



ACADEMIC
PRESS

Available online at www.sciencedirect.com

SCIENCE @ DIRECT®

JOURNAL OF
ENVIRONMENTAL
ECONOMICS AND
MANAGEMENT

Journal of Environmental Economics and Management 46 (2003) 183–206

<http://www.elsevier.com/locate/jeem>

Economic impacts of marine reserves: the importance of spatial behavior

Martin D. Smith^{a,*} and James E. Wilen^b

^a *Nicholas School of the Environment and Earth Sciences, Duke University, Box 90328, Durham, NC 27708, USA*

^b *Department of Agricultural & Resource Economics, University of California, Davis, USA*

Received 15 March 2002

Abstract

Marine biologists have shown virtually unqualified support for managing fisheries with marine reserves, signifying a new resource management paradigm that recognizes the importance of spatial processes in exploited systems. Most modeling of reserves employs simplifying assumptions about the behavior of fishermen in response to spatial closures. We show that a realistic depiction of fishermen behavior dramatically alters the conclusions about reserves. We develop, estimate, and calibrate an integrated bioeconomic model of the sea urchin fishery in northern California and use it to simulate reserve policies. Our behavioral model shows how economic incentives determine both participation and location choices of fishermen. We compare simulations with behavioral response to biological modeling that presumes that effort is spatially uniform and unresponsive to economic incentives. We demonstrate that optimistic conclusions about reserves may be an artifact of simplifying assumptions that ignore economic behavior. © 2003 Elsevier Science (USA). All rights reserved.

JEL classification: Q22

1. Introduction

An important paradigm shift is underway in marine policy that will profoundly alter the future management of our coastal resources. The shift is toward the use of spatial zoning measures that will effectively partition the ocean into a system of areas with regulated exploitation and areas protected with marine reserves.¹ A confluence of emerging science and political interest in marine

*Corresponding author.

E-mail address: marsmith@duke.edu (M.D. Smith).

¹ The emerging literature on marine reserves uses a variety of terms that are sometimes synonymous and sometimes not. We will use marine protected areas, marine reserves, and no-take zones interchangeably to mean areas in which no exploitation is permitted, including especially commercial and recreational fisheries exploitation.

protected areas provides the underpinnings of this new vision. In the sciences, there is growing consensus among marine ecologists, biologists, and many fisheries managers that conventional season length and gear restriction management methods have failed, are bound to fail in the future, and that a new approach is therefore needed. In the political arena, there is a growing view among NGOs and influential environmental lobbying groups that a network of protected areas similar to our terrestrial park system will best achieve long-term biodiversity and conservation goals.

Up to this point, biologists have promoted this new spatial zoning view with little input solicited from economists and other policy analysts. But as real proposals emerge for specific systems of protected areas, there will be increasing calls for economic analyses of actual policy options. Economists are only beginning to think about how to introduce space into conventional models of renewable resources and how spatially differentiated policies might compare with second best undifferentiated policies. With a case study of the northern California red sea urchin fishery, we present a comprehensive empirical investigation of how marine reserves would perform as a fishery management tool. We analyze both economic and biological consequences of implementing reserves. To this end, we focus particularly on harvester spatial behavior, and we assess the degree to which accounting for economically driven behavior alters the conclusions of recent biological investigations of marine reserves. We find, somewhat to our surprise, that the main simplifying modeling assumptions made by biologists to handle harvester behavior do not actually “cancel out” in the final analysis of reserves as a fishery policy instrument. Instead, virtually every economic simplification in biological modeling biases the case in favor of using reserves to manage fisheries. Our results thus call into question whether the optimism displayed for reserves as a fisheries management tool is warranted.

In Section 2 we review both the biological and economic literature on marine reserves. We then describe the sea urchin fishery case study in Section 3 and focus particularly on its spatial character and its short- and long-term dynamics. In Section 4 we discuss and estimate a model of spatial behavior and then use it, in Section 5, to simulate the impacts of marine reserves with an integrated bioeconomic model. In Section 6 we draw some broad conclusions for both research and policy analysis of marine protected areas.

2. Related biological and economic work

The notion of using closed areas as a fishery management tool emerged over a decade ago among marine ecologists and conservation biologists. The first proposals were modest and focused on reserves as laboratories, calling for small areas off coastal research institutes in which ecologists could study unexploited systems in order to gauge the ecological impacts of exploitation. By the early 1990s, the idea had morphed into a grander vision that called for significant areas to be set aside, often on the order of 20–30% of the coastline. The transformation in the scale of the proposals coincided with several important papers on fisheries management. Most of these studies concluded that the world’s fisheries were in a state of crisis, that conventional methods were to blame, and that a new approach to management was needed.²

²See for example the oft-cited paper by Ludwig et al. [13], which argues that reductionist approaches cannot overcome the natural variation and irreducible uncertainty inherent in natural systems in order to guide us toward

Early modeling papers thus focused on the impact that permanent spatial closures might have on maintaining fisheries at safe levels of exploitation.

The first modeling work on marine reserves is Polacheck [20], which uses a Beverton and Holt [1] model to examine how a spatial closure might affect an exploited fishery. By assuming exogenous larval recruitment in the fishery and exogenously determined fishing effort, Polacheck shows that marine reserves always increase spawning biomass within the reserve itself and under some circumstances, but not many, reserves increase fishing yield. Empirical studies of reserves proceeded to confirm the finding that spawning biomass increases *within* closed areas [7].

One may not need voluminous empirical work to believe that removing exploitation from an ecosystem will increase biomass and broaden the age distribution by providing protection for larger and more fecund fish. Less certain, however, is whether the increase in spawning stock biomass and its composition within a reserve can provide a net increase to the fishery *outside* the reserve.³ The first generation modeling work identifies mechanisms that are germane to this question, namely the mobility of adults and the level of pre- and post-reserve exploitation in the fishery. These studies show that reserves most likely provide a net yield increase from migration into the open area when adult mobility is neither too low nor too high. The intuition is that, for a given size reserve, adults must not move so widely that the reserve does not afford them protection, but there must be some movement of adults (or juveniles) to increase the fishable abundance in the remaining open area.

The second generation of papers generalizes the earlier approaches by closing the relationship between spawning biomass and recruits to the fishery. This is an important addition to understanding because, in addition to spillover of adults, protected areas may also produce eggs and larvae that subsequently redistribute to exploitable populations in the remaining open areas. A few studies from the late 1990s assume that adults within and outside the reserve produce larvae, which disperse in some manner and then recruit into both the fishery and the reproductive population.⁴ A significant and frequently cited second generation paper is by two economists, Holland and Brazee [10]. In a detailed age-structured two-patch population model, they depict sophisticated biological mechanisms, including density-dependent stock/recruitment relationships in both the reserve and open area, migration of adults according to a density-dependent mechanism, and (uniform) larval dispersal. Perhaps more importantly, their model incorporates economic variables and is fully dynamic so that it computes the present values of transition paths. Holland and Brazee confirm the Polacheck results that spawning stock biomass will always

(footnote continued)

anything like sustainable yield. They urge that managers “confront uncertainty” by explicitly accounting for it in common sense ways including taking actions that are robust to uncertainty, that allow monitoring and learning, that are reversible, that hedge risks and that incorporate scientific principles from decision making under uncertainty. Walters [29] provides more detail on these ideas in his important book on adaptive management.

³Some of the fisheries literature does not seem aware that it is a net increase overall and not just a gross increase in yield that is needed in the remaining open areas to make a reserve worthwhile as a fisheries enhancement measure. That is, from the fishing industry perspective, it is not enough that a reserve increases harvest outside the reserve, but rather that the increase be large enough to compensate for the area removed from fishing. In general, this is a significant hurdle, and as the reserve size gets larger, *ceteris paribus*, this hurdle rises.

⁴Most models assume a common pool dispersal process whereby total system larvae are distributed instantaneously and uniformly over the whole system.

increase with reserves. They also find that whether this increase creates conditions to generate a net increase in the present value of economic benefits depends importantly on the discount rate and the pre-reserve exploitation level, as well as bioeconomic parameters. The role of discounting is intuitive because reserves always decrease harvests initially and then increase harvests as spillovers begin to emerge. At high discount rates it may not be worth the sacrifice necessary to rebuild sustainable harvests to higher levels. Thus, the discount rate is an essential determinant of whether reserves generate net economic benefits. In contrast, the first generation modeling ignored transition paths and hence missed this important characteristic of reserves. Holland and Brazee find that pre-reserve exploitation rates are the other important determinant of whether a reserve pays off; dramatically, overharvested fisheries are more likely to be worth an investment in rebuilding the overall biomass by using a reserve.⁵

One key simplifying assumption in virtually all of the early analyses of reserves is that effort is fixed both before and after reserve formation. Under a spatial closure, most analysis presumes that effort simply displaces to the remaining open area. This analysis thus does not account for the fact that economic conditions will, in part, determine pre-reserve fishing effort and that the reserve itself will alter relative profitabilities and hence subsequent effort decisions by fishermen. Sanchirico and Wilen [23,24] relax this assumption in a conceptual analysis that determines spatially explicit effort endogenously.⁶ They find that, under open access, most reserve scenarios produce a biological benefit but that there are very few combinations of biological and economic parameters that give rise to both a harvest increase and a biological benefit. In particular, they find that harvest increases are likely only when the designated reserve patch has been severely overexploited in the pre-reserve setting.

