct costs of harvesting (c_t), we assume the cost of harvest is decreasing in the stock of market-sized oysters and increasing in the total annual harvest. These assumptions are valid in the case of the Maryland oyster fishery. The larger the number of oysters available for harvest, the less effort it will take to achieve any level of harvest. Similarly, to harvest more oysters, more time out on the water has to be expended to catch them. We specify a cost function that is equal to the direct harvesting costs plus an adjustment for the harvesting costs, which decreases as the stock approaches the carrying capacity:

$$C(q,x_t) = c_t\left(1+\frac{k-x_t}{x_t^2}\right) = \frac{\alpha\,\mu}{\beta} q x_t\left(1+\frac{k-x_t}{x_t^2}\right), \tag{6}$$

where $k = 5{,}089.2$ is the fixed carrying capacity of the Northern Chesapeake Bay in millions of market-sized oysters (Jordan and Coakley 2004).

Ecological Value of Oysters

In addition to their harvest value, the stock of oysters *in situ* provides substantial ecological benefits to the Northern Chesapeake Bay. Oysters play an important water quality role, filtering out particles that block the movement of sunlight through the water column thus increasing the amount of light that reaches submerged aquatic vegetation and, consequently, improving benthic habitat for species that need places to hide (Coen *et al.* 2007; Fulford *et al.* 2007; and Grabowski and Peterson 2007). The habitat created by the submerged aquatic vegetation is thought to create a very large value through its role in recreational fisheries (Kahn and Kemp 1985). In their filtering, oysters also concentrate nutrients in pseudo-feces, which, with help from other benthic organisms, are either buried in sediment or denitrified out of the water column (Newell *et al.* 2005).

Oysters could help in achieving the Chesapeake 2000 Agreement goals of improving the water quality in the Chesapeake Bay through their ability to remove nitrogen and phosphorus from the water column. In the Chesapeake Bay, a million oysters can reduce approximately 753 kg of nitrogen and 272 kg of phosphorus, on average, from the water column per year (Newell *et al.* 2005). Estimates of the cost of reducing a kilogram of nitrogen delivered to the Bay range from \$4.6 for planting cover crops to \$1,125 for erosion

---

[10] This is equivalent to assuming a constant catch per unit effort, which may be valid when dredging over a uniformly distributed oyster bar. However, this assumption is relaxed via the construction of the cost function in equation (6).

and sediment control measures, but has been estimated to average \$24.07 per kilogram of nitrogen removed (Newell *et al.* 2005). Using the average value of nitrogen removal, this implies an ecological services value (v) of \$18,135.69 per million oysters filtering nitrogen out of the water of the Chesapeake Bay per year.[11] We use this ecological services value as a low estimate of the current ecological value of oysters in the Chesapeake Bay, since it ignores the value of phosphorus removals, as well as the other benefits oysters provide to recreational fisheries (Hicks, Haab, and Lipton 2004) and other users of the Bay.

However, oysters' filtering capacity increases at a decreasing rate, so that each additional oyster will filter less nitrogen out of the water (Cerco and Noel 2007). In addition to the declining effectiveness of oysters' filtering capacity, the value of nitrogen removals is likely decreasing with improvements in nitrogen concentrations in the Bay. As the Bay becomes clearer, the majority of the value to improvements in recreational fishing and boating will already have been experienced, and additional clarity is less valuable. Both of these factors imply that the ecological value of oysters is decreasing in the cumulative amount of oysters in the Bay. We, therefore, estimate the ecological value ($EV_1(x_t)$) of the standing oyster stock as:

$$EV_1\left(x_t\right)=vx_t\left(x_t\Big/\sum_{t=1}^{t}x_t\right). \tag{7}$$

We denote the above ecological value function EV_1, which is a reasonable lower bound on the ecological service values provided by oysters, to contrast it with two additional ecological valuation functions below. The second ecological valuation function holds the ecological value of oysters at a constant value per million oysters. The rationale for this is twofold. First, for the ecological value to be decreasing in the cumulative stock of oysters, the population of oysters needs to be large enough to not only remove the nitrogen being introduced into the Bay each year, but also decrease the concentration of nitrogen in the Bay. Given the current state of nitrogen introductions to the Bay, it is unclear whether or not any amount of oysters will be able to reduce the concentration of nitrogen to a level where the oysters exhibit diminishing marginal productivity with respect to nitrogen filtration services. Secondly, removing nitrogen from the water is one of many services they provide which has ecological value in the Chesapeake Bay. Therefore, the value of nitrogen reductions from the Bay could be conceived as a lower bound estimate for the value of the ecological services provided by the oysters. While it is possible that the value of nitrogen reductions will decrease with cumulative oyster stocks, it is possible that the benefits to recreational fishing, boating, and swimming may increase over time. Thus, we calculated the $EV_2(x_t)$ function to be linear in ecological service value:

$$EV_2\left(x_t\right)=vx_t. \tag{8}$$

The first two ecological service functions represent two extremes as $EV_1(x_t)$ quickly declines toward zero as the cumulative stock of oysters increases, while $EV_2(x_t)$ increases linearly with the stock and ecological service values quickly overshadow any benefit from harvesting. Therefore, we construct the third ecological services function, $EV_3(x_t)$, to be a function of the current stock size relative to a moving sum of the stock over the past five seasons, such that:

---

[11] This ecological value corresponds to \$6.28 per bushel of oysters, which is over a quarter of the average value of a bushel of harvested oysters (\$25.72) over the period 1994–2007.

$$EV_3(x_t) = \begin{cases} \mathrm{v}x_t \left(x_t \Big/ \sum_{t=1}^{t} x_t \right) & \text{if} \quad t \le 5 \\ \mathrm{v}x_t \left(x_t \Big/ \sum_{t-5}^{t-1} x_t \right) & \text{if} \quad t > 5 \end{cases} . \tag{9}$$

These three ecological service functions are presented in figure 1 under a non-stochastic, no harvest model of stock dynamics. We run the model with the three ecological value functions, in addition to a model which sets the value of ecological services to zero (EV_0), to evaluate the impact different ecological valuations have on the optimal harvest rates and moratorium length in this fishery.

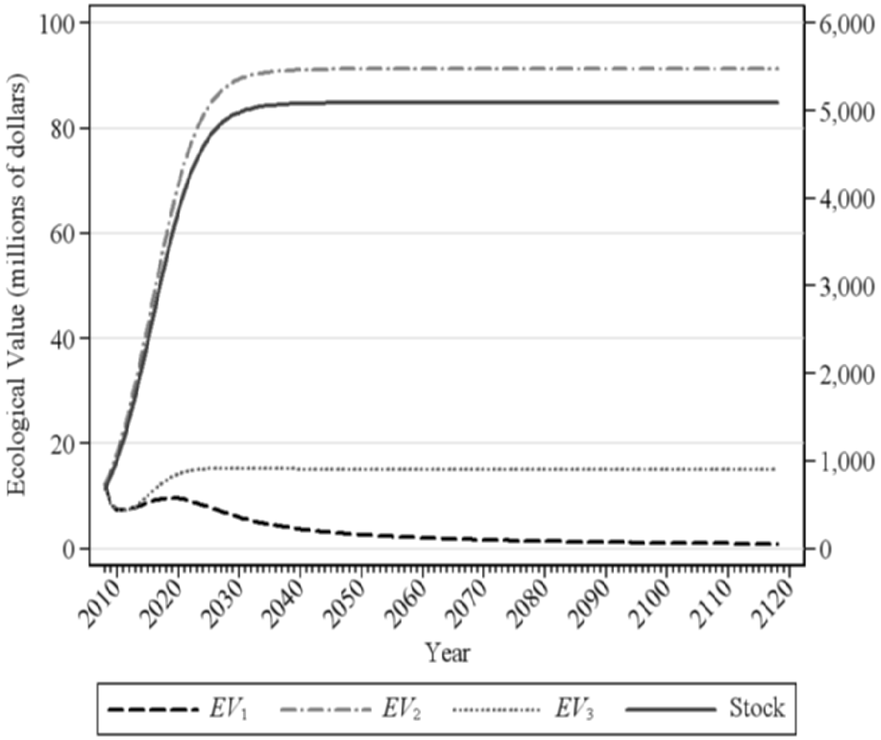

Figure 1. Annual Value of Ecological Services using Different Ecological Value Functions with a Non-stochastic, No Harvest Model of Oyster Stock Dynamics

Note: As the stock quickly approaches the carrying capacity (5,089 million oysters), the EV_1 function quickly declines toward zero, the EV_3 function stays relatively constant around \$15 million, while the EV_2 function provides a value of over \$90 million each year.

