GENERATING VALUE IN HABITAT-DEPENDENT FISHERIES:

THE IMPORTANCE OF FISHERY MANAGEMENT INSTITUTIONS

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ABSTRACT

This paper considers the dynamic producer and consumer benefits from improving habitat that supports a commercial fishery under two different fishery management institutions. By coupling state equations that represent the effects of estuarine eutrophication on fish populations with a multispecies, two-patch spatial bioeconomic model that endogenizes output price through residual demand, the analysis computes welfare changes from a major reduction in nutrient pollution. This, in turn, reduces the incidence of hypoxia (low dissolved oxygen) and enhances prey availability. The North Carolina blue crab fishery serves as the empirical application, and water quality improvements pertain to the Neuse River Estuary and the contiguous Pamlico Sound. The analysis simulates dynamic rent and consumer surplus changes from a 30% decrease in nitrogen loading under both open access (the status quo) and a partially rationalized fishery (constant total effort). Producer benefits from the environmental quality change are higher for the rationalized fishery than for open access but are of the same order of magnitude for some parameter values. Consumer benefits are larger than producer benefits and are comparable across institutions. However, the total benefits from improving environmental quality are small relative to the benefits from rationalizing the fishery and leaving environmental quality the same.

INTRODUCTION

The Dead Zone in Gulf of Mexico and growing awareness of other eutrophic coastal waters have focused policy attention on the fishery impacts of nutrient pollution. Anthropogenic nitrogen loading in estuaries—largely attributable to agricultural runoff and atmospheric

deposition from fossil fuel combustion—stimulates primary productivity and increases oxygen demand. Combined with hydrodynamic conditions, increased oxygen demand can lead to areas of low oxygen (hypoxia) or even no oxygen (anoxia), resulting in fish kills, toxic algal blooms, and sub-lethal effects such as fish migrations (NRC 2000). Given the substantial costs of reducing anthropogenic nitrogen and the policy challenges of dealing with multiple industries over large-scale watersheds, it is natural to ask how much would commercial fisheries benefit from reductions in nutrient pollution and the corresponding improvements in estuarine habitat? I explore this question with a bioeconomic model of the blue crab fishery in North Carolina's Neuse River Watershed.

To account for several unique features of blue crab ecology, Smith and Crowder (2005) develop a two-patch predator-prey model that incorporates both direct and indirect effects of hypoxia. The direct effect is the tendency for blue crabs to avoid low oxygen levels and crowd into oxygenated areas, while the indirect effect is the tendency for hypoxia to kill the preferred prey of the crabs. I extend this model to account for the regional market for blue crabs and use it to compute both transition rents and dynamic consumer surplus (CS) changes for a 30% reduction in nitrogen loadings. This nitrogen reduction is the specific policy change recommended by a recent Total Maximum Daily Load Plan for the Neuse River watershed (NC DENR DWQ 2001), while blue crab is the largest commercial fishery in North Carolina (NC). For computing the value of improved habitat for NC blue crabs, open access is a plausible baseline institution. After some experimentation with limited entry in the 1990s, in September of 1999 the NC Marine Fisheries Commission decided that it would not continue to pursue limited entry and would focus exclusively on open access options.

I compare surplus changes under the existing open access institutional structure of the fishery with the same values under a simple fishery rationalization scheme. In all scenarios, the increase in fishery rents from reducing nutrient pollution is higher in the rationalized fishery than in the open access fishery. CS changes are similar across institutional regimes. However, I find that the producer gains from rationalizing the blue crab fishery without an environmental change are more than an order of magnitude greater than the corresponding gains from reducing nutrient pollution by 30% and leaving the fishery as open access. This suggests that cleaning up nutrient pollution—though it would benefit producers some— is really not the problem for the NC blue crab fishery, and the real problem lies in a lack of institutions to rectify open access to the crab resource.

Literature review

The approach in this paper draws on four strands of the bioeconomic literature on which the model draws: multispecies models, models of renewable resource habitat dependence, spatial models, and empirical bioeconomics of an open access fishery. Quirk and Smith (1970) first suggest a bioeconomic model for multispecies fisheries. My approach builds on theoretical multispecies bioeconomic models that include two interacting species with explicit functional forms in which each species has both logistic growth terms and Lotka-Volterra predator-prey interaction terms (Hannesson, 1983; Ragozin and Brown, 1985; Wilen and Brown, 1986; Flaaten, 1991; Tu and Wilman, 1992; and Wacker,1999).[1] This structure can represent one species preying upon the other or two species competing for a common resource depending on the signs of the interaction terms. More recently, Brock and Xepapadeas (2004) theoretically analyze an open access generalized predator-prey fishery model and highlight the importance of