Taken as a whole, the conceptual biological and economic literature warrants a serious empirical investigation of reserves that accounts for spatial and dynamic aspects of the biological system and the harvest sector. In this paper we consider how incorporating realistic depictions of harvester behavior affects the implications of marine spatial closures. A priori, economically motivated harvester behavior ought to matter in several ways. First, since the pre-reserve status quo is important to the net effect of a policy change, it should be of interest to know how economic variables condition the initial circumstances in a spatial bioeconomic system. Second, since a spatial closure will affect the subsequent spatial distribution of relative economic returns, we expect that effort redistribution will have complicated spatial and intertemporal effects, both in the short run and in the long run. Third, since most real spatial systems embody complicated biological and economic heterogeneities that affect profit differentials over space, we expect these to color the conclusions about the economic impacts of reserves. For all of these reasons, simplified assumptions about effort distribution and its determinants are likely to confuse the debate about these new forms of marine policy instruments.

⁵ At the same time, what this shows more fundamentally is that it can be a sound investment to rebuild an overexploited stock by reducing effort. Closing a fraction of a fishery's area is one way to reduce overall fishing mortality (as Polacheck [20] concluded), but another way is to simply crank down conventional methods of effort and fishing mortality control. Hastings and Botsford [9] have established that area controls are equivalent to conventional effort controls under certain reasonable circumstances. Some fisheries observers point out, however, that closed areas may be easier to enforce than conventional effort control measures such as mesh size, days at sea, etc.

⁶ Hannesson [8] also allows for endogenous effort in an open access model of marine reserves but incorporates less generality in the biological and economic models than Sanchirico and Wilen [23,24].

3. Case study: the northern California red sea urchin fishery

The northern California red sea urchin fishery is an ideal case study with which to examine empirically marine reserves and the dynamics of harvester spatial behavior. Urchin population dynamics are consistent with a biological structure that is favorable for reserve formation, and regulators are currently considering spatial management in the urchin fishery.

Red sea urchins, *Strongylocentrotus franciscanus*, are found along the Pacific Coast in rocky inter-tidal kelp forests from southern California to Alaska.⁷ Urchins have hard spiny shells and are harvested for the gonads (roe) found inside their shells. Divers harvest urchins on 1-day trips to fishing grounds. Their vessels are designed especially to travel fast to the fishing locations. Once on a site, a diver begins harvesting while connected to the surface by a line that supplies compressed air from the boat. A tender aboard the vessel monitors the air compressor and the diver's line. Harvesters use rakes to remove urchins from the rocky bottom, filling mesh bags that are then loaded onto the vessel by the tender. Weather conditions are critical determinants of when and where divers choose to dive, particularly in northern California where a trip is typically made only 14% of available open days.

At the end of a fishing day, harvesters deliver the whole urchins to processors. At the processing plant, workers split the shells, scoop out the gonads, and then wash and bathe the gonads in alum solutions to firm the roe. Roe skeins are then carefully packed in special wooden trays holding 250 g, and the product is shipped overnight to Japan. In Japan, the roe is sold mainly at the Tokyo Central Wholesale Market, in competition with roe from Japan and other North American fisheries. The roe is relatively high valued, with current Tokyo Wholesale prices in the range of \$25 per pound, translating into a markdown to divers of approximately \$1 per pound of whole urchin. A typical day trip currently brings in 750 pounds of urchin, grossing \$750 per trip for the diver/tender team.⁸

Sea urchins have been harvested in southern California since the early 1970s and in northern California since 1988.⁹ Regulatory restrictions include closed seasons, minimum size limits, and a limited entry program. Current regulations require that northern California urchins be at least 3.5 inches in diameter,¹⁰ a size reached at approximately 6 years of age. A full closure is in effect throughout all of July, and 1-week closures are operative from May to September. Within each open week, 3-day per week openings are in force in June and August, and 4-day openings prevail in April and October. In total, the number of potential open days per year is approximately 240 in northern California. The limited entry program was introduced into the California-wide fishery in

⁷ We draw the factual background in this section from Kato and Schroeter [12], Kalvass and Hendrix [11], the log book and landings ticket data described below, and the website of the Sea Urchin Harvesters Association of California, <http://www.seaurchin.org>.

⁸ The recovery of roe from a whole urchin is about 10%. Hence, ten pounds of whole urchins are needed to generate a pound of roe, mostly explaining the markdown.

⁹ Ironically, urchins were considered pests by abalone divers in the 1960s because they competed for the lucrative abalone for habitat and food. Abalone divers often poured quicklime on urchins in order to remove them and open up habitat for the commercially valuable abalone. In the 1970s, a market for California red sea urchins developed as the Japanese stocks were overharvested [22]. The fishery was harvested mainly in southern California by ex-abalone divers after the abalone fishery collapsed. The continued growth in the Japanese market led to opening the Northern California fishery in the late 1980s.

¹⁰ Sea urchin diameter is measured across the test, i.e. the urchin shell from which the spines extend.

1989, grandfathering all existing permit holders (roughly 850) into the fishery. The program has tightened over the intervening years in order to steer participation to a long-term goal of 300 divers. Each diver must now land 300 pounds per trip for a minimum of 20 trips during either the current or preceding year in order to be granted a license.¹¹ The license holder must do the fishing, and licenses may not be leased or sold.

Sea urchin biological characteristics are consistent with those identified by biological modelers as holding promise for successful use of reserves. First, urchins are “patchy” in that they are found in multiple discrete areas where substrate and habitat characteristics are suitable for the species. Second, adult movement within each patch is relatively low (an average of 7–15 cm per day) and hence closed areas promise protection of spawning biomass. Urchins reach sexual maturity approximately at age 5, and egg production increases with age at an increasing rate.¹² Third, larvae are redistributed considerable distances from spawning areas by currents, winds, and sea surface changes and hence protected urchins have the capability of replenishing and sustaining remaining open areas.

In addition to favorable biological characteristics, the urchin fishery is also an ideal case study for examining spatial behavior of harvesters. Most importantly, urchin trips are day trips that are made repeatedly by the same individuals under conditions that vary with biological and economic variables. This means that it is possible to assemble a large panel data set with substantial intertemporal variation in determinants of behavior. The data set that we use to examine spatial harvester behavior in the northern California sea urchin fishery is unusually rich. It consists of over 57,000 actual individual dives made by up to 358 divers in each year over the period between 1988 and 1997. Divers record information in mandatory logbooks and on landings tickets. Each landings ticket reports port of landing, processor code, quantity landed, price paid, and diver code. Each logbook entry contains dive location (latitude), dive duration, average depth, and divers per vessel. We also collected daily weather data on wind speed, wave height, and wave period from weather buoy records, averaged it over the period preceding a typical trip decision, and linked it to each potential dive trip decision. The fact that each trip is a day trip is convenient since we can model decisions as repeated nested discrete choices.¹³ The model that we report here estimates parameters of a repeated choice structure for several hundred divers making decisions over ten years and amounts to over 400,000 choice occasions.¹⁴ During this period, the urchin fishery was harvested from a virgin fishery to the present level approaching a steady state in which returns to fishing equilibrate across space. Prices vary over the period as a result of exchange rate changes, quality changes, and landings variability. Weather conditions vary on a daily basis and with some inter-annual variation as well.

¹¹ Provisions are also made to allow one new entrant for each ten licenses retired.

¹² Brown and Roughgarden [5] make a similar returns to scale argument to show that harvesting one patch from a metapopulation can be the optimal strategy.

¹³ A repeated decision structure is more tractable than trying to model situations with a fixed end point. In addition, fisheries in which multi-day and multi-area trips are made must contend with the complications introduced by decisions that are essentially searching decisions made along the journey to a targeted destination.

¹⁴ Total choice occasions are days for which the urchin fishery is open to harvesting. Because weather conditions so frequently keep fishermen ashore, the actual number of trips is only a fraction of total choice occasions, about 14% of open days. In addition, Sunday closures in the Tokyo Central Wholesale Market induce lagged responses that reduce diving activity on Fridays and Saturdays.

