ARE MARINE PROTECTED AREAS IN THE TURKS AND CAICOS ISLANDS ECOLOGICALLY OR ECONOMICALLY VALUABLE?[1]

Murray A Rudd[2], Andy J. Danylchuk , Shannon A. Gore and Mark H. Tupper[3]

[2] Center for Marine Resource Studies (Turks & Caicos Islands), School for Field Studies, 16 Broadway, Beverly. Email: mrudd@sfs-tci.org
[3] Current address, Florida Marine Research Institute, St. Petersburg, Florida

Abstract
Marine Protected Areas (MPAs) are often advocated by ecologists as a method of conserving valuable fish stocks while ensuring the integrity of ecological processes in the face of increasing anthropogenic disturbance. In the Turks and Caicos Islands there is little evidence that current MPAs are ecologically beneficial but there are indications that boundary changes may enhance queen conch and finfish production. Implementing boundary changes usually requires political will and, hence, quantifiable economic benefits. Assessing the value of reef fish is particularly important because they are potentially valuable for consumptive and non-consumptive purposes. We demonstrate the non-consumptive economic value of increased Nassau grouper size and abundance to the dive tourism industry through a paired comparison conjoint survey of visiting divers. Our results suggest that accounting for the non-consumptive economic value of increased Nassau grouper abundance and size may have a large impact on the economic viability of ecologically functional MPAs.

Keywords: *MPA economics; Nassau grouper; conjoint analysis; nonmarket valuation; paired comparison*

Introduction
The inshore marine environment provides humans with a wide variety of ecological and economic services (Moberg and Folke, 1999) and is especially important in tropical developing countries where economic opportunities are limited. Many demersal fisheries operate at or beyond their sustainable limits (National Marine Fisheries Service, 1997; National Research Council, 1999) and the demand for fish continues to grow. The management of a marine fishery is a difficult task (Botsford et al., 1997; Costanza et al., 1998) and in the tropics, where ecologically complex ecosystems are under heavy pressure from rapidly increasing anthropogenic stress and are typically managed by institutionally weak governments, the problem is exacerbated (Roberts and Polunin, 1993; Roberts, 1997; Johannes, 1998; Mascia, 2000).

Traditional fisheries management has focused on the optimal exploitation of individual stocks of commercially important species despite the fact that most demersal fisheries involve multiple species. Regulatory approaches aim to control either fishing mortality and/or effort by means of quotas, gear restrictions, size limits, vessel permits, and/or seasonal closure (King, 1995). These strategies often have high transaction costs (North, 1990) and are thus ineffective in many cases (Roberts, 1997).

In the tropics, the management of the inshore environment has proved problematic due to the complexity of the dynamic coral reef – seagrass – mangrove ecosystem and confounding anthropogenic pressure (Johannes, 1998). Many tropical species are particularly vulnerable to overexploitation due to the wide variety of fishing methods used in artisanal commercial reef fisheries (Munro and Williams, 1985). Grouper, for example, comprise about 10% of the total coral reef finfish yield worldwide and is amongst the most endangered family. In 1996, 21 species of grouper were proposed for the IUCN 'Red List'; of these three species are critically endangered (Hudson and Mace, 1996).

In recent years, marine protected areas (MPAs) have received much attention as an alternative approach to traditional fisheries management (Plan Development Team, 1990; Roberts, 1997; Murray et al., 1999). Ecologically, MPAs are thought to be able to simultaneously address problems that traditional management cannot. The primary goals of MPAs are to protect critical habitat and biodiversity, and to sustain or enhance fisheries by preventing spawning stock collapse and providing recruitment to fished areas (Medley et al., 1993; Johnson et al., 1999; Murray et al., 1999).