[1] This model structure originates in the biology literature (Gause, 1934).

economic speed of adjustment relative to the speed of biological dynamics. Economic speed of adjustment plays a key role in my analysis as well, though the main qualitative result—that benefits from fishery rationalization far outweigh benefits from reduced pollution—holds even with slow adjustment. Ragozin and Brown (1985) observe that the commercial value of a non-harvested prey species can be inferred from the fishery rents of the harvested predator if the predator-prey dynamics are modeled. This argument is in the spirit of revealed preference valuation methods that rely on complementarity, but it applies to welfare of producers rather than consumers. The current paper models producers and consumers. The complementarity passes through multiple ecological channels—nutrients to algae, algae to hypoxia, and hypoxia to impacts on predators and prey—and it is multi-faceted—hypoxia-induced prey mortality and hypoxia-induced predator migration.

Empirical multispecies bioeconomic modeling has been limited. Conrad and Adu-Asamoah (1986) model three tuna species independently and compare empirical results with a model in which two of the species compete. They find optimal effort levels when the species compete at less than one half of optimal effort levels when the species are considered independent. The data and empirical technique, however, limits their analysis only to steady-state results. Flaaten and Stollery (1996) compute the cost of Minke whale predation on commercially valuable prey species. Their model analyzes only the prey steady state and thus simplifies the predator-prey dynamics. They use bioenergetic intake of the predator to parameterize the model. Using a similar bioenergetic approach, Kaplan and Smith (2001) parameterize a predator-prey model to compute the sea urchin fishery cost of expanding the

California sea otter protection area.[2] These authors take the analysis a step further by computing the fishery losses as the dynamic shadow value of the predator.

There are two contrasting approaches to trophic interactions in multispecies bioeconomic models; one involves structural bioenergetic modeling of species competition and the other involves hyper-reduced-form modeling. Brock and Xepapadeas (2002) analyze a model in which species compete for a resource represented by a separate state variable, and these authors derive results for equilibrium species specialization. Also focusing on bio-energetic competition is the general equilibrium approach for a multispecies system in which individual species optimize over energy intake (Finnoff and Tschirhart 2003). At the opposite end of the spectrum is the trophic portfolios approach (Sanchirico and Smith, 2003; Sanchirico, Smith, and Lipton, 2005) in which structure is abandoned altogether—hence the characterization of hyper-reduced-form—and multispecies management is cast in terms of finding empirical regularities in species harvests as related stochastic processes.

As demand increases for economists to value ecosystem services, it is important to review the conceptual literature on environmental dependence of a renewable resource. McConnell and Strand (1989) show that rationalizing a fishery is necessary to capture the greatest gains from a water quality improvement. They also show that under some circumstances, when there is open access to the fish stock a water quality improvement can have perverse welfare consequences. Swallow (1990) theoretically models the dependence of a renewable resource on a non-renewable resource. He motivates the model with estuarine-dependence of fisheries and views agricultural and urban development as an irreversible impact on the environmental basis of the renewable resource. In order for the resource industry to

[2] This bioenergetic basis for parameterizing a predator-prey model works in the urchin-otter system because it appears that Southern sea otters have lexicographic preferences for urchins.

survive in his model, development stops when there are still profitable development opportunities. Though Swallow specifically discusses the Pamlico Sound as an example, the model in the current paper deals with different issues. It departs from the Swallow approach by treating all aspects of the system as renewables; the only irreversible effects would be total collapses of the resource.

Empirical welfare models of environmental dependence of a fishery focus entirely on equilibrium-based estimates but lay the groundwork for thinking about an ecosystem service as an input for the production of fish. Kahn and Kemp (1985) use an equilibrium-based catch function to estimate indirect effects of water pollution on Chesapeake Bay fisheries. Ellis and Fisher (1987) and Freeman (1991) calculates equilibrium changes in consumer and producer surplus (for Gulf Coast blue crabs) from increases in wetlands. Barbier and Strand (1998) model habitat dependence of a fishery by empirically examining the value of mangroves as nursery grounds for the shrimp fishery in Campeche, Mexico. Like much of the empirical bioeconomic literature, these authors focus on open access. However, they impose open access in each period by assuming instantaneous adjustment in the fishing sector. This assumption means that rents are always zero. Barbier, Strand, and Sathirathai (2002) and Barbier (2003) find CS changes in open access equilibrium for mangrove-fishery linkages in Thailand. Knowler, Barbier and Strand (2001) follow a similar approach for estimating the effects of nutrient enrichment on black sea anchovy but also acknowledge non-monotonicity in nutrient effects on productivity. All of these studies can recover changes in harvest and revenues that are attributable to habitat loss but are unable to measure economic welfare losses in the harvest sector. I depart significantly from this approach and study transition rents as well as dynamic CS changes.