Salvage Value

After the end of the model's 100-year time frame (T=101), we assume that there is a salvage value ($\lambda(x_T)$) of the oyster stock equal to the profits from harvesting the steady-state growth in perpetuity plus the ecological service value corresponding with the ending stock size in perpetuity:

$$\lambda(x_T) = \left(R(p_T, q, x_T) - C(q, x_T) + EV(x_T)\right)\delta^{-1}(1+\delta)^{-T}, \tag{10}$$

such that $x_{T+1} = x_T$ and δ =.04 is the social rate of discount. This salvage value function provides an incentive to keep the stock at a high level at the end of the model so that steady-state harvests and ecological services in the future will be larger. This effect will be countervailed by the fact that these values occur after 100-years, and discounting will cause these harvests to be worth less in net present value terms.

Moratorium Costs

We evaluate the optimal season to begin harvesting (t*) because the stock is currently at historically low levels, and it may be optimal to allow the stock to rebuild for a number of years without harvest morality. In the years prior to t*, each vessel participating in the fishery experiences a prorated annual moratorium cost of (κ = 6,127), which accounts for maintenance costs for the vessels, dock and license fees, depreciation, and capital costs equal to \$15,929 prorated over a 100-day oyster season of a possible 260 possible days out on the water in the Northern Chesapeake Bay (Wieland 2006).[12] Therefore, the cost for each year of a moratorium (M_t) is equal to the number of boats operating in the previous year times the prorated annual moratorium cost, such that:

$$M_t = \kappa b_{t-1}. \tag{11}$$

We ignore the costs of a moratorium on the processing sector as Chesapeake Bay production is small relative to the national average, and processors are currently importing the majority of their oysters from outside the region (Murray 2002).

Given the functional form of the revenue and cost functions, we assume that if the stock falls below 1% of carrying capacity (50.89 million oysters), the fishery closes because profits from harvest will be negative. For all periods where the stock is below this level, the fishery experiences the moratorium cost of M_t until the stock recovers above 50.89 million oysters such that profits from harvest are again positive.

---

[12] We assume there is no depreciation in human capital resulting from a fishing moratorium, as knowledge of spatially explicit oyster populations is transparent and readily accessible. The natural oyster bars were delineated in 1913 and since then publicly available information on oyster bar health has been provided by the Bay Bottom Survey (1974–1983) (Smith *et al*. 2001; Smith *et al*. 2005). While it is likely that watermen in aggregate have more extensive knowledge of sites with harvest potential in any given year, the information available to new and potential entrants about populations at the bar level (since oysters are sessile) provides similar knowledge as to the whereabouts of potentially harvestable oysters. Therefore, it is likely that the loss of human capital will be minimal.

The Total Value of the Oyster Resource

We now define the total value of the oyster resource (V) to be a combination of the moratorium costs experienced until period $t^*–1$, profits from oyster harvesting from period t^* through $T–1$, plus the ecological service value provided by the stock of oysters from period 1 through $T–1$, all discounted at the social rate of discount, plus the salvage value of the stock at the end of the simulation, such that:

$$V=\left(\sum_{t=1}^{t^*-1}-M_t+\sum_{t^*}^{T-1}\left(R(p_t,q,x_t)-C(q,x_t)\right)+\sum_{t=1}^{T-1}EV(x_t)\right)\left(1+\delta\right)^{-t}+\lambda\left(x_T\right). \quad (12)$$

The objective of the managers (social planners) is to maximize equation (12) by choosing the optimal period to start harvesting (t^*) and optimal constant harvest rate (q), subject to oyster stock dynamics.

Oyster Stock Dynamics

There is significant debate within the scientific community over how to best specify the stock dynamics of oysters in the Chesapeake Bay. The most detailed oyster demographic model is developed in Vølstad *et al.* (2007) for the Final Programmatic Environmental Impact Statement for Oyster Restoration in Chesapeake Bay Including the Use of a Native and/or Nonnative Oyster. However, the model was largely rejected by the Oyster Advisory Panel due to the lack of adequate data to parameterize the model, and no suitable alternative currently exists. Therefore, in this article we use a counterfactual model developed in Wieland and Kasperski (2008) to calculate the intrinsic growth rate (r) of the oyster stock and take a carrying capacity (k) estimate from the existing biological literature in a standard surplus production growth model. While this approach ignores the role of environmental influences (Liddel 2007) and larval transport on oyster growth (North *et al.* 2006), it does include the impact of high-mortality events (Vølstad *et al.* 2007), and it tracks the market-sized population fairly accurately over the period 1994–2007, as shown by figure 2.

We take the average of the positive net recruitment ratios from table 1 as our estimate for the intrinsic growth rate of the Maryland market-sized oyster population (r = .239).[13] The five years with negative recruitment will be included in the model as described below. We model the population dynamics of the market-sized oyster population as partially following a Ricker growth function of the population of market-sized oysters in the previous period. This model is similar to Jordan and Coakley (2004), and we also use their estimate of the carrying capacity of market-sized oysters in Maryland.[14]

These growth parameters remain constant throughout the simulations. However, random population events are introduced as a function of the parameters α_t and γ. We assume that α_t is a random number uniformly distributed between zero and one. A new α_t is gen-

---

[13] In addition to using the counterfactual model, we also estimated the intrinsic growth rate by calibrating our oyster stock dynamics model to minimize the sum of squared error between the actual stock over the past 14 years and the model's predicted stock, assuming the timing of negative mortality events are known. The calibrated growth rate was equal to r =.235, which is very similar to our counterfactual estimate of r = .239.

[14] The estimate by Jordan and Coakley (2004) of the suitable habitat area comes from the Maryland bay bottom survey, which indicated that only 10% of nominal oyster habitat actually supported oyster populations (Smith *et al.* 2001). For their carrying capacity estimate, they assumed that 10% of cultch areas supported 10 market-sized oysters per m^2, and 10% of sand and cultch and mud and cultch areas supported three market-sized oysters per m^2.

erated for each year of the model, and α_t also varies between simulations. If α_t is greater than our critical value γ, the stock grows according to its Ricker growth function, with $r = .239$ and $k = 5{,}089$ million market-sized oysters. However, if $\alpha_t < \gamma$, this implies that the stock experiences a high mortality event, and the stock falls to φ% (where $0 < \varphi < 1$) of last year's stock, minus fishing mortality during that season. We let φ equal one plus the average negative recruitment ratio of the five high mortality years in our data, such that $\varphi = .83$. Thus, when there is a high mortality event, the oyster abundance falls to 83% of the previous year's stock minus any oysters that were harvested that season. As Maryland's portion of Chesapeake Bay oyster stocks has experienced natural mortality that exceeded recruitment in five of the past fourteen years (table 1), we set $\gamma = 5/14$.