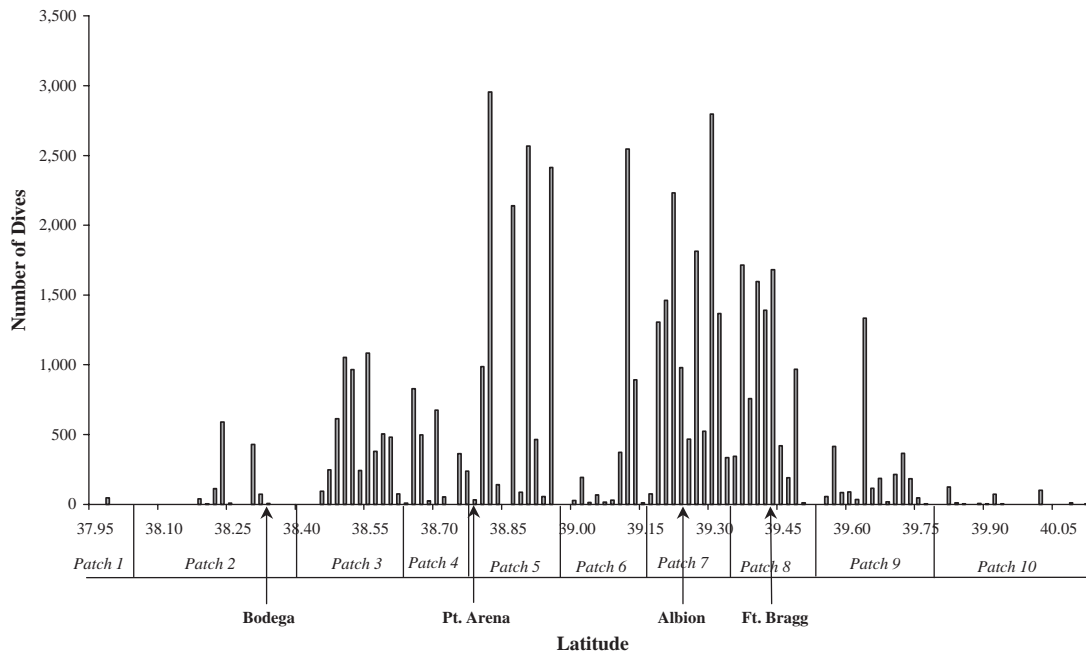


Fig. 1. Diving activity coastal histogram. Trips in North-Central California 1988–1997.

In northern California, harvesters land sea urchins in six ports. From south to north, the ports are Half Moon Bay, Bodega, Point Arena, Albion, Fort Bragg, and Crescent City. Crescent City near the Oregon border and Half Moon Bay on the San Francisco peninsula contribute less than 10% of the total harvest, while the remaining harvest is landed at the four ports in the middle, all within 200 road miles north of San Francisco International Airport.

To begin our analysis of the urchin fishery, we identify and define patches. In our definition, a patch is a bioeconomic concept, not purely a biological one. Thus, patches are discrete subpopulations of *harvested* sea urchins. Before settling on patch definitions, we first looked at the spatial distribution of fishing effort generally. Not surprisingly, fishing effort fans out in a manner that declines geometrically with distance from the port. Because the four main ports are relatively close, when aggregating effort across ports, there is considerable overlap. That is, some areas are fished from multiple ports. Still, there are areas close to ports in which no exploitation occurs. Using these effort data and knowledge that there are breaks in suitable sea urchin habitat, we partitioned the entire data set into eleven patches along the coast of northern California. Fig. 1 is a histogram over the whole period with diving activity characterized by “patch” number.¹⁵ With the exception of the Farallon Islands, patches correspond to latitudinal corridors that extend from the shore out to sea. Reducing the patches to a single latitude-based spatial dimension is sensible in this fishery because urchins only live close to shore. The *x*-axis shows degrees of latitude (converted to decimals rather than degrees and minutes). The figure shows the large concentrations of effort located near the four ports with highest landings, together with breaks between patches associated with substrate and habitat variation. Throughout the fleet, divers

¹⁵ We do not show the eleventh, which is off the Farallon Island off San Francisco Bay. See [27] for details.

Table 1
Diver spatial mobility across patches

No. of patches active in	Number of active divers											
	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
1	50	60	146	139	139	99	107	49	37	36	43	43
2	44	51	72	72	62	65	67	48	44	45	33	39
3	20	19	59	60	52	46	45	30	25	33	27	24
4	10	8	36	32	36	32	25	21	21	22	18	9
5	4	0	20	23	38	18	13	15	0	7	5	9
6	1	0	6	12	13	15	13	3	4	3	3	1
7	0	0	2	8	15	9	2	1	0	0	1	0
8	0	0	0	4	3	0	0	1	0	0	0	0
9	0	0	1	3	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0	0	0	0	0	0
11	0	0	0	0	0	0	0	0	0	0	0	0
Total divers	129	138	342	353	358	284	272	168	131	146	130	125
Weighted average no. patches	2.05	1.82	2.25	2.53	2.68	2.60	2.33	2.54	2.35	2.51	2.40	2.24

show considerable spatial mobility across patches, as Table 1 demonstrates. In any given year, individuals visit between one and nine patches with fleet annual average patches visited generally ranging between 2 and 3 over the sample period.

In Table 1, one can also see the open access implications of opening up the previously unexploited northern California fishery starting in 1988 before regulators began to act. From zero activity in 1987, harvesters from the pool of divers in the southern California fishery and elsewhere were drawn rapidly into the new fishery, with over 100 new divers participating in 1988. The northern California part of the fishery peaked in 1992 with 358 total divers. During the peak period, some divers fished in as many as 6–8 northern California patches during the year, although the majority concentrated on one or two patches from a single port. As the northern California fishery matured and as the unexploited stocks were drawn down and profitability reduced, regulations were implemented, and fishermen exited, some returning to the southern California fishery and others dropping out entirely in response to the stringent limited entry requirements. Total California-wide numbers dropped from 826 in 1994 to 440 in 1998, whereas in northern California, numbers dropped from 358 to about 125 in recent years. In the last couple of years, there has been a slight reduction in average mobility across northern California ports, and the fishery appears to have settled into a pattern approximating a bioeconomic equilibrium, with harvesters fishing more intensively to maintain a living under reduced abundance conditions. The average number of trips per year has increased from about 26 per year in 1992/93 to a current level of about 34 trips per year. Average depth per dive (mean 35.45 ft over all years/locations) has similarly increased at a rate of about 1 foot per year as the extensive margins expanded out from all port locations.

4. Modeling spatial behavior

In this section, we report results using the logbook/landings ticket-based data set on divers' daily spatial behavior to estimate a model of individual choice. It is convenient to begin with the hypothesis that individual divers make daily choices that maximize some random utility function:

$$U_{ijt} = v_{ijt} + \varepsilon_{ijt} = f(\mathbf{X}_{it}, \mathbf{Z}_{i1t}, \mathbf{Z}_{i2t}, \dots, \mathbf{Z}_{iMt}; \boldsymbol{\theta}) + \varepsilon_{ijt}, \quad (1)$$

where \mathbf{X}_{it} includes harvester-specific and time-specific characteristics that are constant across choices, \mathbf{Z}_{ijt} includes choice-specific characteristics such as travel costs and expected resource abundance that may also be harvester- and time-specific, $\boldsymbol{\theta}$ is a parameter vector, and ε_{ijt} is a random component that is unobservable to the analyst. This random utility model posits that, given M possible dive locations and the possibility of not diving, diver i in period t will choose location k if the utility of choice k is higher than that of the other $M - 1$ location choices as well as the choice of not to dive in period t .

The nesting structure for daily choices is straightforward. On any given open day, divers choose to go or not go fishing. If they choose to go fishing, they choose among the set of locations. For identification, we normalize the utility of not going to zero. Note, however, that the utility of not diving captures the utility of leisure, work opportunities outside of fishing, and the value of avoiding exposure to unsafe diving conditions.

Although there are numerous discrete formulations that can represent this choice structure, we adopt a Repeated Nested Logit (RNL) framework similar to Morey et al. [17] and based on McFadden [14].¹⁶ Smith [28], which to our knowledge is the first analysis to model discrete participation and location choices jointly in commercial fisheries, identifies reasons for taking this approach in modeling spatial decisions of commercial fishing fleets. Here we summarize these reasons with an eye toward building a bioeconomic simulation model. First, a forward-looking model is unnecessary because even under limited entry, there is some element of non-excludability in harvest, and fishers do not perceive that they will realize future gains from restraining present harvest. Second, the RNL is relatively easy to estimate with the large size data set that we have. A more advanced technique that relies on simulation-based estimation, e.g. Mixed Multinomial Logit [16], could require a super computer to estimate over this 400,000 observation data set. Third, we need an estimation scheme with a closed form solution in order to embed the estimated behavioral choice model into the bioeconomic simulation model discussed in the next section.¹⁷ Finally, the RNL also has well-known properties that are desirable in general, including the flexibility to admit different variances at different decision nodes.¹⁸

¹⁶The choice structure is basically the same as the RNL proposed by Morey et al. [17], which deals with repeated participation and site choice in recreation demand. The key difference in our setting is that we do not explicitly model income effects, and our site choice is conditional on the diver's port. In contrast, Morey et al., in dealing with recreation demand, do model income effects and include separate decision branches for region choice followed by specific site choice.

¹⁷In contrast, a simulation-based estimation approach would require nesting complicated numerical integrations within each time step of the bioeconomic simulation model below and would lead to computational infeasibility.

¹⁸The drawbacks of RNL are equally well known and include assumptions of independence across both time and individuals.