Sanchirico and Wilen (1999) develop a lumped-parameter theoretical bioeconomic model to study spatial issues in fisheries management. Their model has logistic growth of a single species in each patch, biological patch linkages based on density dependence, total fishing effort adjustment based on the V. Smith (1969) model, and spatial effort distribution based on relative profitability. The two-patch model in the current paper follows Sanchirico and Wilen but ties biological dispersal to relative prey availability rather than population density. The spatial distribution of fishing effort, in contrast, follows the discrete choice approach of Smith and Wilen (2003).

There is a small amount of empirical bioeconomic literature on entry and exit in commercial fisheries under open access.[3] In the first empirical work, Wilen (1976) focuses on the aggregate fleet size, assumes homogeneous fishing vessels, and models the economic dynamics with a single speed-of-adjustment parameter. He empirically estimates a V. Smith (1969) open access model with historical data on the north Pacific fur seal fishery. Bjorndal and Conrad (1987) follow a similar approach for the North Sea herring fishery. Both papers find empirical support that the fishing fleet overshoots the open access bioeconomic equilibrium such that effort and biomass cycle towards the equilibrium. Though high quality effort data are lacking, my analysis follows Wilen's model to simulate dynamic bioeconomic outcomes.

The following section formally develops the model and briefly discusses its parameterization. The third section presents policy simulations, and the final section discusses the implications for reducing nutrient pollution and fisheries management.

THE MODEL

[3] Indeed, most high-profile work in fisheries economics has been conceptual (Wilen, 2000).

The model is a system of ordinary differential equations developed in Smith and Crowder (2005) with the addition of a residual demand model to endogenize price. Each equation represents a relevant state of the system: the stock of nutrients, algae levels, spatially explicit populations of the blue crab and prey availability, total fishing effort, the spatial distribution of fishing effort, and the price level. Denoting *t* as time, the change in stock of nutrients (*N(t)*) in the estuary with respect to time is:

$$\dot{N}(t) = L(t) - \omega N(t), \qquad [1]$$

where *L(t)* denotes nutrient loadings and ω captures the natural rate of decay.[4] Assume that nutrient loadings are constant over time, $L(t) = \bar{L}$, and any policy change would modify this constant rate. Algae (*A(t)*) follows logistic growth with the intrinsic growth rate normalized to one and the carrying capacity as a scalar (μ) times the nutrient stock:

$$\dot{A}(t) = A(t)\left[1 - A(t)/\mu N(t)\right]. \qquad [2]$$

In this way, the nutrient stock limits the long-run stock of algae hence the severity of hypoxia.

Hypoxia is spatially delineated, and to characterize space parsimoniously, the model represents a two-patch bioeconomic system. Patch 1 is susceptible to hypoxia and modeled as such, whereas patch 2 is assumed to experience only background effects of low oxygen, and these background effects are folded into other lumped parameters. There are four state equations that describe the population dynamics: predators in each patch and prey in each patch. Blue crabs are the harvested predators and are subject environmental limits on their abundance that include prey abundance. Prey are unharvested and are a composite of clam species—the preferred prey of blue crabs (Taylor and Eggleston 2000)—that are unable to move across patches. Denote predators and prey in Patch *i* as $X_i(t)$ and $Y_i(t)$ respectively. There are five components of

[4] As a starting point, the model assumes no hysterisis.

predator population dynamics: logistic growth of each species, the Lotka-Volterra predator-prey interaction terms, harvest, migration due to hypoxia[5], and migration unrelated to hypoxia:

$$\dot{X}_1(t) = r_x X_1(t)\left[1 - X_1(t)/k_x\right] + \alpha Y_1(t) X_1(t) - h_1(t) - \xi A(t) X_1(t) + \phi\left[Y_1(t)/X_1(t) - Y_2(t)/X_2(t)\right], \quad [3]$$

$$\dot{X}_2(t) = r_x X_2(t)\left[1 - X_2(t)/k_x\right] + \alpha Y_2(t) X_2(t) - h_2(t) + \xi A(t) X_1(t) - \phi\left[Y_1(t)/X_1(t) - Y_2(t)/X_2(t)\right], \quad [4]$$

where r_x and k_x are crab intrinsic growth and carrying capacity terms, α captures increased growth due to prey availability, $h_i(t)$ is patch-specific harvest, ξ is the hypoxia migration parameter where by assumption algae translates directly into lower dissolved oxygen, and ϕ denotes responsiveness to relative prey density and thus controls spatial connectivity of the patches.