Figure 2. Actual Vs. Predicted Oyster Stock

Note: The predicted stock of oysters is modeled as following the oyster stock dynamics model in equation (13) over the period 1994–2007, assuming the timing of high mortality events is known.

We assume that in normal mortality years the Maryland market-sized oysters grow according to a Ricker escapement growth model. We chose to focus the model on escapement for a number of reasons. Firstly, stock estimates come from Maryland Department of Natural Resources' (DNR) fall survey, and harvests occur during the winter months when the stock is thought to grow very little. Once the season ends, the remaining oysters are left to grow during the summer until the next season when they will again be surveyed in the fall. Those oysters that are reproducing between seasons are only those that have escaped harvest in the past year, and thus an escapement model seems to approximate reality. Therefore, we define the oyster stock dynamics as:

$$
\begin{aligned}
&x_{t+1} = (x_t\text{-}qx_t)\exp\left\{r\left(1\text{-}\frac{(x_t\text{-}qx_t)}{k}\right)\right\} && \text{if } \alpha_t \geq \gamma \\
&x_{t+1} = \varphi x_t - qx_t && \text{if } \alpha_t < \gamma \\
&\forall t = 1,\ldots,T-1, \text{ where } \alpha_t \sim \mathrm{U}[0,1].
\end{aligned}
\tag{13}
$$

Table 1

Stock Change (Million Market-sized Oysters*) in the Absence of Harvests

Year	Stocks	Harvests	Disease Mortality (%)	Net Recruits	Net Recruitment Ratio (%)
1994	738.96	27.85	20.42		
1995	905.52	58.21	25.23	210.08	28.43
1996	948.11	70.27	23.53	96.33	10.64
1997	958.40	61.90	13.45	63.86	6.74
1998	1,065.04	99.60	15.98	190.33	19.86
1999	1,121.92	148.79	26.40	166.38	15.62
2000	956.08	118.72	32.05	–85.17	–7.59
2001	638.58	120.06	39.05	–244.32	–25.55
2002	385.73	51.44	52.04	–228.19	–35.73
2003	511.20	18.17	36.87	136.94	35.50
2004	635.29	7.57	17.26	130.35	25.50
2005	925.87	22.98	12.81	310.62	48.89
2006	774.78	45.67	14.06	–111.83	–12.08
2007	684.02	57.54	20.65	–45.10	–5.82

* Oysters greater than 77 mm in length.

Maximizing the Net Present Value of the Oyster Fishery

Using this growth equation for market-sized oysters, the problem that we analyze is the maximization of the net present value of the Maryland oyster fishery over a period of 100 years. The manager's problem is to choose the optimal period to begin harvesting (t^*) and the optimal harvest rate (q) to maximize the expected value of the oyster resource from equation (12), subject to the stock dynamics from equation (13):

$$
\begin{aligned}
&\underset{t^*,\, q}{Max} \quad E\left\{\left(\sum_{t=1}^{t^*-1} -M_t + \sum_{t^*}^{T-1}\big(R(p_t,q,x_t) - C(q,x_t)\big) + \sum_{t=1}^{T-1} EV(x_t)\right)(1+\delta)^{-t} + \lambda(x_T)\right\} \\
&\text{s.t. } x_{t+1} = (x_t\text{-}qx_t)\exp\left\{r\left(1\text{-}\frac{(x_t\text{-}qx_t)}{k}\right)\right\} \qquad \text{if } \alpha_t \geq \gamma \\
&\qquad x_{t+1} = \varphi x_t - qx_t \qquad \text{if } \alpha_t < \gamma \\
&\qquad \forall t = 1,\ldots,T-1,
\end{aligned}
\tag{14}
$$

where $E\{\bullet\}$ represents the expectation operator.

The specific functional forms used here, which were developed to approximate the biological and economic realities of the oyster resource, do not provide a convenient analytical solution to the problem. Therefore, we rely on numerical optimization and Monte Carlo simulations to characterize a solution. Starting with no harvest moratorium (t^*=1), for each simulation a random draw of $\alpha_t,....,\alpha_T$ and $\hat{p}_t,....,\hat{p}_T$ parameters are generated, and the optimal harvesting percentage (q) which maximizes the net present value of the oyster resource is determined using the interior-point algorithm of the Matlab R2009a function *fmincon.*[15] This approach is similar to setting some constant target fishing mortality rate common to the yield per recruit management targets prevalent in many fisheries (Hilborn and Walters 1992). We will refer to the solution to this optimization procedure as the *SP*'s policy from this point onward.

Results

No Harvest Moratorium

The median value of the 10,000 simulations with no harvest moratorium (t^*=1) for each of the *SP* models and the *OA* models with three different ecological service functions and a no ecological value scenario ($SP–EV_1$, $SP–EV_2$, $SP–EV_3$, and $SP–EV_0$) and ($OA–EV_1$, $OA–EV_2$, $OA–EV_3$, and $OA–EV_0$), respectively, are presented in table 2. The optimal harvest rate from the $SP–EV_2$ model (the highest value for ecological services) is less than the $SP–EV_3$ optimal harvest rate (the medium value for ecological services), which is less

Table 2

Net Present Value with no Moratorium under Different Management Regimes

Model	Net Present Value* ($)	Profit from Harvest* ($)	Ecological Value of *In-situ* Stock* ($)	Steady-state Value of Harvest* ($)	Steady-state Value of *In-situ* Stock* ($)	Optimal Harvest Rate† (%)	Steady-state Stock‡
$SP–EV_0$	65.82	65.68	0.00	0.14	0.00	5.34	813.06
$SP–EV_1$	145.21	35.87	108.76	0.23	0.35	1.44	1,975.64
$SP–EV_2$	1,259.51	13.94	1,202.29	0.45	42.83	0.30	2,851.05
$SP–EV_3$	318.65	18.29	290.44	0.68	9.23	0.40	2,675.49
$OA–EV_0$	53.54	53.16	0.00	0.38	0.00	8.05	193.10
$OA–EV_1$	98.61	63.84	34.31	0.45	0.01	8.05	193.10
$OA–EV_2$	321.41	56.52	264.48	0.40	0.01	8.05	193.10
$OA–EV_3$	128.09	61.36	66.27	0.44	0.01	8.05	193.10

Note: The values in the table represent the median value from 10,000 simulations.
† The harvest rate for the *OA* models is not optimized and is set equal to the mean harvest rate of the period 1994-2007; * in millions of 2007 dollars; ‡ in millions of market-sized oysters.

---

[15] The interior point algorithm was implemented using a termination tolerance on the net present value (TolFun) of 1e–20, a termination tolerance on the optimal harvest rate (TolX) of 1e–25, and constraint tolerance on the optimal harvest rate (TolCon) of 1e-20.

than the $SP–EV_1$ optimal harvest rate (the lower bound value for ecological services), all of which are lower than the optimal harvest rate excluding ecological values ($SP–EV_0$). Thus, the higher value placed on ecological service values, the lower the optimal harvest rate. Similarly, the steady-state stock is the lowest for the *OA* models, followed by the $SP–EV_0$ model, the $SP–EV_1$ model, the $SP–EV_3$ model, and the highest for the $SP–EV_2$ model.