Following McFadden [14], we assume that the ε_{ijt} is independently and identically distributed generalized extreme value. We further assume that utility is linear in individual- and choice-specific variables so that the following model characterizes individual choices:

$$Pr(\text{Go to } j) = \frac{\exp\left\{\frac{\mathbf{z}'_j \boldsymbol{\gamma}}{(1-\sigma)} + \mathbf{x}'_j \boldsymbol{\beta} + (1-\sigma)I\right\}}{\sum_{k=0}^{10} [\exp\left\{\frac{\mathbf{z}'_k \boldsymbol{\gamma}}{(1-\sigma)}\right\} + \exp\left\{\frac{\mathbf{z}'_k \boldsymbol{\gamma}}{(1-\sigma)} + \mathbf{x}'_k \boldsymbol{\beta} + (1-\sigma)I\right\}]} \quad (2)$$

and

$$\begin{aligned} Pr(\text{Do not go}) &= 1 - \sum_{k=0}^{10} Pr(\text{Go to } k) \\ &= \frac{1}{1 + \exp[\mathbf{x}'_t \boldsymbol{\beta} + (1-\sigma)I]}, \end{aligned} \quad (3)$$

where

$$I = \ln \left[\sum_{k=0}^{10} \exp \left\{ \frac{\mathbf{z}'_{kt} \boldsymbol{\gamma}}{(1-\sigma)} \right\} \right]. \quad (4)$$

In the above equations, the i subscripts for the individuals are suppressed since the form of the model is the same for each individual in the data set. Some characteristics \mathbf{x} and \mathbf{z} could, in principle, vary across individuals. Here $\boldsymbol{\beta}$ denotes the parameter vector for characteristics that vary across individuals and/or choice occasions but not across choices, $\boldsymbol{\gamma}$ is the parameter vector for choices that vary across choices, and $(1-\sigma)$ is the coefficient on the nested logit inclusive value.

Table 2 reports the results of a parsimonious specification using the entire 401,151 observation data set.¹⁹ Variables that are not choice-specific include three coast-wide weather variables and a day-of-week dummy variable. The weather variables are WP (wave period), WS (wind speed), and WH (wave height), all computed as averages over weather buoy data for the 12-h period preceding noon of the day in question. The day of the week dummy (DWEEK) is one on Friday, Saturday, and Sunday, reflecting reduced propensity to dive on weekends and on days that precede the Sunday closure of the Tokyo Central Wholesale Market. Variables that are location-specific are distance to the center of the patch in question (DISTANCE), and expected revenue (ER) in each patch.²⁰ The inclusive value is computed from the lower level utility branch.

¹⁹Our strategy for estimating the model involved first drawing a random sample of 30 divers followed over the entire period. This relatively smaller sample of 27,000 choice occasion observations was used for preliminary specification testing. We used the smaller subset to determine the effects of using different backward lags for our expectations variables, different averaging methods for prices, and the impacts of various diver-specific variables computed from the data set. See [28] for empirical results from this smaller data set and comparisons to a more aggregated econometric approach.

²⁰Some of the methodological issues associated with computing expected revenues are explored in [26]. Expected revenues reported here are calculated as the product of expected price and expected catch per trip. The former is a rolling 1-month backward looking average across the entire northern California fishery, so that for each day the expected price is based on the average price for the previous 30.4 days. Expected catch is a patch-specific rolling 1-month backward looking average. These proved to be best fit in our preliminary specification tests. In addition, 1-month averages are convenient for simulating the model in the next section.

Table 2
Nested logit estimates

Variable	Coefficient	Standard error	Z-statistic
<i>Not location-specific</i>			
Constant	1.06	0.048	22.21
WP	−0.18	0.005	−34.69
WS	−0.11	0.003	−36.69
WH	−0.74	0.011	−70.36
DWEEK	−0.74	0.012	−60.02
<i>Location-specific</i>			
DISTANCE	−7.27	0.036	−203.72
ER	0.08	0.001	65.17
σ	0.22	0.027	8.34
Log-likelihood	−189,878		
Observations	401,151		
Pseudo- $R^2(1)$	0.21		
Pseudo- $R^2(2)$	0.81		

Pseudo- $R^2(1)$ is based on the log-likelihood in a conditional logit model with choice-specific constants. Pseudo- $R^2(2)$ is based on the log-likelihood of $n \ln(1/J)$, where $J = 12$ possible choices.

As Table 2 shows, all variables have signs as expected and all are highly significant as one would expect.²¹

These estimated RNL equations may be used to generate simulated effects of changes in economic variables, such as a change in expected revenues. An increase in expected revenues anticipated in patch k causes a *spatial substitution effect* as effort is drawn from other patches. However, the increase in expected revenues in patch k will also have a *participation effect* because urchin fishing becomes a more attractive use of time overall. This participation effect will cause more effort to be distributed over all possible patches. The upshot is that the own effect of an increase in expected revenues will always be positive. The cross effects may be positive or negative, depending upon whether the participation effect outweighs the substitution effect. In general, the signs of cross effects will hinge on all of the variable values, and the substitution effect will be larger for patches that are visited more frequently (because $Pr(k)$ is higher). Table 3 shows some of the computed elasticities of patch choice with respect to revenues.²² A 10% change in expected

²¹ Since σ is significantly different from zero (and from one), the inclusive value coefficient is between zero and one. This has two implications. First, it suggests that the model is globally consistent with stochastic utility maximization [6,15]. Second, it suggests that variances of utility for participation and for location choices are different. This implies that choices across branches of the decision tree are less similar than choices within each branch of the tree. For example, choosing between patches 7 and 8 on a given day is more similar than choosing between 8 and not going fishing on that day.

²² These are computed assuming DWEEK = 0. In addition, since there are 11 travel distances from each port, we average across probabilities associated with these actual distances rather than using the sample mean distance. Using a sample mean would effectively act as if all divers came from a fictive port in the middle of the coast rather than from their actual ports. We also compute elasticities with respect to the weather variables, finding elasticities

Table 3
Nested logit cross-revenue elasticities. Mid-week distance adjusted

		1% Change in ER_j											
		j	0	1	2	3	4	5	6	7	8	9	10
%	k												
Change in P_k	0		1.515	0.042	0.085	0.080	0.086	0.065	0.058	0.047	0.043	0.058	0.112
	1		0.130	0.598	0.093	0.097	0.104	0.080	0.071	0.057	0.052	0.071	0.136
	2		0.117	0.045	1.212	0.088	0.094	0.072	0.064	0.051	0.047	0.064	0.122
	3		0.119	0.046	0.094	1.145	0.095	0.073	0.065	0.052	0.048	0.065	0.124
	4		0.117	0.045	0.093	0.088	1.221	0.071	0.064	0.051	0.047	0.064	0.122
	5		0.124	0.048	0.098	0.093	0.099	0.938	0.067	0.054	0.050	0.067	0.129
	6		0.126	0.049	0.100	0.094	0.101	0.077	0.836	0.055	0.051	0.069	0.131
	7		0.129	0.050	0.102	0.097	0.103	0.079	0.070	0.672	0.052	0.070	0.134
	8		0.130	0.050	0.103	0.097	0.104	0.079	0.071	0.057	0.620	0.071	0.135
	9		0.126	0.049	0.100	0.094	0.101	0.077	0.069	0.055	0.051	0.837	0.131
	10		0.104	0.041	0.083	0.078	0.084	0.064	0.057	0.046	0.042	0.057	1.580
Net change in P_{GO}			2.736	1.063	2.163	2.051	2.190	1.675	1.491	1.196	1.103	1.492	2.857

revenues off patch 8, for example, will result in an increase in total participation of 11.03%. This increase in effort will be distributed mostly to patch 8 which will experience a 6.2% increase, and in small amounts averaging 0.5% to each of the other patches. We use this model of spatial choice to predict the dynamic and spatial distribution of effort to be used in the biological model discussed next.

5. An integrated bioeconomic model of reserve formation

In this section we outline a spatially explicit and dynamic bioeconomic model of the sea urchin fishery. The model is innovative in several respects. First, the model is a true bioeconomic model in that it integrates a population model of sea urchins with a behavioral model of the harvesting sector and generates joint bioeconomic equilibria. Second, the biological model is explicitly spatial and dynamic. We depict the sea urchin population as a metapopulation of 11 discrete patches, each with its own natality/mortality and growth parameters. The populations are linked with a dispersal matrix capable of characterizing any type of qualitative dispersal pattern. We parameterize the dispersal matrix with parameters calibrated to mimic field observations of larval settlement along the northern California coast. Third, the economic model is also explicitly spatial and dynamic. We use the model discussed above to depict industry behavior as an aggregation of individual choices made by divers, each of which is presumed responsive to the relative expected profitability of participation and location. Fourth, the economic and population

(footnote continued)

of: WP(−1.1), WS(−0.51), and WH(−1.42). These suggest that wave height is regarded as more risky than wind speed, a finding that seems sensible when one considers the process of trying to hold a vessel steady in high waves as a diver climbs in and out of the boat and the tender unloads urchins.

models explicitly account for size limits, season restrictions, and limited entry. Finally, the economic model of harvester behavior and the biological model are linked and integrated over both time and space. This allows us to experiment with different spatially explicit policies, change economic and biological parameters, and trace out both short run impacts, long run steady state impacts, and the dynamic and spatial adjustments that take place in transition to steady states.