From an economic perspective, the use of MPAs offers several theoretical advantages over the traditional management measures. MPAs, like terrestrial protected areas, may provide substantial non-consumptive economic use value by providing opportunities for recreation, education, scientific research (Dixon, 1993; Ruitenbeek, 1999) and indirect use value by

[1] We would like to thank Sarah Carew, Erin Davies, Jen Shapira and Gail Weaver for assistance designing and testing the pilot survey. Thanks are also due to the TCI dive operators who helped distribute the surveys and gave freely of their time and expertise and to Darden Environmental Trust, who provided funding for the conjoint software.

increasing ecological resilience (Perrings et al., 1995). Protected areas may also provide non-use values for humans who value the existence of protected environments and species even though they have no plans to personally visit or use them. In addition to providing value to humans through the provision of valuable ecological goods and services, MPAs have other potential advantages. Transaction costs (North, 1990) associated with information collection, contracting, monitoring and enforcement are potentially lower when using MPAs compared to other information-intensive fishery management tools (Roberts and Polunin, 1993; Agardy, 1994; Costanza et al., 1998; Johannes, 1998; Mascia, 2000).

Unless the ecological impacts of MPAs are demonstrated and linked directly with economic value, it is unlikely that decision-makers within government will consider MPAs as viable policy tools for managing coastal resources. In order to demonstrate the utility of MPAs, it is imperative that the potential economic benefits of marine conservation be clarified. A complete accounting of the benefits of conservation may actually help tip the balance of a cost-benefit analysis in favor of the conservation option.

In the tropics, the tourism industry is an important part of many economies (ARA Consulting et al., 1996). Increasing demand for nature-based tourism ensures that protected areas (Gössling, 1999) and a healthy environment (Huybers and Bennett, 2000) are important production inputs for the tourism industry. Viewing wildlife is recognized as providing economic value to participants and is a basis for both terrestrial (Gössling, 1999) and marine (*e.g.*, Loomis and Larson, 1994; Davis and Tisdell, 1998) tourism industries.

The objective of our study is to assess the existence of non-consumptive economic benefits of MPAs in the Turks and Caicos Islands (TCI), British West Indies. We hypothesize that viewing healthy coral reefs and vibrant fish communities adds value to the experience of visiting tourists. Our research specifically examines the value of Nassau grouper (*Epinephelus striatus*) through a paired comparison conjoint survey. It assesses the added value that increased grouper size and abundance contributes to the dive experience for visiting divers to the TCI. In the balance of this paper, we outline the potential ecological benefits of MPAs in South Caicos, TCI. Next we outline a framework for assessing the potential economic value of these ecological benefits of MPAs. We then review the paired comparison survey methodology and results, and conclude with a discussion on the implications of the results for the TCI dive industry and government policy makers.

The Turks and Caicos Islands

The Turks and Caicos Islands (TCI) are located at the southern end of the Bahamian archipelago and are relatively pristine compared to other countries in the Greater Caribbean basin. Commercial fishing for spiny lobster (*Panulirus argus*) and queen conch (*Strombus gigas*) has been a mainstay of the local economy for decades but tourism emerged as the number one industry in the country by the early 1990s (Turks and Caicos Islands Government, 1996). While the pristine reefs are a prime attraction for tourists, increasing tourism is putting pressure on the nearshore coral reef environment. Growing tourism combined with tariff protection for the TCI fishing industry has led to an expansion in the market demand for local seafood and resulted in increasing fishing pressure on potentially vulnerable stocks of conch, lobster and carnivorous reef fish such as groupers, snappers and grunts (Christian-Smith and Darian, 2000).

A National Parks Order formally established 33 terrestrial and marine protected areas in 1992 (Homer, 2000). Government management of MPAs has been in a state of flux over the last ten years and there is currently a restructuring in process that will soon cleave responsibility for park operation away from the Department of Environmental and Coastal Resources (DECR) to a new National Park Service (NPS).

There are four designated MPAs on South Caicos: Admiral Cockburn Land and Sea National Park (ACLSNP); East Harbor Lobster and Conch Reserve (EHLCR); Admiral Cockburn Nature Reserve; and Bell Sound Nature Reserve. ACLSNP and EHLCR are adjacent (Figure 1) and effectively encompass the Admiral Cockburn Nature Reserve. The Bell Sound Reserve is on the north side of the island and was implemented to protect bonefish habitat and stocks.