Table 2 also demonstrates the inefficiency of a continuation of the *OA* policy. Not only does the stock decline to less than half of the current population, but by reducing the harvest rate, the social planner increases the net present value of the fishery by 23% with no ecological values ($OA–EV_0$ vs. $SP–EV_0$), 47% with Ecological Service Function 1 ($OA–EV_1$ vs. $SP–EV_1$), 292% with Ecological Service Function 2 ($OA–EV_2$ vs. $SP–EV_2$), and 149% with Ecological Service Function 3 ($OA–EV_3$ vs. $SP–EV_3$).

Optimal Harvest Delay

To determine whether a moratorium on harvest will increase the net present value of the fishery, we alter the starting harvest period such that $t^* = 2,\ldots,25$. For each starting harvest year, the same optimization procedure as above is run to determine the optimal harvest rate (q) as well as the net present value of the oyster resource. The median values of the 10,000 simulations are presented in table 3, which suggests that in this fishery a moratorium is optimal for all models included—with and without an ecological services component. The optimal moratorium length for the social planner's problem is determined to be that which maximizes the median net present value from all 10,000 simulations and ranges between 4 and 13 years.

Comparing the net present value from $SP–EV_1$ from tables 2 and 3 suggests that by imposing a 10-year moratorium, the resulting stock provides ecological services that will increase the value of the resource by over $17 million. Accounting for capital and depreciation costs, as well as vessel maintenance and fees to keep the vessel active throughout the moratorium, profits from harvest accruing to the watermen remain relatively unchanged. These results are consistent with a "most rapid approach path" or "bang-bang" transition to the steady-state stock level to maximize the value of the resource (Clark and Munro 1975; Spence and Starrett 1975; Clark 1990).

Similar to the no harvest moratorium scenario, table 3 shows that the steady-state stock and optimal harvest rate are increasing and decreasing, respectively, in the ecological values included in the model such that the $OA–EV_0$ model has the lowest steady-state stock and highest harvest rate, and $SP–EV_2$ has the highest steady-state stock and lowest harvest rate. The average optimal constant harvest rates for each potential moratorium year (held constant from the first year of harvest through period (T–1)) are shown in figure 3. The optimal harvest rate for the social planner's models are always below the *OA* harvest rates for all ecological service values, which suggests that the *OA* policy permits excessive fishing pressure and does not maximize the net present value of the oyster resource as a whole.

The distributional impact of a moratorium and limitation on total harvest between profits from harvest, the ecological services value, and steady-state stock and harvests are also presented in tables 2 and 3. Moving from the current 8.05% harvest rate with no moratorium ($OA–EV_1$ from table 2) to the social planner's optimal moratorium length of 10 years and a harvest rate of 3.64% ($SP–EV_1$ from table 3) results in a $28 million decrease in profits for watermen, $92 million increase in ecological service values accruing to all users of the Bay, and an increase in the steady-state stock of market-sized oysters of 1,119 million.

The total net present value as well as the ecological services component and the profit from harvest component of the net present value estimates for all potential moratorium lengths for the $SP–EV_1$ and $OA–EV_1$ models are presented in figures 4 and 5, respectively.

Table 3
Optimal Moratorium Length under Different Management Regimes

Model	Optimal Moratorium Length (yrs.)	Net Present Value[*] ($)	Profit from Harvest[*] ($)	Ecological Value of *In-situ* Stock[*] ($)	Steady-state Value of Harvest[*] ($)	Steady-state Value of *In-situ* Stock[*] ($)	Optimal Harvest Rate[†] (%)	Steady-state Stock[‡]
$SP–EV_0$	8	84.07	83.94	0.00	0.13	0.00	7.30	417.42
$SP–EV_1$	10	162.23	36.00	125.93	0.11	0.19	3.64	1,312.25
$SP–EV_2$	13	1,268.04	9.28	1,217.48	0.38	40.89	0.32	2,719.90
$SP–EV_3$	4	320.19	18.62	291.99	0.41	9.16	0.40	2,552.25
$OA–EV_0$	4	55.45	55.04	0.00	0.40	0.00	8.05	209.94
$OA–EV_1$	14	115.17	57.41	57.19	0.55	0.01	8.05	255.92
$OA–EV_2$	24	610.11	41.14	568.36	0.60	0.01	8.05	313.20
$OA–EV_3$	24	184.03	43.25	140.14	0.63	0.01	8.05	313.20

Note: The values in the table represent the median value from 10,000 simulations.
[†] The harvest rate for the *OA* models is not optimized and is set equal to the mean harvest rate of the period 1994–2007; [*] in millions of 2007 dollars; [‡] in millions of market-sized oysters.

Figure 4 suggests that the lower harvest rate in the *SP–EV*$_1$ model results in ecological services values which are higher than the entire net present value for the *OA–EV*$_1$ model. Given the large proportion of the net present value that comes from the ecological services component of *SP–EV*$_1$ model, it is not surprising that figure 5 shows that profits from harvest for the *SP–EV*$_1$ model are less than those from the *OA–EV*$_1$ model.

Figure 6 displays the median steady-state stock for the *OA–EV*$_0$, *SP–EV*$_0$, *SP–EV*$_1$, *SP–EV*$_2$, and *SP–EV*$_3$ models, which shows that the increase in profits from harvest from the *OA* models comes at a cost of reducing the steady-state stock relative to the social planner's models. The *OA–EV*$_0$ model results in a fairly constant steady-state stock of around 300 million oysters, which is a net decline in the oyster stock since the beginning of the model. On the other hand, all of the social planner's policies (with the exception of the *SP–EV*$_0$ model) result in an increased steady-state stock. There is a slight decline in the steady-state stock with increases in moratorium length, which is a result of increasing harvest rates in those periods to increase the value of the fishery after a very long period of not harvesting.

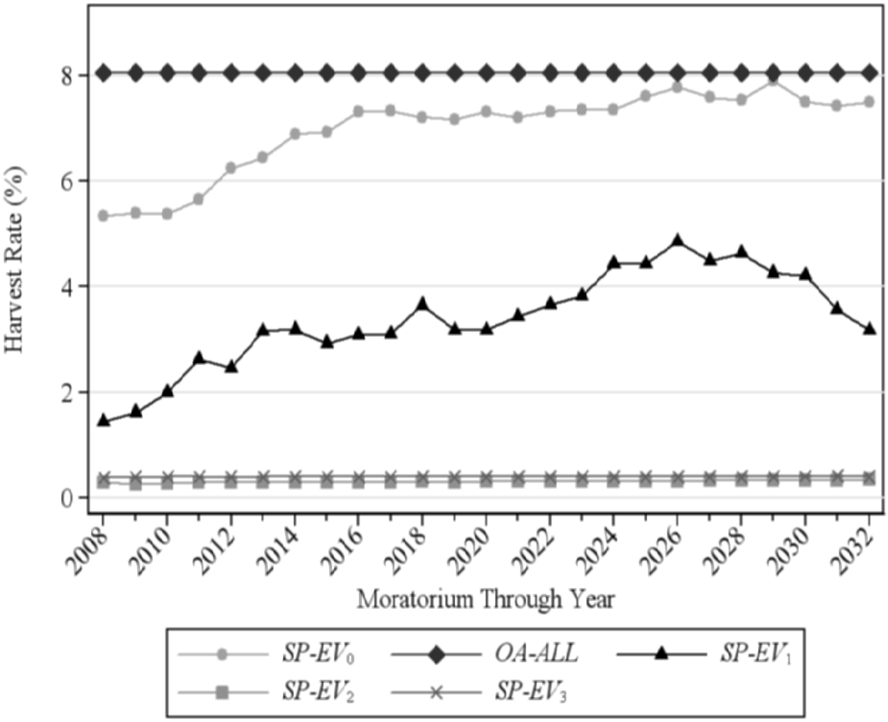

Figure 3. Optimal Harvest Rates under Different Ecological Value Functions for Moratoriums of Different Lengths

Note: The x-axis represents a moratorium through that year such that harvesting begins in the following year. Thus, 2008 represents the no-moratorium scenario. The y-axis represents the median optimal harvest rate held constant from the first year of harvest through period (*T*-1).