The population dynamics of the preferred prey species are similar, but the effects of hypoxia are quite different. As in equations (3) and (4), prey in each patch has a logistic growth component and a predator-prey interaction component. Prey carrying capacity is normalized to one, there is no harvest, and the prey are stationary. Immobility means that the hypoxia is a source of prey mortality in Patch 1. This appears the same as the prey having two predators, one of which is algae, i.e. prey are "eaten" by poor habitat. Patch 1 prey dynamics have all three components, while Patch 2 prey state dynamics have just two:

$$\dot{Y}_1(t) = r_y Y_1(t)\left[1 - Y_1(t)\right] - \beta X_1(t) Y_1(t) - \rho A(t) Y_1(t), \text{ and} \quad [5]$$

$$\dot{Y}_2(t) = r_y Y_2(t)\left[1 - Y_2(t)\right] - \beta X_2(t) Y_2(t). \quad [6]$$

The prey state equations thus require just three parameters: intrinsic growth of prey (r_y), predator-prey interaction coefficient (β), and hypoxia-related prey deaths (ρ). The parameter ρ is

[5] Following the empirical findings of Selberg, Eby, and Crowder (2001) and Eby and Crowder (2002), the direct effects of hypoxia on blue crabs is migration from deoxygenated areas.

lumped in that it captures both the effect of primary productivity on sediment oxygen demand and the effect of sediment oxygen demand on prey mortality.

There are three states that characterize the economic system: effort in patch 1 ($E_1(t)$), effort in patch 2 ($E_2(t)$), and output price ($p(t)$). Following Smith (2002) and Smith and Wilen (2003), it is useful to reformulate spatially explicit effort as total fishing effort $E(t)$ and the share in each patch (π_i). Formally,

$$E(t) = E_1(t) + E_2(t),\ E_i(t) = \pi_i(t)E(t), \text{ and } \pi_1(t) + \pi_2(t) = 1, \forall t. \quad [7]$$

The blue crab fishery in North Carolina is has been essentially open access for most of its history. There are some limits on gear technology but no formal limits on entry, and though there was a temporary moratorium on new permits, arguably it was not in effect long enough to fundamentally alter the open access dynamics of the fishery. Thus, total effort in the model follows a dynamic open access rent dissipation model originally formulated by V. Smith (1969) Total fishery rents ($\Pi_{total}(t)$) are simply the difference between fishing revenues and fishing costs where c represents a combined constant marginal cost of fishing equipment, opportunity cost of time, and the opportunity cost of investing in the equipment (cost of capital and depreciation):

$$\Pi_{Total}(t) = p(t)\left[h_1(t) + h_2(t)\right] - cE(t). \quad [8]$$

Crab fishermen are price-takers, but aggregate supply from the Neuse can affect regional prices. Following open access theory, total effort evolves according to average rent per unit of effort rather than marginal rent:

$$\dot{E}(t) = \gamma\left\{p(t)\frac{\left[h_1(t)+h_2(t)\right]}{E(t)} - c\right\}. \quad [9]$$

The share of effort in each patch responds to relative density of blue crabs according to a logistic probability density function. For Patch 1, effort share is:

$$\pi_1(t) = \frac{\exp(\theta X_1(t))}{\exp(\theta X_1(t)) + \exp(\theta X_2(t))}. \quad [10]$$

This approach is consistent with the empirical literature on broad fishery choice (Bockstael and Opaluch, 1983) as well as specific fishing location choice (Smith, 2002). A single parameter θ captures the responsiveness (over space) to different levels of abundance. To stack the system of differential equations, take the time derivative of (10) and simplify to obtain:

$$\dot{\pi}_1(t) = \frac{\theta\left[\dot{X}_1(t) - \dot{X}_2(t)\right]}{\exp(\theta\left[X_1(t) - X_2(t)\right]) + 2 + \exp(\theta\left[X_2(t) - X_1(t)\right])}. \quad [11]$$

A Schaefer (1957) production function relates effort and stocks to harvest:

$$h_i(t) = qE_i(t)X_i(t), \quad [12]$$

where q is a catchability coefficient assumed constant across the patches. Substituting (12) and (7) into (9):

$$\dot{E}(t) = \gamma\left\{p(t)q\left[\pi_1(t)X_1(t) + (1-\pi_1(t))X_2(t)\right] - c\right\}. \quad [13]$$

To model price, suppose that there is a constant-elasticity regional demand for blue crab ($Q_d(t)$) with parameters a and b:

$$Q_d(t) = ap(t)^b. \quad [14]$$

One can decompose regional quantity supplied ($Q_s(t)$) into patch-specific quantities harvested and quantity supplied outside the Neuse-Pamlico area ($\tilde{Q}$), which is assumed to be constant:

$$Q_s(t) = \tilde{Q} + h_1(t) + h_2(t). \quad [15]$$

By imposing supply-demand equilibrium in each period and inverting the demand curve,

$$p(t) = \left[\frac{\tilde{Q} + h_1(t) + h_2(t)}{a}\right]^{(1/b)}. \quad [16]$$