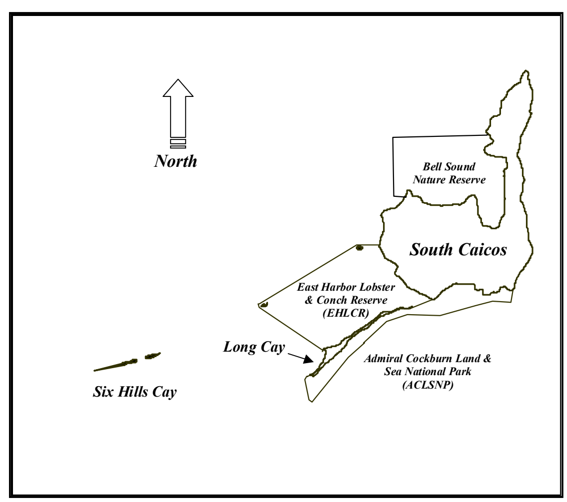

Figure 1: Marine Protected Areas of South Caicos, Turks and Caicos Islands

The Ecological Value of South Caicos MPA

Spiny Lobster

Stocks of spiny lobsters are heavily exploited around the world due to their high market value. From Bermuda to southern Brazil, *Panulirus argus* is one of the most heavily fished and commercially significant shellfish throughout its range (Butler and Herrnkind, 1997) and are the principal commercial fishery resource in the TCI. The fishery is managed by the DECR and management regulations include a closed season, an 83-mm minimum carapace length, a prohibition on the harvest of berried or tarred females, and a prohibition on the use of SCUBA or chemicals such as detergent or bleach. In addition, lobster fishing is prohibited in the South Caicos MPAs.

Spiny lobster larvae are pelagic and individuals may remain in the water column from 6- to 12-months and are therefore subject to dispersal over a range of hundreds or even thousands of kilometers (Lipcius and Cobb, 1994). This suggests that fluctuations in ocean currents may alter annual patterns of larval movement and make it difficult to causally link the effects of increased larval production within MPAs to fisheries benefits in other areas.

Some evidence does show that MPAs protect juvenile lobsters until they mature and move out of the reserves into surrounding areas (Davis and Dodrill, 1985; MacDiarmid and Breen, 1993). However, empirical evidence demonstrating that populations of exploited species may recover is limited (Kelly et al., 2000a) and documented benefits of these reserves to the conservation and management of lobsters are sparse (Childress, 1997).

There is little fishing activity in the vicinity of the South Caicos MPAs while more distant fishing grounds outside the area are heavily exploited. Unpublished data collected by the Center for Marine Resource Studies (CMRS) shows the average length of lobster caught in several fishing grounds around South Caicos (Table 1). The average carapace length of lobsters from the shallow, sheltered and accessible areas such as Six Hills (8-km from the fishing port, Figure 1) is often under the 83-mm legal minimum. Remote areas that are deeper, less accessible and exposed to adverse sea conditions – such as East Side, Bush Cay and White Cay (up to 40-km from the harbor) – have much higher average lengths.

Table 1: Mean Length (mm) of Lobsters Landed in Fishing Regions near South Caicos

	Spiny Lobster Average Carapace Length (mm)			
Sampling Period	**East Side**	**Six Hills**	**Bush Cay**	**White Cay**
Fall 1993	101.2	88.0	99.1	101.0
Spring 1994	114.4	77.3	106.6	101.5
Summer 1994	102.1	78.8	105.7	111.0
Summer 1998	100.7	80.8	97.5	97.5
Fall 1998	103.0	83.0	101.0	101.0
Average	**104.3**	**81.6**	**102.0**	**102.4**

Legal minimum carapace length = 83 mm

Many of the undersize lobster are landed early in the season during the phenomenon known locally as “The Big Grab” (Olguin et al., 1998). Any TCI citizen is entitled to a fishing license for a nominal fee and during the month of August many part-time fishers take leave from other jobs and travel to South Caicos for the lucrative opening of lobster season (August landings have accounted for 25% to 40% of total annual landings since 1989). These part-time fishers are not usually skilled free divers and their technical skill limits many of them to fishing shallower areas where lobsters are smaller.