Figures 4 and 5 also show that the *SP–EV*$_1$ model has a higher net present value than the *OA–EV*$_1$ model for all possible moratorium lengths. The social planner's models reduce the harvest rate during short moratoriums to allow the stock to increase and also

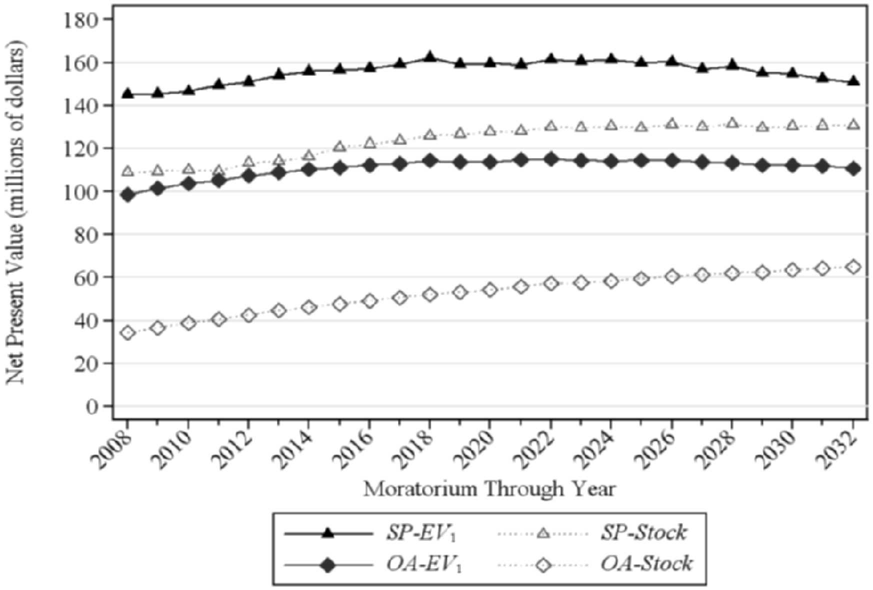

Figure 4. The Ecological Value Component of Net Present Value (*Stock*) and Total Net Present Value (EV_1) under the *OA* and *SP* Models for Moratoriums of Different Lengths

Note: The x-axis represents a moratorium through that year such that harvesting begins in the following year. Thus, 2008 represents the no-moratorium scenario.

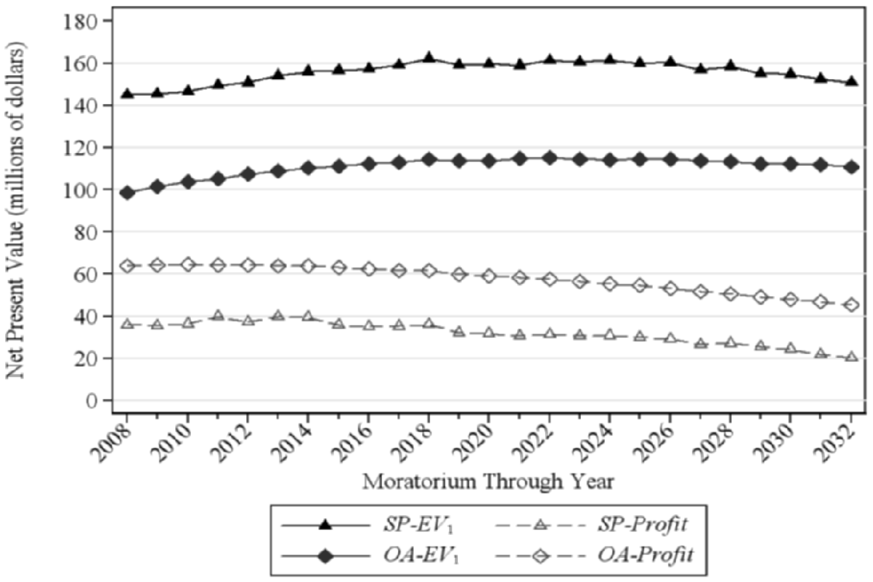

Figure 5. The Profit from Harvest Component of Net Present Value (*Profit*) and Total Net Present Value (EV_1) under the *OA* and *SP* Models for Moratoriums of Different Lengths

Note: The x-axis represents a moratorium through that year such that harvesting begins in the following year. Thus, 2008 represents the no-moratorium scenario.

increase the harvest rate after long moratoriums after which the stock is approaching the carrying capacity. The net present value of the *SP–EV*$_1$ and *OA–EV*$_1$ models is generally concave in figures 4 and 5, which suggests that even fairly short moratoriums can increase the net present value of the fishery. While there might not be the political will to close the fishery for the full 10 years as suggested by the *SP–EV*$_1$ model, our model predicts that any shorter length of moratorium with appropriate limitations on total harvest can significantly increase the value of the resource, regardless of management regime. Given the historically low population level, a positive optimal moratorium length is consistent with Herrera (2007), who finds that closures can be optimal in the first best case if the resource is overexploited and Clark, Clarke, and Munro (1979), who suggest that a brief moratorium might be optimal in severely depleted fisheries.

The model suggests that when the ecological services provided by oysters are valued, including these values in the bioeconomic model results in dramatic changes in the optimal policy. Depending on the value assigned to the ecological benefits provided by oysters, the model suggests a moratorium between 8 years with no ecological values (*SP–EV*$_0$) and 13 years with linear ecological values (*SP–EV*$_2$) with harvest rates of 7.30% and 0.32%, respectively. Including ecological services values also suggests that the steady-state stock should be higher than when these values are excluded. As we know, oysters do provide ecological services, this provides one argument for taking a more precautionary approach towards management and limiting harvests below the optimal harvest rate for the *SP–EV*$_1$ model (the lower bound on ecological values) of 1.44% with no moratorium and 3.64% with a ten-year moratorium.

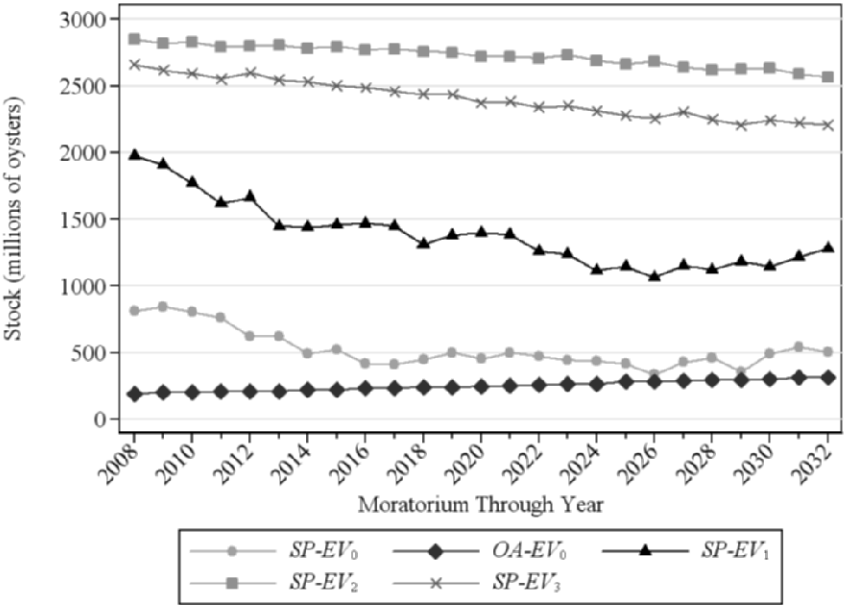

Figure 6. Average Steady-state Stocks under the *OA* and *SP* Models for Moratoriums of Different Lengths

Note: The x-axis represents a moratorium through that year such that harvesting begins in the following year. Thus, 2008 represents the no-moratorium scenario.