5.1. The metapopulation model

The metapopulation model developed to examine spatial management policies in the red sea urchin fishery consists of 11 discrete age- and size-structured subpopulations linked by a dispersal matrix.²³ Each separate subpopulation has a size structure described by a von Bertalanffy equation, so that the size of an individual of age a in patch j is given by

$$Size_{j,a} = L_{\infty}^j (1 - e^{-k_j a}), \quad (5)$$

where a is a monthly time index from 1 to 360 and L_{∞}^j and k_j are patch-specific growth parameters. Note that L_{∞}^j is the terminal size of an individual organism, and this parameter ultimately dictates the maximum amount of biomass per individual. The model begins computations with a set of initial abundance matrices for each site. To generate these initial abundance matrices, we simulate a non-harvested steady state that mimics the situation in 1988 when the northern California resource first came under exploitation. The populations are then aged by advancing the abundance values for each month to the next older month so that $A_{i,a} = A_{i,a-1}$, where A denotes the number of organisms in the cohort. After the populations are aged, the numbers surviving in the population are computed, along with the catch. Survival is determined by a Beverton–Holt mortality relationship, which embeds both patch-specific natural mortality rates m_j as well as fishing mortality rates f_j if the size is above the minimum size limit L_{limit} . We link the economic model of diver behavior to the population model by making monthly fishing mortality rates a function of predicted diver trips. Accounting for both natural and fishing mortality, survival of the number of individuals to age a becomes

$$A_{j,a} = \begin{cases} A_{j,a} e^{-m_j} & \text{if } Size_{j,a} < L_{\text{limit}}, \\ A_{j,a} e^{-m_j - f_j} & \text{if } Size_{j,a} > L_{\text{limit}} \end{cases} \quad (6)$$

and total catch (C) consists of the sum of harvests of all sizes greater than the minimum size over all patches, which is

$$C = \sum_{j=0}^{10} \sum_{a=0}^{360} \frac{f_j}{m_j + f_j} (1 - e^{-(f_j + m_j)}) w Size_{j,a}^b A_{j,a}, \quad \forall Size_{j,a} > L_{\text{limit}}, \quad (7)$$

where w and b are allometric parameters relating weight and urchin test diameter. These parameters essentially convert number of organisms of each size to an aggregate measure of biomass, and harvest is a function of that biomass based on the fishing mortality parameters f_j . Note that for a given organism terminal size, L_{∞}^j , there is a corresponding terminal biomass. This, in turn, implies a maximum possible catch for a given number of organisms. The allometric

²³The metapopulation model is more fully described in [2].

parameters give rise to the possibility of an increasing returns production technology because $b > 1$ is the usual case, which means that the second derivative of catch with respect to size is positive.²⁴

The metapopulation model also computes egg production, larval dispersal, settlement and survival. Egg production is computed after survival has been calculated for each month. If the month is a spawning month, then egg production in patch j is computed with

$$e_j = \sum_{a=0}^{a=360} \alpha x^\beta A_{j,a}, \quad \text{where } x = \begin{cases} \text{Size}_{j,a} & \text{if } \text{Size}_{j,a} > L_{\text{maturity}}, \\ 0 & \text{if } \text{Size}_{j,a} < L_{\text{maturity}}. \end{cases} \quad (8)$$

This equation sums the egg production from each size class, where there is only positive production for sizes greater than the size at reproductive maturity. The exponent on size (β) is greater than one, since egg production is known to be increasing and convex in organism size. Thus, the egg production relationship gives rise to another dimension of increasing returns production technology due to a combination of the minimum sexual maturity size and the positive second derivative of the egg production with respect to size.

After eggs are produced, they are distributed spatially over the system, using a dispersal matrix which can take on a number of different qualitative forms. During the months in which larval dispersal is assumed to take place, settlement of larvae is calculated. For each month of the egg production period, a fraction of egg production is presumed to survive and this is distributed via the dispersal matrix from each of the patches to each individual patch according to

$$\mathbf{s}^{\text{in}} = p\mathbf{D}\mathbf{e}. \quad (9)$$

This 11×1 vector gives the array of settlement associated with the array of egg production from the system, modified by the survival probability p , and distributed by the dispersal matrix \mathbf{D} . If all of the patches cover all possible dispersal sites, then the rows of \mathbf{D} sum to one. Beyond that possible restriction, \mathbf{D} is general and can characterize a variety of dispersal mechanisms. The most commonly assumed dispersal mechanism is uniform dispersal, in which the production of larvae in each location redistributes uniformly over the entire system, often referred to as a common larval pool assumption.

The number that actually settle successfully follows the following stock-recruitment function:

$$s_j^{\text{out}} = \frac{s_j^{\text{in}}}{a^{-1} + c^{-1}s_j^{\text{in}}}. \quad (10)$$

This specification enables the model to simulate various density-dependent larval survival mechanisms in the system, all of which may be patch specific. Once the settlement is calculated for any given site, the successful settlers (s_j^{out}) become the next period's age zero entry and the growth process starts again. Appendix A contains baseline values of the parameters that we used to

²⁴In our case, $b = 2.68$ for sea urchins. The parameter b , in the absence of empirical work, is often assumed to be 3. The idea is that diameter (or length) is a one-dimensional measure and to get to mass, one goes through an approximation of volume first, and then mass is proportional to volume. The extent to which the cube of an organism's one-dimensional measure is a good approximation for its volume will vary by species, but it is typical for b to be greater than 1.

calibrate the spatial biological model. The parameters are based on ongoing fieldwork being conducted off the coast of northern California by colleagues investigating the sea urchin in a joint long-term research program.²⁵ Raw data on growth increments and adult size distributions have been gathered in dive transects at several exploited and unexploited sites along the coast. These have been used to compute growth and natural mortality coefficients using maximum likelihood techniques.

The dispersal process used in our model is a special feature and does not rely on the common larval pool assumption. Instead, we specify the dispersal matrix to reflect current understanding arising out of our joint work with biologists. That work has collected sea urchin larvae on a continuous basis and correlated larval settlement patterns with sea surface, wind, and current patterns to understand dispersal processes. The current thinking is that during the upwelling season from April to July, there is a strong offshore and equator-directed flow of current that sweeps larvae southward. The larvae then collect in “retention zones” or gyres to the south of two promontories of Point Reyes and Point Arena. Then, during relaxation of the upwelling events, a reversal of the process creates a nearshore pole-directed flow of currents that sweep and distribute larvae back along the coast. We thus find relatively more settlement in areas just north of the two gyres. The dispersal matrix used here reflects this realistic system that incorporates oceanographic influences on larval dispersal.

5.2. Calibrating the bioeconomic model

To simulate the implications of spatial closures, we combine the spatially explicit biological model with the model of spatial behavior estimated with the logbook/landings ticket database. The key link in the integrated model is the connection between monthly fishing mortality in each patch f_{jt} (in Eqs. (6) and (7)) and monthly trips, or

$$f_{jt} = (Trips_{jt})hq = \left(o_t \sum_{p=1}^4 d_p p_{pjt} \right) hq, \quad (11)$$

where h is diving hours per trip, q is the scaling or catchability coefficient (assumed constant across time), o_t is the number of choice occasions in that month,²⁶ d_p is the number of divers in port p , and p_{pjt} is the probability of a trip from port p to patch j in month t based on the RNL estimates. We presume that h is fixed across all ports and time periods, and we use q to calibrate model forecasts to actual harvests aggregated across space on an annual basis.²⁷ That is, we adjust the catchability coefficient so that the integrated model (1) appears to be on the same trajectory to a steady state as the actual data at the end of the sample period and (2) tracks the initial draw-down phase of the actual data. We place more weight on the first criterion than on the second because we are particularly interested in evaluating the steady-state implications for marine

²⁵ See [2–4,18,19,21,25,30] for further details on biological and oceanographic parameters.

²⁶ Choice occasions are determined by the season closure regulations, which are the same across all northern California ports.

²⁷ On average, harvesters spend 3.5 h diving per trip, and though there is considerable variation in dive hours, we have not found any variables that explain substantial amounts of that variation.