The catch trends from various areas, anecdotal information from fishers (Moran, 1992) and biological evidence that *P. argus* commonly undertake long-distance migrations (Herrnkind, 1980) suggest that lobsters are migrating from shallow to deeper waters as they grow older and larger. It appears, however, that they are being intercepted prior to reaching the refuge of deeper water. An effective strategy for lobster conservation in the TCI would involve a large protected area (Childress, 1997) in the core of the Caicos Bank to protect juveniles and their migration routes from shallow to deep habitats. However, it is unlikely that such a plan could be implemented, as it would be even more difficult to enforce remote MPA boundaries than size limits. We see more effective enforcement of current size limits as the solution to lobster overfishing; this would allow substantial numbers of adults to reach the refuge of deep waters and grow to reproductive size.

Queen Conch

The queen conch is a large marine gastropod found throughout most of the Caribbean, southern Florida, and Bermuda (Brownell and Stevely, 1981). Because queen conch are slow moving and inhabit shallow water, they are relatively easy to collect and have thus been a staple food item throughout the Caribbean for hundreds of years (Brownell and Stevely, 1981; Stager and Chen, 1996). Fueled by increases in demand, commercial fisheries developed in many regions during the early- to mid-1900's (Ninnes, 1994). Consequently, fishing pressure on queen conch stocks quickly became intense, bringing some stocks, such as those off of Bermuda, to the point of commercial extinction (Brownell and Stevely, 1981; Mulliken 1996).

In the TCI, commercial landings of queen conch have been recorded since 1904 (Ninnes, 1994). Although landings have been quite variable throughout the years, the size of landed conch has decreased over time and the need to harvest in more distant, deeper waters has increased, suggesting that stocks may be declining (Ninnes, 1994). To address concerns regarding the possible decline of queen conch stocks in the TCI, fisheries legislation was enacted to help regulate harvest rates (Mulliken, 1996). To date, harvesting regulations include size limits, equipment restrictions, licensing and export quotas, and a seasonal closure of the fishery. In addition to these traditional fisheries management techniques, an MPA, EHLCR, was established off of South Caicos in 1993.

As with any marine reserve, one of the anticipated benefits of the MPA was an increase in abundance of queen conch within the reserve (Bohnsack, 1993; Murray et al., 1999). A recent study demonstrated that five years after its establishment, the density of queen conch within EHCLR was nearly double that in similar habitat outside the reserve (Tewfik and Bene, in press). The study also found that adults dominated the age/size structure of the queen conch population within the MPA, whereas juveniles dominated populations outside the reserve. This result is no doubt due to the limited harvesting within the reserve, which enables individuals to fully grow to the adult life stage (Stoner and Ray, 1996).

Given the increase in density of queen conch inside the MPA, it is reasonable to hypothesize that spillover from the reserve might enhance local fishing yields (Bohnsack, 1993; Murray et al., 1999). However, preliminary results of spatial surveys conducted within and adjacent to the MPA suggest that the current reserve boundaries do not promote the spillover of adults into fished areas (Danylchuk, unpublished data). Extremely shallow sandbars run adjacent to two borders of the reserve and queen conch are rarely found in these areas. Since queen conch prefer moderately dense seagrass beds that provide food and shelter (Stoner and Waite, 1990), their near absence on these shallow, sandy areas is not surprising. Moreover, as queen conch grow they tend to expand their home ranges to include deeper waters (Hesse, 1979; Stoner and Ray, 1996), further reducing the likelihood of adults dispersing over shallow sandbars. Although deeper water does occur near the remaining border of EHLCR, this section of the reserve abuts ACLSNP and the offshore boundary rapidly exceeds the depth range for queen conch (Stoner and Sandt, 1992).

Although the potential for spillover of adults from EHLCR into locally fished areas is limited, spawning activity within the reserve may help support local queen conch populations. Studies in the Bahamas indicate that the distribution of queen conch is directly related to larval supply, and high densities of late-stage larvae tend to be found in areas where stable aggregations of juvenile queen conch also occur (Stoner and Davis, 1997). Given that conch larvae have a relatively short planktonic stage and local oceanographic conditions can retain larvae between 10- and 100-km from where they were hatched (reviewed in Appledoorn, 1997), it is likely that larvae originating in the MPA supply other core conch nurseries downstream of the reserve.