Sensitivity Tests

As the conclusions we can draw from our model are largely dependent on the parameters used, it is important to test the sensitivity of our model to changes in four important parameters, the: intrinsic growth rate, discount rate, carrying capacity, and starting population. The results of the sensitivity tests, all of which include the value of ecological services from the EV_1 function, are shown in table 4.

Given the current state of the oyster population in the Chesapeake Bay and the increasing prevalence of parasitic diseases, it is possible that the current oyster population cannot grow as fast as it has in the past. It is also possible that we have underestimated the oyster's growth potential in our calculation of the growth rate. Therefore, as a sensitivity test, we run the model with one half ($r = .12$) as well as two times ($r = .48$) the original intrinsic growth rate. In the model with half the intrinsic growth rate ($r = .12$), after taking the probability of high mortality events into account, the expected growth rate is only 1%; thus it is generally optimal to harvest the stock of oysters to extinction since they are growing at a rate far slower than the discount rate. With a doubled intrinsic growth rate ($r = .48$), the optimal moratorium is six years, the steady-state stock is higher, and the harvest rate is 8.06%, which is nearly identical to the *OA* harvest rate.

Due to the long time frame of this study, the discount rate (δ) plays an important role in determining the optimal harvest rates in the future. In the initial analysis, we chose a relatively modest social discount rate of $\delta = .04$. We choose two additional discount rates to test the sensitivity of our model results to the discount rate: a low discount rate scenario ($\delta = .02$) and a high discount rate scenario ($\delta = .07$). The low discount rate scenario results in a longer moratorium and lower harvest rate, as the future is discounted less heavily. The high discount rate imposes the same 10-year moratorium, but a higher harvest rate as immediate profits are relatively more valuable.

As a result of current and potential future habitat degradation, it is possible that we have overestimated the carrying capacity of market-sized oysters in the Northern Chesapeake Bay. Reducing the carrying capacity[16] to k=3,678.2 does not change the optimal moratorium length, but increases the optimal harvest rate, resulting in a lower steady-state stock.

It has also been suggested that the carrying capacity of a sessile species, such as the oyster, is a function of the population due to the habitat they create for themselves. As an additional sensitivity test, we create a linear model of carrying capacity as a function of the stock of oysters in 2001,[17] such that $k_{2001} = 5{,}089.2 = \phi x_{2001}$. This implies a value of 7.97 for ϕ. Using this linear model, the carrying capacity quickly reaches nonsensical levels, so we impose a ceiling on the carrying capacity such that it does not increase above 33,440 million oysters and denote this model as the high carrying capacity model. This number corresponds with Newell's (1988) estimate of Maryland's oyster populations before 1870, multiplied by the average percentage of market-sized oysters taken from DNR's fall survey over the period 1994–2007.[18] Modeling the carrying capacity as a function of the stock has the effect of shortening the moratorium length but lowering the optimal harvest rate as the stock is allowed to grow much larger in this scenario and provides substantial ecological services value.

DNR has estimated that the stock in 2007 was 684 million market-sized oysters or 13% of carrying capacity. While the science of oyster stock assessments in the North-

---

[16] This estimate is the sum of the carrying capacity of the low, medium, and high salinity zones in the Maryland portion of the Chesapeake Bay from Jordan and Coakley (2004). This estimate is lower than the total Maryland carrying capacity because they lacked salinity data for all areas with oyster habitat.

[17] 2001 is used because it is the year Jordan and Coakley (2004) use as their basis to estimate the carrying capacity of 5,089.2 million oysters.

[18] This estimate comes from Newell's (1988) estimated standing stock of 229 million bushels of oysters, multiplied by 350 oysters per bushel. Assuming the average size distribution over the period 1994–2007 from DNR's fall survey (42% are market sized), the carrying capacity estimate is $k = 33{,}440$ million market-sized oysters.

Table 4
Optimal Moratorium Length under Different Parameter Assumptions

Model	Optimal Moratorium Length (yrs.)	Net Present Value* ($)	Profit from Harvest* ($)	Ecological Value of *In-situ* Stock* ($)	Steady-state Value of Harvest* ($)	Steady-state Value of *In-situ* Stock* ($)	Optimal Harvest Rate (%)	Steady-state Stock‡
$SP\text{-}EV_1$	10	162.23	36.00	125.93	0.11	0.19	3.64	1,312.25
Half Growth Rate (r =.12)	14	86.37	9.69	76.68	0.00	0.00	3.38	56.39
Double Growth Rate (r =.48)	6	351.41	144.10	205.49	1.23	0.60	8.06	3,236.12
Low Discount Rate (δ =.02)	21	241.08	61.91	174.46	2.09	2.63	4.17	1,346.61
High Discount Rate (δ =.07)	10	115.03	19.43	95.59	0.00	0.01	3.94	1,175.95
Low Carrying Capacity (k = 3,678.2)	10	140.07	33.29	106.57	0.11	0.09	4.78	787.66
High Carrying Capacity (k = 33,440)	1	446.12	59.81	372.59	4.17	9.54	0.36	13,011.82
Starting Population =.25k	5	208.96	56.08	152.65	0.09	0.14	4.09	1,252.37
Starting Population = .05k	20	109.01	19.13	89.40	0.16	0.32	3.19	1,315.36

Note: The values in the table represent the median value from 10,000 simulations.
* In millions of 2007 dollars; ‡ in millions of market-sized oysters.

ern Chesapeake Bay is rapidly improving, it remains an inexact science. Therefore, we estimate the model with starting populations of 5% of carrying capacity (starting stock of 254 million oysters) and 25% of carrying capacity (1.3 billion oysters). Reducing the stock to 5% of carrying capacity has the effect of dramatically increasing the length of the moratorium, as it takes longer for the stock to achieve a reasonable size to begin harvesting. Increasing the stock to 25% of carrying capacity causes the optimal moratorium to be shorter, with a higher harvest rate causing the steady-state stock to slightly decrease.

While changing parameters results in different moratorium lengths, a positive moratorium length was optimal in all sensitivity tests. Examining the effects of changing the growth rate, discount rate, carrying capacity, and starting population, our model demonstrates that in this fishery, to maximize the value of the resource as a whole, it is always optimal to close the fishery for a number of years to allow the stock to recover.

Conclusion

In the past few decades, oysters in the Northern Chesapeake Bay have fallen to historically low abundances as a result of disease mortality, degraded water quality, and a reduction in suitable habitat. In addition to the increasing pressures from the environment, the open-access fishery has been taking 8.05% of the market-sized oysters annually, on average. We develop a novel bioeconomic model of the oyster fishery, which accounts for the value of ecological benefits that oysters provide *in situ* in the Northern Chesapeake Bay. Given the historically low population levels and high potential value as both a commercially harvested species and source of ecological benefits, we use the model to determine the optimal season to begin harvesting (optimal moratorium length) and a subsequent harvest rate that maximizes the net present value of the oyster resource.