Finally, take a time derivative to find the price path and stack with the rest of the ODE's in the model:

$$\dot{p}(t) = {}^{(1/b)}\left[\frac{\tilde{Q} + q\pi_1(t)E(t)X_1(t) + q\left(1-\pi_1(t)\right)(t)E(t)X_2(t)}{a}\right]^{(1/b)-1}$$
$$\times\frac{q}{a}\left\{E(t)\left[\dot{\pi}_1(t)X_1(t) + \pi_1(t)\dot{X}_1(t) - \dot{\pi}_1(t)X_2(t) + \left(1-\pi_1(t)\right)\dot{X}_2(t)\right]\right. \qquad [17]$$
$$\left. + \dot{E}(t)\left[\pi_1(t)X_1(t) + \left(1-\pi_1(t)\right)X_2(t)\right]\right\}.$$

To summarize, the system can be described by nine state variables [$N(t)$, $A(t)$, $X_1(t)$, $X_2(t)$, $Y_1(t)$, $Y_2(t)$, $E(t)$, $\pi_1(t)$, and $p(t)$], parameters, and initial conditions.

A full description of the parameters used for the first eight state equations is in Smith and Crowder (2005). Table 1 provides a brief summary. For the parameters of the residual demand model, simulations are run with three different price elasticities to allow for inelastic, unit-elastic and elastic demands. I compute $\tilde{Q}$ based on the observations that in recent years North Carolina catches constitute roughly 25% of the U.S. East Coast blue crab market (NCDENR –DMF FMP 2004) and catches in the Neuse-Pamlico area are about 35% of the NC fishery (Eggleston et al. 2004). Then I back out the corresponding parameter a for each level of b to calibrate the price to the average NC blue crab price for 2000-2002.

RESULTS

Two sets of 27 simulations are run over a time horizon of 100 years. The number of simulations captures all permutations of three parameters that are allowed to take on three different values: γ (open access economic speed of adjustment), ϕ (biological spatial connectivity), and b (price elasticity of demand). The simulation results allow variation in these three parameters for the following reasons. Although rent is dissipated in the steady state, the

extent to which the system generates transition rents depends on the particular values of all parameters, and Smith and Crowder (2005) find that results are particularly sensitive to the speed of adjustment γ. In contrast, Smith and Crowder (2005) do not explore the sensitivity of model results to ϕ. Because there is no empirical basis for the level of ϕ—other than choosing values that are internally consistent with the remainder of the model—it is worth considering several values. Finally, considering both elastic and inelastic regional demands will provide insight on how large or small surplus changes could be in response to reduced nutrient pollution in just one portion of the regional supply. This, in turn, will suggest how important it is to do further study of regional blue crab market, including scope, scale, and the degree of market integration.

To understand the role of fishery management institutions in generating value from an environmental quality change, two sets of simulations are run. The first set corresponds to the open access model described above. Open access is essentially the status quo for NC blue crabs. The second set corresponds to a quasi-optimized model in which a regulator can fix the total effort in the system once and for all such that no bioeconomic overshooting occurs and rents are not fully dissipated. The spatial distribution of effort still evolves according to (11). To implement the quasi-optimized model, one need only replace the right-hand side of the state equation for total effort (13) with zero, and then do a grid search over initial conditions of E for effort that maximize total present value rents. These rents and the corresponding levels of state variables are feasible solutions that provide a lower bound on what the fishery would be worth under rationalization. This is the sense in which the model is quasi-optimized.

The quasi-optimized model is appealing for several reasons. First, it is numerically tractable. In contrast, a fully optimized model using optimal control involves solving a system of

highly nonlinear differential equations.[6] Second, given the resistance to fishery rationalization in NC, it is unlikely that anything close to a fully optimized fishery could be implemented. A blunt restriction on total effort—perhaps tied to a restriction on fishing pots—is arguably a more politically feasible solution in the near term. Thus, the quasi-optimized model is a more relevant rationalization baseline for comparing to open access than a fully optimized system. Finally, even if the NC regulator could implement a full rationalization through effort quotas or optimal effort taxes, it is unlikely that the regulator would have the resources to enforce spatially explicit effort controls.