To realize the full potential of the EHLCR as a fishing reserve, it may be necessary to revisit the siting of the reserve boundaries. At present, the spillover of adults to fished areas is impeded by the lack of contiguous, suitable habitat, limiting the contribution of the MPA to the local fishery. As such, it may be worth examining whether the size of the reserve could be reduced to enhance the local harvest. However, further research is needed to determine whether changing the location of the boundaries will have any substantial effects on the breeding population and subsequent larval supply to fished areas downstream of the reserve.

Reef Fishes

Since fisheries are size-selective, the establishment of MPAs is expected to increase both the average size and abundance of exploited species. In general, MPAs have proven effective in this capacity, particularly with regard to large carnivorous species such as groupers (Serranidae) and snappers (Lutjanidae) which are long-lived, slow-growing fishes with delayed reproduction (Polunin and Roberts, 1993; Rakitin and Kramer, 1996; Sluka et al., 1996, 1998; Johnson et al., 1999; Tupper and Juanes, 1999). These families are among the most important both commercially and recreationally throughout subtropical and tropical waters (Sluka et al., 1996; Sluka and Sullivan, 1998; Beets and Friedlander, 1999). Since fecundity of fishes increases exponentially with length (Wootton, 1990), an increase in both average size and abundance of fish within MPAs should lead to substantially greater fish production than in adjacent fished areas (Plan Development Team, 1990; Roberts and Polunin, 1991). Theoretically, small MPAs could produce as many eggs as much larger fished areas (Man et al., 1995). Whether or not this production will enhance local fisheries depends on local oceanographic processes that transport larvae from protected spawning areas (Tupper and Juanes, 1999).

In order to be effective, MPAs must be planned with the ecology of target species in mind (Murray et al., 1999). In particular, MPAs must encompass the habitats used by a target species and must also encompass most of the species' home range (Kramer and Chapman, 1999). The more time fish spend outside the MPA, the higher their risk of fishing mortality. Thus, MPAs are rarely effective in protecting highly migratory species. Some groupers, such as the gag, *Mycteroperca microlepis*, and the Nassau grouper, *Epinephelus striatus*, may migrate several hundred kilometers to spawning aggregation sites, during which time they are vulnerable to fishing (Tupper, 1999). These species may spend significant amounts of time outside of small MPAs, reducing the benefits of protection.

Abundance and size of several commercially important reef fishes, including Nassau grouper, were measured within several habitat types in fished reefs and in a small MPA around South Caicos (Tupper, unpublished data). There was no difference between fished and unfished areas in mean size or density of any grouper species. It was determined that Nassau grouper preferred windward Pleistocene reef formations along the edge of the drop-off, at a depth of approximately 20-m. These high-relief formations, which provide abundant caves and crevices in which groupers could shelter and ambush prey, were not present within the MPA. This suggests that habitat preference may be more important than fishing pressure in determining the distribution and abundance of Nassau grouper. In contrast, Sluka et al. (1996) found no significant relationship between habitat variables and the abundance of Nassau grouper. They also found that Nassau grouper, in addition to several other grouper species, were more abundant and larger within the Exumas Cays Land & Sea National Park (central Bahamas) than on adjacent unprotected reefs. However, The Exuma Cays do not possess the same type of high-relief windward reefs nor the steep drop-off that typifies the TCI shelf edge (M. Tupper, personal observation). Several studies have shown an association between groupers and high-relief habitats (Nagelkerken, 1981; Sluka, 1995; Sluka et al., 1996, 1998). It is likely that in the absence of preferred habitat types, fishing pressure would become the major factor influencing distribution and abundance of exploited reef fishes.

Another possible explanation for the differing results of protection in the Exuma Cays and TCI is that fishing pressure on Nassau grouper and other reef fishes is probably higher in the Exuma Cays. In the TCI, the fishery is directed mainly at queen conch and spiny lobster; finfish are most often taken opportunistically as bycatch of the lobster fishery (Kassakian, 1999). In order to see any effects of establishing an MPA, the area must be subject to a lower level of fishing pressure than surrounding areas. It is possible that the level of fishing pressure around South Caicos is too low to cause a detectable difference in fish size and abundance between the MPA and surrounding areas.

Table 2: Density (number of individuals per 100 m^2) of grouper species in protected and fished areas of the wider Caribbean region[1].