The ecological services provided by oysters in the Chesapeake Bay are presumed to provide value by supplying habitat for other species, improving water clarity, and removing excess nutrients. The model suggests that the current open-access policy is not efficient, and a lower harvest rate will increase the net present value of the oyster resource, even under the lower bound estimate of ecological service values. While this will result in a loss in harvest profits to watermen, it will allow the stock to rebuild and provide substantial ecological benefits to other users of the Northern Chesapeake Bay. Thus, the model expresses the tradeoff faced by managers of increasing watermen's profits from harvest (resulting in a degraded oyster stock and lower ecological services values), versus reducing allowable harvests (resulting in increased oyster stocks and greater ecological service values accruing to all users of the Bay). Notably, reducing harvests results in a higher overall net present value of the oyster resource.

Given the historically low abundance, the model also suggests that shutting down the fishery for a number of years to allow the stock to recover can increase the value of the resource for any harvest rate. As optimal harvest rates are often higher following the end of a moratorium, profits accruing to watermen are generally similar with and without the moratorium. Depending on the values assigned to the ecological services provided by oysters, the optimal moratorium is between 4 and 13 years.

While oysters are unique in their ability to provide ecological services on the scale that they once did in the Chesapeake Bay, these findings are not only applicable to this fishery. Many species provide ecological services which are valuable to humans through ecosystem interactions. An example is the stock of capelin, which serves as an essential food source for rebounding cod stocks in the North Atlantic (Rose and O'Driscoll 2002). Generalizing from the current study, the value of the capelin resource is not only their harvest value, but also includes the value associated with the ecological service they provide, which has value to cod fishermen. The FAO estimated that in 2005 25% of the world's fisheries were over exploited (FAO 2007); thus moratoriums are one tool that may potentially increase managers' ability to create sustainable fish populations and increase the value of marine resources.

References

Armstrong, C.W. 2007. A Note on the Ecological-economic Modeling of Marine Reserves in Fisheries. *Ecological Economics* 62:242–50.

Barker, L., K.N. Greenhawk, and T. O'Connell. 2008. Oyster Population Estimates for the Maryland Portion of Chesapeake Bay. Population Size, Biomass, and Exploitation Rate Time Series Estimates for 1994–2007. Annapolis, MD: Maryland Department of Natural Resources.

Bowes, M.D., and J.V. Krutilla. 1989. *Multiple-Use Management: The Economics of Public Forestlands*. Washington, DC: Resources for the Future.

Breitburg, D.L., T. Loher, C.A. Pacey, and A. Gerstein. 1997. Varying Effects of Low Dissolved Oxygen on Tropic Interactions in an Estuarine Food Web. *Ecological Monographs* 67(4):489–507.

Brown, G., and J. Roughgarden. 1997. A Metapopulation Model with Private Property and a Common Pool. *Ecological Economics* 22:65–71.

Caddy, J.F. 1993. Toward a Comparative Evaluation of Human Impacts on Fishery Ecosystems of Enclosed and Semi-Enclosed Seas. *Reviews in Fisheries Science* 1(1):57–95.

Calish, S., R.D. Fight, and D.E. Teeguarden. 1978. How Do Nontimber Values Affect Douglas-fir Rotations? *Journal of Forestry* 76:217–21.

Cerco, C.F., and M. Noel. 2007. Can Oyster Restoration Reverse Cultural Eutrophication in Chesapeake Bay? *Estuaries and Coasts* 30(2):331–43.

Chesapeake Bay Commission. 2003. The Cost of a Clean Bay: Assessing Funding Needs Throughout the Watershed. Annapolis, MD: Chesapeake Bay Commission. Available online at: http://www.chesbay.state.va.us/Publications/C2Kfunding.pdf

Clark, C.W. 1990. *Mathematical Bioeconomics. The Optimal Management of Renewable Resources*, 2nd ed. New York, NY: Wiley-Interscience.

Clark, C.W., F.H. Clarke, and G.R. Munro. 1979. The Optimal Exploitation of Renewable Resource Stocks: Problems of Irreversible Investment. *Econometrica* 47(1):25–47.

Clark, C.W., and G.R. Munro. 1975. The Economics of Fishing and Modern Capital Theory: A Simplified Approach. *Journal of Environmental Economics and Management* 2:92–106.

Coen, L.D., R.D. Brumbaugh, D. Bushek, R. Grizzle, M.W. Luckenbach, M.H. Posey, S.P. Powers, and S.G. Tolley. 2007. Ecosystem Services Related to Oyster Restoration. *Marine Ecology Progress Series* 341:303–7.

Fahrenthold, D.A. 2009. Bay Advocates Sue EPA. Coalition Demands Action, but Suit Lacks Specifics. *The Washington Post*, 6 Jan., B01.

Food and Agricultural Organization (FAO). 2007. *The State of World Fisheries and Aquaculture 2006*. Rome: FAO Fisheries and Aquaculture Department.

Freeman, A.M. 1995. The Benefits of Water Quality Improvements for Marine Recreation: A Review of the Empirical Evidence. *Marine Resource Economics* 10:385–406.

Fulford, R.S., D.L. Breitburg, R.I.E. Newell, W.M. Kemp, and M. Luckenbach. 2007. Effects of Oyster Population Restoration Strategies on Phytoplankton Biomass in Chesapeake Bay: A Flexible Modeling Approach. *Marine Ecology Progress Series* 336:43–61.

Grabowski, J.H., and C.H. Peterson. 2007. Restoring Oyster Reefs to Recover Ecosystem Services. *Ecosystem Engineers 4*. K. Cuddington, J. Byers, W. Wilson, and A. Hastings, eds., pp. 281–98. Amsterdam: Elsevier.

Grafton, R.Q., L.K. Sandal, and S.I. Steinshamn. 2000. How to Improve the Management of Renewable Resources: The Case of Canada's Northern Cod Fishery. *American Journal of Agricultural Economics* 82:570–80.

Hannesson, R. 1998. Marine Reserves: What Would They Accomplish? *Marine Resource Economics* 13:159–70.

Hartman, R. 1976. The Harvesting Decision when a Standing Forest has Value. *Economic Inquiry* XIV:52–8.

Herrera, G.E. 2007. Dynamic Use of Closures and Imperfectly Enforced Quotas in a Metapopulation. *American Journal of Agricultural Economics* 89(1):176–89.

Hicks, R.L., T.C. Haab, and D. Lipton. 2004. *The Economic Benefits of Oyster Restoration in the Chesapeake Bay*. Annapolis, MD: Chesapeake Bay Foundation.

Hilborn, R., and C.J. Walters. 1992. *Quantitative Fisheries Stock Assessment: Choice, Dynamics, and Uncertainty*. London: Chapman & Hall.

Holland, D.S., and R.J. Brazee. 1996. Marine Reserves for Fisheries Management. *Marine Resource Economics* 11:157–71.

Houde, E.D., and E.S. Rutherford. 1993. Recent Trends in Estuarine Fisheries: Predictions of Fish Production and Yield. *Estuaries* 16:161–76.

Jordan, S.J., and J. Coakley. 2004. Long-Term Projections of Eastern Oyster Populations under Various Management Scenarios. *Journal of Shellfish Research* 23:63–72.

Kahn, J.R., and W.M. Kemp. 1985. Economic Losses Associated with the Degradation of an Ecosystem: The Case of Submerged Aquatic Vegetation in Chesapeake Bay. *Journal of Environmental Economics and Management* 12(3):246–63.