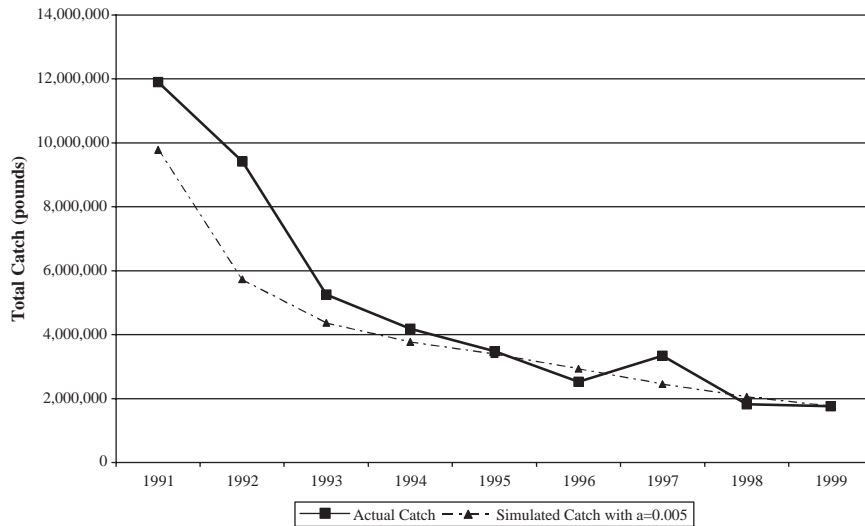


Fig. 2. Calibration of bioeconomic simulation model.

reserves.²⁸ In all simulations, since expected revenues are rolling 1 month backward averages, we use lagged catch per trip multiplied by an exogenous price to predict trips and shares of divers in each location. To simplify matters, we use a reduced-dimension model that includes only the four main northern California ports because fishing from the southern- and northern-most ports is erratic and accounts for less than 5% of the total fishing trips in the data set. Fig. 2 shows how the integrated model calibrates with the actual harvest path during the draw-down of the northern California urchin population over the past decade.

To understand how economic behavior affects forecasted impacts of marine reserves, we compare simulations of the integrated bioeconomic model (the ECON model) with a standard biology-only model that we dub the NOECON model. Both models use the dual gyre dispersal pattern that mirrors current thinking about coastal oceanography off northern California, and both incorporate the minimum size limit of 3.5 in. The NOECON model assumes, consistent with the biological literature, that total fishing effort before and after reserve formation is constant and is distributed uniformly across space. In contrast, the ECON model is based on our RNL behavioral model and incorporates seasonal and weekly closures by varying choice occasions over time.

Although the ECON model is the correct model, we need to calibrate the NOECON model also in order to evaluate the impacts of our behavioral model. We first calibrate NOECON to the draw-down phase of the actual system-wide harvest to mirror what the biological modeler would do. However, NOECON by nature of its behavioral simplification is unable to both track the draw-down phase and end up at the same steady state as ECON, which we believe to be the true steady state. Thus, we provide a second calibration of NOECON to the ECON steady state. This latter calibration allows us to explore the implications of ignoring spatial behavior without the confounding effect of inter-temporal effort dynamics.

²⁸ Attempts to use an “objective” calibration criterion, e.g. minimize sum of squared errors, tended to do well for early draw-down phase but poorly for the steady state trajectory.

Table 4

Marine reserves and economic behavior the northern California red sea urchin fishery

	Steady-state harvest (1000 pounds)	Steady-state egg production (billions)	Discounted ^a revenues (\$1000)
<i>With discrete choice behavioral model—ECON ($a = 0.005$)</i>			
No closure	830	1316	17,440
Close patch 8	752	1441	15,074
<i>With no economic model—NOECON ($a = 0.005$) approach path calibration</i>			
No closure	386 ^b	267	8096
Close patch 8	545	397	8204
<i>With no economic model—NOECON ($a = 0.005$) steady-state calibration</i>			
No closure	829 ^c	434	17,400
Close patch 8	868	553	16,423

^a Uses a 5% constant discount rate and assumes \$1 per pound of sea urchin.^b Calibrated approach path catch to actual catch.^c Calibrated steady-state harvest to behavioral model.

5.3. The importance of behavior

With the calibrated models we examine how a reserve placed in various patches might affect the whole fishery. Table 4 reports simulation results of closing patch 8, the heavily fished area off Fort Bragg.²⁹ This table demonstrates the importance of incorporating spatial behavior. First, the NOECON model dramatically overstates the extent of decline when calibrated to the draw-down phase of early fishery exploitation. The intuition for this is obvious once stated. Early in a fishery's development, abundance is high and revenues are high, drawing large amounts of effort into the fishery. This results in a draw-down phase of high fishing mortality, and if one assumes that this level of effort continues, that ultimately overpredicts the amount of effort, overstates the extent of overfishing, and thus underpredicts steady-state harvest. For example, calibrated to the draw-down phase, the NOECON model predicts a steady state harvest of only 386,000 pounds and a dramatically reduced egg production, reflecting the assumed low reproductive potential of the overexploited biomass. In contrast, the ECON model continuously adjusts effort to profitability. As the fishery is drawn down, effort exits and fishing mortality falls, generating steady-state biomass and harvest levels that are much larger. Under the ECON scenario, the steady-state harvest is more than double the NOECON prediction, at 830,000 pounds. Second, perhaps equally important, the predicted egg production in the ECON model is almost five times that

²⁹ We use patch 8 to demonstrate the impacts of closures mainly because the findings by Holland and Brazee [10] and Sanchirico and Wilen [24] suggest that reserves are likely to be beneficial to fisheries production in two instances: (a) either when the closed patch is heavily exploited; or (b) when the closed patch operates as a source rather than a sink. The gyre pattern and north-south pattern of larval flow makes patch 8 a type of source, although the definition is not always clear in the literature.

predicted with the NOECON model. The higher egg production emerges from a magnitude effect and a distribution effect. Since overall exploitation is lower, total reproductive biomass and egg production are larger. But the ECON model also predicts a spatial distribution of effort that depends upon relative spatial profits rather than assuming a uniform distribution. In this more realistic setting, areas that are high cost (e.g. more distant) will be lightly exploited and hence will serve as de facto “reserves”, even without explicit spatial closures.³⁰ In contrast, the NOECON model essentially assumes uniform overexploitation in these de facto reserves.

Table 4 also shows how misleading biological models without behavior may be about the impacts of reserve formation. For example, the NOECON model calibrated to the approach path predicts that a closure of patch 8 will produce a 40% increase in steady-state harvest, coupled with a 48% increase in system-wide egg production. The NOECON model assumes, as is typical in the literature, that displaced effort adjusts immediately and uniformly to the remaining open areas. Thus total harvest falls to zero in the closed area but increases initially in all open areas. In general, whether the present value of the loss over the whole adjustment period is made up by the corresponding gain depends on initial conditions and adjustment speeds. In the NOECON simulation, the model predicts that the initial loss is compensated for over the whole transition path since the overall present value of a spatial closure is positive. Contrast these results with the ECON case, which makes the transition path and ultimate steady state dependent upon relative profits. In the ECON simulation, a closure of patch 8 results in a 10% loss in steady-state harvests, in addition to the transition losses associated with the closure. The result is that discounted revenues fall by 14% compared with no spatial closure, rather than rising as predicted by the NOECON model. The importance of incorporating economic behavior into models intended to forecast the implications of reserves is thus profound. Importantly, the assumption of uniformly distributed and unresponsive effort used for simplicity in the biological literature biases predictions toward overly pessimistic status quo harvest and egg production predictions, and toward overly optimistic predictions of harvest gains and the net economic costs of reserve formation.

While so far we have compared our integrated model with what the biologists would do, it is interesting to isolate the impacts of ignoring spatial behavior without the confounding effect of dynamic changes in the overall level of fishing effort. Thus, the third section of Table 4 calibrates the pure biological model to the same steady-state harvest as ECON. In this case, the NOECON model still predicts a steady state harvest gain, although much smaller than under the approach path calibration. These harvest gains are not enough to compensate for initial harvest losses, and hence the system-wide present value of revenues falls. A more important difference may be in predicted egg production from the system. Because the NOECON model assumes uniform effort distribution, the whole system’s reproductive biomass is predicted to be drawn down relatively uniformly to establish the status quo. In contrast, the ECON model predicts a heterogeneous distribution of effort reflecting distance from ports and other costs of remote patches. These act as de facto sources, hence contributing to predictions of overall egg production that are larger than the NOECON model. Fig. 3 clarifies how this arises, depicting the steady-state size distribution

³⁰ The steady-state spatial coefficient of variation for NOECON is 0.28 whereas for ECON it is 0.43. We know that spatial variation in the NOECON model must be due to net larval dispersal differences. In the ECON model, spatial variation is also due to differential fishing effort responding to differential rents.