The simulations are run in continuous time. To compute rent and CS changes (for comparing the policy change to no policy change), the time step is 0.1 years such that changes are added up (after present valuing) at each interval and then the total is divided by ten. Total instantaneous rents are from equation (8). Note that this rent equation is the same for both the open access and the quasi-optimized models. Instantaneous CS change at time τ is just the integral of (14) over the price change associated with the policy change (from $p_{no}(\tau)$ to $p_{policy}(\tau)$):

$$CS(\tau) = \int_{p_{policy}(\tau)}^{p_{no}(\tau)} as(\tau)^b \, ds = \frac{a}{b+1}\left[p_{no}(\tau)^{b+1} - p_{policy}(\tau)^{b+1} \right]. \qquad [18]$$

A discount rate of 4.5% is used to bring all values to present value (PV).

Table 2 contains the results and suggests a number of lessons. First and foremost, the gains from fishery rationalization (even this simple constant effort solution) far outweigh the gains from reduced nutrient pollution. Net rent increases from rationalization range from $199 million to over $270 million (compare baseline open access rents to baseline quasi-optimized

[6] Details are available from the author.

rents).[7] In contrast, the net rent increases from reduced nutrient pollution under open access range from less than $1 million to $6 million, and the quasi-optimized rent changes range from $6 to $7 million. The CS increases from the nutrient reduction range from $3 million to $25 million in the open access model and from $5 million to $22 million in the quasi-optimized model. Thus, the total welfare increase from the environmental quality improvement is in the neighborhood of 15% or less of the total welfare generated simply by rationalizing the fishery.

A second result from Table 2 is that the net rent increases from the environmental change are larger for the rationalized fishery than for the open access fishery. This result is qualitatively consistent with the theoretical results of McConnell and Strand (1989). Quantitatively, the difference across models varies an order of magnitude. When γ is large, the rent increase from the environmental quality change is small, but when γ is small, the rent increase is close to that from the rent increase in the rationalized fishery.

A third result in Table 2 is that on the consumer side, the CS increases from the environmental change are larger or smaller for the rationalized fishery than for the open access fishery, depending on the economic speed of adjustment. In fact, with slow economic adjustment and inelastic demand, the total surplus change from reduced nutrient pollution is larger under open access than under the rationalized fishery.[8] At first blush this result seems counterintuitive, but it comes about from the dynamics of the system. Under open access, the reduced nutrient pollution puts some short-run rents on the table and creates an incentive for entry. This entry increases short-run harvest relative to the quasi-optimized world and thus lowers price more. In

[7] I do not report changes in CS from rationalization here, but they are positive in all scenarios considered. The price path under rationalization differs from the one under open access, so there are some periods in which CS from rationalization is negative. In the long run, however, the rationalized fishery sustains a higher level of harvest and thus a lower price.

[8] This result may or may not hold if the rationalization followed a spatially explicit optimal effort rule. Obviously, it would not hold if CS were entered into the social planner's problem and not just fishery rents.

essence, some of the bioeconomic overshooting in open access is transferred to consumers, albeit only temporarily. When adjustment is slow, the lower price early in the time horizon persists long enough to make up for the lower long-run harvest of the open access system. When adjustment is fast, the long-run lower harvests and higher prices overwhelm the short run. These results highlight not only the importance of management institutions but also the need to model environmental changes dynamically when they affect renewable resource industries; focusing exclusively on the steady state—as previous authors have done— can produce a qualitatively different policy prescription.

Finally, though there are exceptions, the general pattern in Table 2 shows that more spatial connectivity leads to more rent and CS increases from the environmental quality change under open access, but the opposite is true for the quasi-optimized case. In the latter case, the effort limit is a blunt policy instrument, and the spatial distribution of effort still responds sluggishly to spatially delineated abundance. When there is no spatial connectivity (besides the effects of hypoxia), there is less overall flux in spatial abundance. As a result, the fixed total effort with sluggish spatial adjustment is more often closer to the optimal spatial effort distribution than when there is more spatial connectivity of the biomass. For open access, more flux in spatial abundance means the instantaneous rents in the system are dampened. This, in turn, reduces the entry pressure and sustains the rent increase from the quality improvement for longer relative to the case of lower spatial connectivity. In essence, the spatial externality in effort distribution helps, though only slightly, to offset the open access externality.

DISCUSSION AND CONCLUDING REMARKS

One might reasonably ask why a major reduction in nutrient pollution for a highly eutrophic system only produces fishery rent increases for NC's most valuable fishery in the

range of $1 to $7 million. The main answer is that blue crab fishery rents are only a fraction of the benefits even when the fishery is rationalized. Some fishery benefits flow to other fisheries. Beyond that, only some portion of benefits can be capitalized into commercial fisheries at all; others will accrue to recreational fisheries and non-consumptive uses. Nevertheless, the shifting baselines phenomenon documented by Jackson et al. (2001) provide an alternative explanation that serves as an important caveat for thinking about how to generate value from habitat-dependent fisheries. Jackson et al. (2001) argue that some fisheries were so abundant prior to human exploitation that our data since the onset of modern fishery management does not adequately capture the unexploited potential of the system. As such, restored ecosystems could generate far more fishery value than we have experienced throughout the period of modern industrial fishing.