Kennedy, V.S., and L.L. Breisch. 1983. Sixteen Decades of Political Management of the Oyster Fishery in Maryland's Chesapeake Bay. *Journal of Environmental Management* 164:153–77.

Knowler, D. 2002. A Review of Selected Bioeconomic Models with Environmental Influences in Fisheries. *Journal of Bioeconomics* 4:163–81.

Lee, G.F., and R.A. Jones. 1991. Effects of Eutrophication on Fisheries. *Reviews in Aquatic Sciences* 5(3-4):287–305.

Liddel, M.K. 2007. *A von Bertalanffy based Model for the Estimation of Oyster (*Crassostrea virginica*) Growth on Restored Oyster Reefs in Chesapeake Bay*. Doctoral dissertation, University of Maryland.

Lipton, D. 2008. Final Draft Economic Analysis for Oyster Restoration Alternatives. *Final Programmatic Environmental Impact Statement for Oyster Restoration in Chesapeake Bay Including the Use of a Native and/or Nonnative Oyster*. Norfolk, VA: U.S. Army Corps of Engineers.

Lipton, D., and S. Kasperski. 2008. Estuarine Restoration and Commercial Fisheries. *The Economic and Market Value of Coasts and Estuaries: What's at Stake?* L. Pendleton, ed., pp. 65–81. Washington, DC: Restore America's Estuaries.

Lipton, D., J. Kirkley, and T. Murray. 2006. A Background Economic Analysis for the Programmatic Environmental Impact Statement Regarding the Restoration of the Chesapeake Bay Oyster Fishery Using the Non-Native Oyster, *Crassostrea ariakensis*. *Final Programmatic Environmental Impact Statement for Oyster Restoration in Chesapeake Bay Including the Use of a Native and/or Nonnative Oyster*. Norfolk, VA: U.S. Army Corps of Engineers.

Mistiaen, J.A., I.E. Strand, and D. Lipton. 2003. Effect of Environmental Stress on Blue Crab (*Callinectes sapidus*) Harvests in Chesapeake Bay Tributaries. *Estuaries* 26(2a):316–23.

Murray, T.J. 2002. Making a Case for the Oysters: Putting Ecological Benefits into Economic Terms. *Virginia Marine Resource Bulletin* 34(2):7–10.

Newell, R.I.E. 1988. Ecological Changes in Chesapeake Bay, Are they the Result of Overharvesting the Eastern Oyster (*Crassostrea virginica*)? *Understanding the Estuary: Advances in Chesapeake Bay Research*, M.P. Lynch and E.C. Krome, eds., pp. 536–46. Gloucester Point, VA: Chesapeake Research Consortium Publication 129 (CBP/TRS 24/88). Available online at: www.vims.edu/GreyLit/crc129.pdf

Newell, R.I.E., T.R. Fisher, R.R. Holyoke and J.C. Cornwell, 2005. Influence of Eastern Oysters on Nitrogen and Phosphorus Regeneration in Chesapeake Bay, USA. *The Comparative Roles of Suspension Feeders in Ecosystems*. R. Dame and S. Olenin, eds., pp. 93–120. The Netherlands: Springer.

North, E.W., Z. Schlag, R.R. Hood, L. Zhong, M. Li, and T. Gross. 2006. Modeling Dispersal of *Crassostrea ariakensis* Oyster Larvae in Chesapeake Bay. Annapolis, MD: Maryland Department of Natural Resources.

Nøstbakken, L. 2008. Stochastic Modelling of the North Sea Herring Fishery under Alternative Management Regimes. *Marine Resource Economics* 23:65–86.

Rose, G.A., and R.L. O'Driscoll. 2002. Capelin are Good for Cod: Can the Northern Stock Rebuild without Them? *ICES Journal of Marine Science* 59:1018–26.

Rothschild, B.J., J.S. Ault, P. Goulletquer, and M. Heral. 1994. Decline of the Chesapeake Bay Oyster Population: A Century of Habitat Destruction and Overfishing. *Marine Ecological Progress Series* 111(1–2):29–39.

Smith, G.F., D.G. Bruce, E.B. Roach, A. Hansen, R.I.E. Newell, and A.M. McManus. 2005. Assessment of Recent Habitat Conditions of Eastern Oyster *Crassostrea virginica* Bars in Mesohaline Chesapeake Bay. *North American Journal of Fisheries Management* 25:1569–90.

Smith, G.F., K.N. Greenhawk, D.G. Bruce, E.B. Roach, and S.J. Jordan. 2001. A Digital Representation of the Maryland Oyster Habitat and Associated Bottom Types in the Chesapeake Bay. *Journal of Shellfish Research* 20(1):197–206.

Smith, M.D., and J.E. Wilen. 2003. Economic Impacts of Marine Reserves: The Importance of Spatial Behavior. *Journal of Environmental Economics and Management* 46:183–206.

Smith, M.D., J. Zhang, and F.C. Coleman. 2008. Econometric Modeling of Fisheries with Complex Life Histories: Avoiding Biological Management Failures. *Journal of Environmental Economics and Management* 55:265–80.

Sohngen, B., and R. Mendelsohn. 2003. An Optimal Control Model of Forest Carbon Sequestration. *American Journal of Agricultural Economics* 85(2):448–57.

Spence, M., and D. Starrett. 1975. Most Rapid Approach Paths in Accumulation Problems. *International Economic Review* 16(2):388–403.

Tahvonen, O., and S. Salo. 1999. Optimal Forest Rotation with *in situ* Preferences. *Journal of Environmental Economics and Management* 37:106–28.

Valderrama, D., and J.L. Anderson. 2007. Improving Utilization of the Atlantic Sea Scallop Resource: An Analysis of Rotational Management of Fishing Grounds. *Land Economics* 83(1):86–103.

van Kooten, G.C., C.S. Binkley, and G. Delcourt. 1995. Effect of Carbon Taxes and Subsidies on Optimal Forest Rotation Age and Supply of Carbon Services. *American Journal of Agricultural Economics* 77:365–74.

Vølstad, J.H., M.C. Christman, D. Lewis, E.D. Weber, and J. Dew. 2007. A Demographic Model of Oyster Populations in the Chesapeake Bay to Evaluate Proposed Oyster-Restoration Alternatives. *Final Programmatic Environmental Impact Statement for Oyster Restoration in Chesapeake Bay Including the Use of a Native and/or Nonnative Oyster*. Norfolk, VA: U.S. Army Corps of Engineers.

Wieland, R.C. 2006. Operating Costs in the Chesapeake Bay Oyster Fishery. *Draft Programmatic Environmental Impact Statement for Oyster Restoration in Chesapeake Bay Including the Use of a Native and/or Nonnative Oyster*. Norfolk, VA: U.S. Army Corps of Engineers.

_____. 2007. Managing Oyster Harvests in Maryland's Chesapeake Bay. *Draft Programmatic Environmental Impact Statement for Oyster Restoration in Chesapeake Bay Including the Use of a Native and/or Nonnative Oyster*. Norfolk, VA: U.S. Army Corps of Engineers.

Wieland, R.C., and S. Kasperski. 2008. Estimating Net Present Value in the Northern Chesapeake Bay Oyster Fishery. *Draft Programmatic Environmental Impact Statement for Oyster Restoration in Chesapeake Bay Including the Use of a Native and/or Nonnative Oyster*. Norfolk, VA: U.S. Army Corps of Engineers.

Wilson, S.T. 2009. What Maryland Can Do to Help Clean Up the Bay. *The Washington Post*, 25 Jan., B08.