[1] *Sample areas are arranged from heavily fished (left) to no fishing (right). BMR = Barbados Marine Reserve; ECLSP = Exuma Cays Land & Sea Park; ACLSNP = Admiral Cockburn Land & Sea National Park. Table modified from Chiappone et al. (2000). Barbados data from Chapman and Kramer (1999). South Caicos data from Tupper (unpublished).*

Grouper Species	Barbados	SE Cuba	Dominican Republic	Florida Keys	S Exumas	N Exumas	S Caicos	BMR	ECLSP	ACLSNP
Cephalopholis cruentatus	0.08	2.30	0.95	0.97	0.16	0.60	0.16	0.16	0.27	0.15
C. fulva	0.16	0.63	0.35	0.01	1.30	0.44	1.86	0.24	0.52	1.78
Epinephelus adcensionis			0.04	0.04	0.01	0.01	0.06		0.04	0.08
E. guttatus			0.08	0.04	0.13	0.20	0.26		0.14	0.20
E. itajara							0.02			
E. striatus				0.01	0.16	0.20	0.62		0.35	0.48
Mycteroperca bonaci				0.04		0.01			0.01	0.01
M. tigris				0.02	0.01	0.06	0.14		0.12	0.16
M. venenosa					0.02	0.01	0.08		0.05	0.05

Indeed, Table 2 shows that the density of all species measured at South Caicos rivaled or exceeded densities reported for these species within MPAs elsewhere in the Caribbean (Chapman and Kramer, 1999; Chiappone et al., 2000). Thus, the ACLSNP may not be effective in protecting reef fish stocks around South Caicos simply because they are not currently in need of protection. However, the fishery situation in the Turks & Caicos Islands is rapidly changing. Increasing demand for finfish from the hotel and restaurant industries has rapidly increased the demand for grouper and snapper on the more developed islands of Grand Turk and Providenciales (Christian-Smith and Darian, 2000). The South Caicos MPA may therefore prove useful in the future, particularly if its boundaries are extended to include the preferred habitat of Nassau grouper.

Potential Economic Value of TCI MPAs

Economic Theory

To implement MPAs in developing countries will usually require more than simply providing evidence of their ecological value. Ecological arguments may hold little weight if there are not economic benefits associated with MPAs. If the economic benefits of marine conservation can be tied to the health of economically important industries (*i.e.*, tourism), then MPAs may be considered a realistic management option. A critical step in MPA implementation is to demonstrate the linkages between ecological health and economic opportunity.

Public environmental goods such as environmental quality are those that have no market impacts and are therefore impossible to value using standard economic techniques (Boyle and Bishop, 1987). Concentrating on direct use value (*i.e.*, the value of enhanced fishery production for consumptive uses) and ignoring nonmarket values can lead to the underestimation of the economic benefits of conservation, a bias in the decision-making process, reduced social welfare and a misallocation of societal resources (Randall, 1993). A total economic value (TEV) framework based on Dixon and Pagiola (1998) is useful for considering the potential economic value of the diverse ecological services that MPAs are thought to provide. The main focus in valuation will vary for different MPA ecological services; the primary types of economic value that the various services are likely to provide are highlighted in Figure 2.

A number of methodologies have been used for assessing the value of ecological goods and services. These include Travel Cost Methodology (TCM) (Fletcher et al., 1990), Contingent Valuation Methodology (CVM) (Hannemann, 1984) and, most recently, a number of techniques adapted from marketing research and broadly known as conjoint analysis (Johnson et al., 1995; Roe et al., 1996; Hanley et al., 1998; Farber and Griner, 2000). These methodologies can be divided into two general categories: those that rely on revealed preferences and those that rely on stated preferences. TCM is a revealed preference method because it uses real expenditures that people make on recreational travel to statistically analyze willingness to pay (WTP) for environmental quality. CVM and conjoint surveys, on the other hand, are methodologies that rely on stated preferences. In these surveys, people are queried about their preferences in surveys that present hypothetical market situations. These surveys allow the derivation of values for goods and services that would otherwise have to be excluded from cost-benefit analyses.