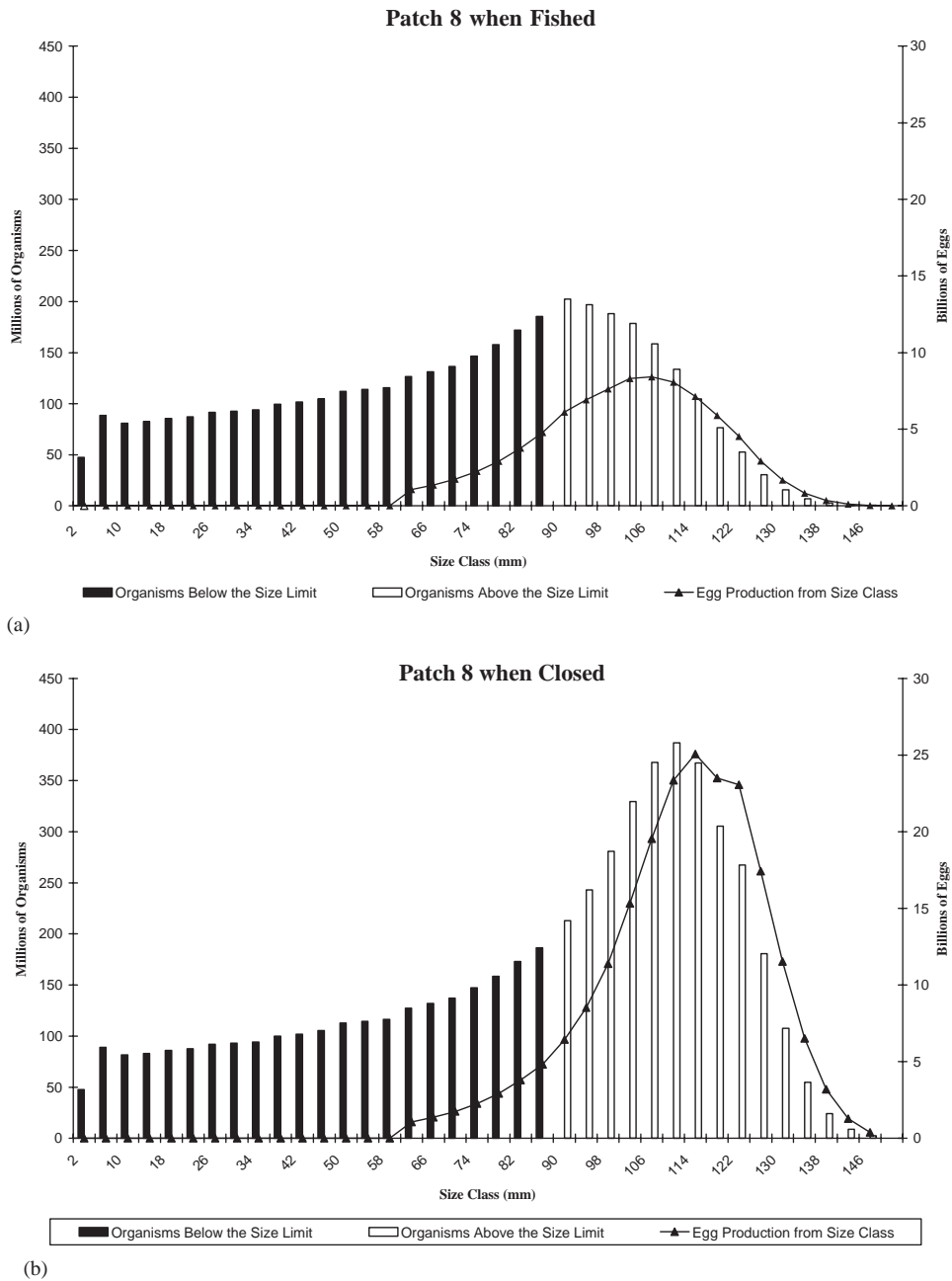


Fig. 3. Steady-state size distribution and egg production.

and egg production from a heavily exploited patch and from a closed patch. The closed patch has a wider size distribution with more large individuals that produce more total eggs. Egg production peaks beyond the size class with the most number of organisms both because egg production is

Table 5
Impacts of dispersal and spatial behavior

	System-wide totals for the NOECON model		System-wide totals for the ECON model	
	SS harvest (1000 pounds)	SS eggs (billions)	SS harvest (1000 pounds)	SS eggs (billions)
No closure	828.6	433.6	830.5	1316.0
<i>Separate patches</i>				
Close Farralons	670.6	566.5	819.1	1322.5
Close patch 1	749.4	580.3	827.1	1317.5
<i>First gyre</i>				
Close patch 2	807.5	584.1	755.2	1395.4
Close patch 3	812.4	571.3	765.3	1382.3
Close patch 4	816.3	559.3	761.2	1385.3
<i>Gyre border</i>				
Close patch 5	911.6	559.7	755.0	1425.6
<i>Second gyre</i>				
Close patch 6	868.3	594.7	743.6	1446.9
Close patch 7	869.1	572.6	746.7	1447.5
Close patch 8	868.0	553.3	752.4	1440.6
Close patch 9	862.3	522.2	787.3	1374.3
Close patch 10	839.5	470.8	829.6	1316.0

increasing in organism size and is convex in size.³¹ These depict how the de facto closures associated with relatively uneconomic patches can contribute in important ways to a system's egg production. Ignoring this possibility causes the pre-reserve status quo predictions to bias reproductive biomass predictions low.

Table 5 shows that the received wisdom about reserve siting based on oceanographic dispersal may not be robust when policy makers account for economic behavior. This table shows the implications of siting reserves in various different patches in order to achieve different objectives such as harvest gains and system reproductive capacity gains. We calibrate the NOECON model to the same steady-state harvest level as the ECON model but with a uniform effort distribution. Again, the ECON model predicts a drop in steady-state harvests under any option chosen, whereas the NOECON model predicts increases in over half of the patch closure options. The NOECON model predicts harvest decreases from closing the southern-most patches and patches in the southern gyre around Point Arena. This would lead one to favor other areas as candidates for closure. In contrast, in the ECON model, closing those southern-most patches is *least* costly in term of harvest loss because they are most lightly exploited before the reserve. In terms of total system-wide egg production, the gains predicted by the ECON model are not as large proportionately as those predicted with the NOECON model. The NOECON model predicts relatively large egg production gains in patches 1 and 2, primarily because the model overpredicts pre-reserve harvest pressure in those relatively unprofitable patches. The ECON model, however, predicts relatively large egg production gains from closing patch 2 because it is lightly exploited pre-reserve and a large contributor to the first gyre. Large egg production gains are also predicted

³¹ In viewing this figure, total egg production in each panel is the area under each egg production line.

from closing patches 6–8, but for different reasons. Patch 8 production gains come from closing a heavily exploited area, but patch 6 gains come from its critical role as a source feeding the second gyre. Overall, then, ranking the best sites depends upon both economic and biological dispersal. Ignoring economic dispersal profoundly affects what would appear to be wise siting choices, often missing the true configuration of the pre-reserve status quo, or incorrectly predicting eventual larval dispersal and harvest adjustments by failing to anticipate the behavioral response to spatial closures.

6. Summary and conclusions

This paper predicts some economic and biological impacts of marine reserve creation with particular attention to how economically motivated behavior determines outcomes. We address a potentially important shortcoming of the vast marine reserves literature, namely the assumption that fishing effort is fixed and uniformly distributed. Instead, we presume that effort in realistic settings responds to economic incentives, particularly differential profit opportunities that are dynamic and spatial. To address the importance of behavior, we construct and estimate a spatial choice model using a comprehensive database from logbooks and landings tickets. It is important to realize that fisheries scientists collect these data to guide regulatory decisions rather than to aid economic research *per se*. In spite of this, we show how this type of data can inform analyses of economic choices. The RNL econometric model confirms what we expect, namely that divers respond to differences in expected returns across different patches. The RNL model also shows that fishermen respond negatively to weather risk and travel distance and positively to expected returns, which are composed of both relative abundance and expected price. Although divers actually choose to participate only about 14% of the time, even our parsimonious model successfully explains a reasonable fraction of their variation in behavior.

We link the spatial model of choice behavior to a biological model intended to capture the most important features of our case study. The biological model is detailed and is the first to depict a multi-patch system with explicit empirically driven hypotheses about larval dispersal. Most existing literature uses two patch models; ours is an 11-patch system calibrated with parameters derived from field data. A unique feature of the biological model is its incorporation of the “dual gyre” nature of coastal circulation in northern California. This feature represents current thinking about dispersal processes that affect urchin larvae, and it adds detail and complexity that generate new hypotheses about reserve impacts and reserve siting questions. We simulate the model on a monthly time scale and the output from the model consists of aggregates such as system-wide egg production, total harvest, and present values of revenues. Only a handful of studies consider the role of larval dispersal analytically; this paper is among the first to study fishery policy with an empirically driven representation of dispersal.

Our results confirm that economic behavior is a critical determinant of the predicted impacts of reserves. Moreover, we show that the typical assumptions made by biologists for analytical tractability consistently bias the predicted impacts in a manner that makes reserves look more favorable than they actually might be. For example, a model that calibrates fishing mortality during a draw-down phase overpredicts the extent of overexploitation compared with an economically based model that incorporates the natural decline in profitability as the steady state is approached. Since aggregate harvest is more likely to be increased in overexploited fisheries,

results based on mis-calibrated fishing mortality optimistically favor reserve creation. However, we also show that even without draw-down phase mis-calibration, the assumptions of uniform effort rather than economically motivated effort produce mistaken characterizations of reserve siting in realistic settings. We show, for example, that some remote or high cost or high risk areas are naturally less profitable and hence exploited less by fishermen. These patches form *de facto* reserves and hence contribute to overall egg production in ways not revealed by simpler uniform effort distribution models. However, the manner in which they bolster both harvest and system reproduction is complicated because their role depends upon economic, biological, and oceanographic factors. And the remoteness of particular patches is relative and dependent upon fishery independent factors such as port location and roadways.