In contrast, on the consumer side CS changes from reduced nutrient pollution are sizable. CS increases reflect small price decreases over a large market. In a static supply-demand model, the lion's share of CS changes would reflect transfers from producers to consumers. In both the open access and the quasi-optimized models, the gains from reduced nutrient pollution are shared by producers and consumers. However, for the open access model, the large CS increases are not transfers from producers; they are new surplus that otherwise would have been dissipated away. For the quasi-optimized model, a portion of the CS increase can be thought of as a transfer from producers. Table 2 reports net rent increases, so essentially producers are able to gain enough from nutrient reductions to more than offset transfers to consumers.

Interpreting CS changes for benefit-cost analysis is not straightforward. The environmental changes that I consider exclusively affect the Neuse-Pamlico. The costs of these changes would be borne by the state of NC and by individual counties within the Neuse

watershed. Yet, the consumer benefits are spread over consumers in the regional market. This market spans the eastern U.S., and to the extent that some blue crabs are processed, connects to the world market. Thus, the costs of nutrient reduction and benefits to producers are concentrated, but the benefits to consumers are diffuse. Moreover, in a general equilibrium sense, the extent to which improved water quality in the Neuse reduces regional blue crab prices, producers in other parts of the region could be harmed. This conjecture is contingent on rationalization of other blue crab fisheries in the region such as the Chesapeake. Otherwise, price decreases simply mitigate entry or facilitate exit in a rent dissipating process. Overall, the ambiguity of the distributional aspects of CS changes as well as the sensitivity of them to the demand elasticity assumption supports the need for more regional demand analysis of fish products considered in bioeconomic models.

All fisheries require habitat to support them, but generating value from that habitat requires proper institutions to manage the harvested resource. Policy decisions about fishery management and about marine and estuarine pollution are often made by different agencies, though recent emphasis on ecosystem-based fishery management (Pew Commission 2003; US Oceans Commission 2004) has forced policy-makers to think about them jointly. Increasing awareness about the environmental basis for fisheries is a welcome development. Nevertheless, if it comes at the expense of deflecting attention away from the first-order drivers of fisheries problems, then it may be counterproductive. In spite of the large and geographically diffuse consumer benefits and rent increases for blue crab fishermen from reduced nutrient pollution, total benefits are still small compared to the ability to generate value by rationalizing the NC blue crab fishery. From one case study, one would not want to draw too many inferences about the state of marine and estuarine pollution. However, this case does illustrate that all of society's

efforts to return the environmental basis of fisheries to a pristine state does little to fix the fishery problem if access to the fish resources is not controlled. On the long list of anthropogenic threats to the marine environment, open access still stands out as number one.

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Table 1. Summary of Parameterization

Equation(s)	Parameter Name	Value	Source/Comment
1	Lbar	1.20	From Neuse TMDL (adjusted down from 1.3 to account some for stock in the system)
1	omega	0.95	A slow decay rate based on persistence of nutrients
2	mu	0.40	Borsuk, Stow, and Reckhow (2004)
3, 4	rx	0.75	Eggleston et al. (2004)
3, 4	kx	25.85	Eggleston et al. (2004). Carrying capacity scaled down to reflect the size of the fishery in the Neuse River and Pamlico Sound only.
3, 4	alpha	0.37	Backed out using Excel's nonlinear solver and imposing the steady-state condition in patch 1 (or 2) with no hypoxia
3, 4	phi	0, 25, 50	0 indicates biological spatial connectivity all related to oxygen; scale intuition for others based on relative carrying capacities.
3, 4	chsi	0.50	Allows for a wide range of responsiveness (and the potential for collapses in patch 1), less than or equal to corresponding rx to prevent negative populations.
5, 6	beta	0.001	Chosen to scale predation such that abs(beta*kx) <ry
5, 6	ry	0.97	Clams are assumed fast growing (they are k-limited and not r-limited). Magnitude based on the survival regression in Borsuk, Powers and Peterson (2002) using minimum growth rate necessary to prevent population collapse under worst in-sample dissolved oxygen.
5	rho	0.79	Computed elasticity from Borsuk, Stow, Higdon, and Reckhow (2001) combined with Borsuk, Powers and Peterson (2002)
8, 9, 13	c	240.45	Rhodes, Lipton, and Shabman (2001) converted to 2002 dollars
9, 13	gamma	15.00, 45.37, 150.00	45.37 estimated jointly with 1-period ahead forecasts on catch per trip data (1982-2002) converting pots to trips and excluding 1994-1999 due to overreporting of pots. Pots converted to trips based on 2000-2002 trip data. Others allow for "slow" and "fast" adjustment.
10, 11	theta	0.40	Smith and Wilen (2003, 2004)
12, 13, 17	q	0.00003	Using parms from Eggleston et al. (2004) calibrated to total catch across time and pattern of catch
14, 16, 17	a	172, 160, 149 (mil.)	Backed out from a, Qtilde, and p(0).
14, 16, 17	b	-0.5, -0.99, -1.5	Inelastic, unit-elastic (set to -.99 for integrating out consumer surplus), and elastic regional demand
15, 16, 17	Qtilde	168 (mil.)	NC DENR - FMP (2004) and Eggleston et al. (2004)
	Neuse share of total NC	0.35	Share of the fishery in the Neuse River and Estuary + the Pamlico Sound (2000-02) from Eggleston et al. (2004)
Initial Conditions			
1	N(0)	1.26	Loadings divided by w (start off at steady-state stock)
2	A(0)	0.51	Algae starts in a steady-state.
3, 4	X_1(0)=X_2(0)	12.92	1/2 of kx (bundled to pop. Parameters)
4, 5	Y_1(0)=Y_2(0)	0.50	1/2 of y carrying capacity
11	pi_1(0)	0.50	Initial effort allocated uniformly over space.
13	E(0)	32,077	35% of effort in the fishery (mean over 2000-2002)
17	p(0)	0.87	Mean real price (weighted avg aggregated across hard, soft, and peeler, 1998-2002), in 2002 dollars using CPI Size D South

Table 2. Results of Policy Simulations under Open Access and Quasi-Optimized Fishery Management – 100 Year Time Horizon, 4.5% Discount Rate

			Open Access			Quasi-Optimized		
				30% Nitrogen Reduction			30% Nitrogen Reduction	
Speed of Adjustment gamma	Spatial Connectivity phi	Price Elasticity b	PV Baseline Rent (million 2002 $)	PV Rent Change (million 2002 $)	PV Consumer Surplus (million 2002 $)	PV Baseline Rent (million 2002 $)	PV Rent Change (million 2002 $)	PV Consumer Surplus (million 2002 $)
15	50	-0.5	86.530	5.357	24.982	267.210	6.993	22.004
15	50	-0.99	86.943	5.549	10.789	281.130	7.429	9.401
15	50	-1.5	87.058	5.605	6.742	286.080	7.315	5.704
15	25	-0.5	86.572	5.339	24.753	267.440	6.873	21.474
15	25	-0.99	86.997	5.527	10.708	281.420	7.270	9.144
15	25	-1.5	87.115	5.580	6.696	286.400	7.140	5.533
15	0	-0.5	85.819	5.723	25.229	269.220	5.982	17.908
15	0	-0.99	86.367	5.861	11.040	283.090	6.377	7.810
15	0	-1.5	86.523	5.899	6.931	287.910	6.533	4.974
45.37	50	-0.5	35.227	2.197	16.935	267.210	6.993	22.004
45.37	50	-0.99	35.165	2.208	7.170	281.130	7.429	9.401
45.37	50	-1.5	35.145	2.210	4.442	286.080	7.315	5.704
45.37	25	-0.5	35.249	2.191	16.834	267.440	6.873	21.474
45.37	25	-0.99	35.188	2.203	7.125	281.420	7.270	9.144
45.37	25	-1.5	35.170	2.202	4.417	286.400	7.140	5.533
45.37	0	-0.5	34.549	2.528	18.441	269.220	5.982	17.908
45.37	0	-0.99	34.538	2.523	7.933	283.090	6.377	7.810
45.37	0	-1.5	34.533	2.521	4.952	287.910	6.533	4.974
150	50	-0.5	11.836	0.667	11.928	267.210	6.993	22.004
150	50	-0.99	11.875	0.663	5.072	281.130	7.429	9.401
150	50	-1.5	11.893	0.659	3.153	286.080	7.315	5.704
150	25	-0.5	11.845	0.663	11.616	267.440	6.873	21.474
150	25	-0.99	11.884	0.661	5.059	281.420	7.270	9.144
150	25	-1.5	11.904	0.655	3.160	286.400	7.140	5.533
150	0	-0.5	11.511	0.844	14.553	269.220	5.982	17.908
150	0	-0.99	11.551	0.840	6.265	283.090	6.377	7.810
150	0	-1.5	11.568	0.841	3.879	287.910	6.533	4.974