For reasons discussed above, our overall assessment of reserves as a fisheries policy tool is more ambivalent than the received wisdom in the biological literature. Although our model incorporates a rich depiction of biological and physical oceanographic processes, the conclusions are more in accord with the very small amount of conceptual economic analysis that has been devoted to the topic. We find, as did Holland and Brazee [10], that reserves can produce harvest gains in an age-structured model but only when the biomass is severely overexploited. We also find, as Holland and Brazee did, that even when steady state harvests are increased with a spatial closure, the discounted returns are often negative, reflecting slow biological recovery relative to the discount rate. We find results that confirm the Sanchirico and Wilen [23,24] analytical work that shows that easily exploited (low bioeconomic ratio) patches are most likely to be the best sites to produce both harvest and reproductive potential gains. Their modeling implicitly assumes that harvest gains accrue from emigrating adults responding to density buildups in the reserve patches. The model used in this paper depicts a different process in which larvae numbers depend upon spawning stock biomass, but larval spatial distribution is driven by oceanographic transport of larvae. Many marine biologists believe that larval transport mechanisms may be more important determinants of dispersal in marine spatial systems than adult migration. We show that some of the sink/source notions about site selection currently in vogue appear to be based on naïve views of spatial dynamics that fail to incorporate realistic detail about either biophysical processes or fisher behavior. In particular, whether a particular patch is a source or sink depends on its relative level of exploitation as well as its physical placement in an oceanographic system. Patches with high intrinsic productivity in an unexploited system or a lightly exploited system may be less productive to the system as a whole when differential harvesting pressure (driven by relative economic opportunities) affects the spatial distribution of abundance and hence larval production. These results suggest, in contrast to the tone of recent dialogue among marine scientists about reserves, that there are still unanswered questions about whether they can deliver what they promise based on the simple modeling done to date. At the very least, our integrated bioeconomic model raises new questions about whether oceanographic dispersal is the key driver of spatial closure impacts, or whether harvester dispersal may be equally important.

Acknowledgments

The authors thank Loo Botsford, Dan Holland, Doug Larson, David Layton, Bill Provencher, Jeffrey Williams and two anonymous referees for helpful comments and suggestions and thank

Table 6

Parameter	Description	Value
k	Growth	0.24
m	Natural mortality	0.09
L_{inf}	Terminal size (mm)	118
L_{limit}	Min. size limit (mm)	89
L_{mature}	Min. size of sexually mature organism	60
f	Fishing mortality	0.29
w	1st allometric weighting parm.	0.001413
b	2nd allometric weighting parm.	2.68
α	1st egg production parm.	5.47E-06
β	2nd eggs production parm.	3.45
p	Survival probability	1.0
a	Resiliency settlement parm.	0.005–0.05
c	Carrying capacity settlement parm.	1.2E + 07–2.4E + 07

Dale Lockwood for programming assistance. This research is funded in part by a grant from the National Sea Grant College Program, National Oceanic and Atmospheric Administration, US Department of Commerce, under Grant NA06RG0142 project number R/F-179 through the California Sea Grant College System, and in part by the California State Resources Agency. The views expressed herein are those of the authors and do not necessarily reflect the views of NOAA or any of its sub-agencies, or the Resources Agency.

Appendix A

Baseline parameter values for bioeconomic simulations are given in [Table 6](#).

References

- [1] R.J.H. Beverton, S.J. Holt, On the Dynamics of Exploited Fish Populations, Chapman & Hall, London (1957, reprinted in 1993).
- [2] L.W. Botsford, D. Lockwood, L. Morgan, J.E. Wilen, Marine reserves and management of the northern California red sea urchin fishery, CALCOFI Report No. 40, 1999, pp. 87–93.
- [3] L.W. Botsford, J.F. Quinn, S.R. Wing, J.G. Brittnacher, Rotating spatial harvest of a benthic invertebrate, the red sea urchin, *Strongylocentrotus franciscanus*, in: Proceedings of the International Symposium on Management Strategies for Exploited Fish Populations, University of Alaska Sea Grant College Program, 1993.
- [4] L.W. Botsford, B. Smith, J.F. Quinn, Bimodality in size distributions: the red sea urchin *Strongylocentrotus franciscanus* as an example, *Ecolog. Appl.* 4 (1) (1994) 42–50.
- [5] G.M. Brown, J. Roughgarden, A metapopulation model with private property and a common pool, *Ecolog. Econom.* 22 (1997) 65–71.
- [6] A. Daly, S. Zachary, Improved multiple choice models, in: D. Henscher, Q. Dalvi (Eds.), Identifying and Measuring the Determinants of Mode Choice, Teakfield, London, 1979, pp. 335–357.
- [7] B. Halpern, The impact of marine reserves: does size matter?, *Ecolog. Appl.* (2002), forthcoming.
- [8] R. Hannesson, Marine reserves: what should they accomplish?, *Marine Resource Econom.* 13 (1998) 159–170.

- [9] A. Hastings, L.W. Botsford, Equivalence in yield from marine reserves and traditional fisheries management, *Science* 284 (1999) 1537–1538.
- [10] D.S. Holland, R.J. Brazee, Marine reserves for fisheries management, *Marine Resource Econom.* 11 (1996) 157–171.
- [11] P.E. Kalvass, J.M. Hendrix, The California red sea urchin, *Strongylocentrotus franciscanus*, fishery: catch, effort, and management trends, *Marine Fisheries Rev.* 59 (2) (1997) 1–17.
- [12] S. Kato, S.C. Schroeter, Biology of the red sea urchin, *Marine Fisheries Rev.* 47 (3) (1985) 1–20.
- [13] D. Ludwig, R. Hilborn, C. Walters, Uncertainty, resource exploitation, and conservation: lessons from history, *Science* 260 (1993) 17–18.
- [14] D. McFadden, Modeling the choice of residential location, in: A. Karlqvist, L. Lundqvist, F. Snickbars, J.W. Weibull (Eds.), *Spatial Interaction Theory and Planning Models*, North-Holland, Amsterdam, 1978, pp. 75–96.
- [15] D. McFadden, Quantitative methods for analyzing travel behavior of individuals: some recent developments, in: D. Hensher, P. Stopher (Eds.), *Behavioral Travel Modeling*, Croom Helm, London, 1979, pp. 279–318.
- [16] D. McFadden, K.E. Train, Mixed MNL models for discrete response, *J. Appl. Econometrics* 15 (5) (2000) 447–470.
- [17] E.R. Morey, R.D. Rowe, M. Watson, A repeated nested-logit model of Atlantic salmon fishing, *Amer. J. Agric. Econom.* 75 (1993) 578–592.
- [18] L. Morgan, Spatial variability in growth, mortality and recruitment in the Northern California Red Sea Urchin Fishery, Ph.D. Dissertation, University of California, Davis, 1997.
- [19] L. Morgan, S.R. Wing, L.W. Botsford, C. Lundquist, J. Diehl, Spatial variability in red sea urchin recruitment in northern California, *Fisheries Oceanogr.* 9 (1) (2000) 83–98.
- [20] T. Polacheck, Year around closed areas as a management tool, *Nat. Resource Modeling* 4 (2) (1990) 327–353.
- [21] J.F. Quinn, S.R. Wing, L.W. Botsford, Harvest refugia in marine invertebrate fisheries: models and applications to the red sea urchin, *Strongylocentrotus franciscanus*, *Amer. Zoologist* 33 (1993) 537–550.
- [22] J.A. Reynolds, J.E. Wilen, The sea urchin fishery: harvesting, processing, and the market, *Marine Resource Econom.* 15 (2) (2000) 115–126.
- [23] J.N. Sanchirico, J.E. Wilen, Bioeconomics of spatial exploitation in a patchy environment, *J. Environ. Econom. Management* 37 (1999) 129–150 doi:10.1006/jeem.1998.1060.
- [24] J.N. Sanchirico, J.E. Wilen, Bioeconomics of marine reserve creation, *J. Environ. Econom. Management* 42 (2001) 257–276 doi:10.1006/jeem.2000.1162.
- [25] B. Smith, L.W. Botsford, S.R. Wing, Estimation of growth and mortality parameters from size frequency distributions lacking age patterns: the red sea urchin (*Strongylocentrotus franciscanus*) as an example, *Canad. J. Fisheries Aquatic Sci.* 55 (1998) 1236–1247.
- [26] M.D. Smith, Spatial search and fishing location choice: methodological challenges of empirical modeling, *Amer. J. Agric. Econom.* 82 (5) (2000) 1198–1206.
- [27] M.D. Smith, Spatial behavior, marine reserves, and the Northern California Sea Urchin Fishery, Ph.D. Dissertation, University of California, Davis, 2001.
- [28] M.D. Smith, Two econometric approaches for predicting the spatial behavior of renewable resource harvesters, *Land Econom.* 78 (4) (2002) 522–538.
- [29] C. Walters, *Adaptive Management of Renewable Resources*, Macmillan Publishing Company, New York (1986, reprinted by Fisheries Centre, University of British Columbia, 1997).
- [30] J.E. Wilen, M.D. Smith, D. Lockwood, L.W. Botsford, Avoiding surprises: incorporating fishermen behavior into management models, *Bull. Marine Sci.* 70 (2) (2002) 553–575.