	Total Economic Value					
	Use Value				Non-Use Value	
	Direct		Indirect	Option	Existence	Bequest
Potential Ecological Benefits	Consumptive	Non-Consumptive				
Increased Abundance		☑				
Increased Size		☑				
Spillover to Fishing Grounds	☑					
Larval Export	☑					
Increased Ecosystem Resilience			☑	☑		
Maintain Biodiversity					☑	☑

Figure 2: Total Economic Value Framework and Areas of Focus for MPA Valuation

There has been an increasing amount of environmental research using conjoint methodologies developed in other fields of economics (Louviere, 1988; Carson et al., 1994). These include choice experiments (Roe et al., 1996; Hanley et al., 1998; Farber and Griner, 2000; Huybers and Bennett, 2000) and paired comparison conjoint analysis (Johnson et al., 1995; Johnson et al., 1998). The strength of conjoint approaches derives from the use of nearly orthogonal survey designs that statistically isolate the effects of individual attributes on choice (Hanley et al., 1998). In a paired comparison conjoint analysis, survey respondents are presented with a choice of product profiles, each of which consists of a number of attributes. Each attribute can take on a number of specific levels that influence the value consumers perceive for the product. Whereas CVM asks respondents whether they are willing to pay a fee to improve environmental quality, a paired-comparison conjoint survey asks respondents to provide a rated comparison of two different profiles. We ask: what is your preference, on a scale of 1 to 9, of the baseline profile compared to an alternative profile?

While beyond the scope of the current paper, it is possible to derive measures of consumer welfare that are comparable with those derived using CVM using re-scaling of the ratings, a probit model and maximum-likelihood estimation techniques (Johnson et al., 1998). For current purposes, we concentrate on demonstrating the added value that increased environmental quality can provide to divers. This can be done assuming a simple linear utility function and using ordinary least squares (OLS) to estimate regression coefficients that can then be used in market simulations. While theoretically correct estimates of consumer welfare are important for comprehensive cost-benefit analyses and possibly for setting park entrance fees, the demonstration of the economic potential of MPA protection is in itself a strong educational tool for government decision-makers in developing countries.

Survey Methodology: Diver Preferences for Nassau Grouper

The focal species for our study was Nassau grouper because it is thought to have both consumptive and non-consumptive use values for the restaurant and dive industries, respectively. The goals of this survey are to: (1) identify key environmental attributes that add value to the experience of TCI dive tourists; and (2) assess the price sensitivity of divers to changing levels of these key attributes.

Key attributes were identified through interviews with dive tour operators, and experienced sport and professional divers in the TCI. Appropriate levels were chosen based on expert judgement, survey pre-testing and a pilot survey. Revisions to the pilot survey were incorporated to a second draft survey and further feedback obtained from survey respondents and dive operators. The final survey instrument used a total of five attributes and a total of twenty levels: size of dive group (3-7, 8-14, 15-23, and 24-30); presence of other species (1 or more lobster, 1 or more sea turtles, 1 or more reef shark, and none of the above); Nassau grouper abundance (1, 3, 6, or 12 fish per dive); Nassau grouper average size (small 2.27-kg, medium 6.80-kg, and large 13.61 kg); and dive price ($40, $41, $45, $50 or $60 per 20-minute single tank dive).

The four sizes of dive group span the most common range for dive charters in the TCI. We found through initial interviews that seeing “big stuff” was very important to divers and that sharks, turtles, dolphins, eagle rays and whales are some of the charismatic megafauna divers most commonly expressed interest in seeing. We included an ‘other animal’ attribute in the survey, which included reef shark, sea turtle and spiny lobster. Aggregations of lobster are known to attract sport divers in New Zealand (Kelly et al., 2000b), however the potential recreational value of lobsters had not previously been considered in a Caribbean context before and was therefore of substantial interest.

The conjoint survey questions were designed using Sawtooth Software’s Conjoint Value Analysis (CVA) software package (Sawtooth Software, 1996). An optimally efficient paired comparison survey would be both orthogonal (*i.e.*, attributes vary completely independently) and balanced (*i.e.*, each attribute is shown an equal number of times). This survey used five attributes with a total of twenty levels, yielding a potential design space of over 917,000 possible paired comparisons. Because paired comparison surveys are cognitively challenging and can lead to respondent exhaustion (Huber, 1997), it is important to limit the number of questions in a survey. In this survey, we limited the number of questions to eighteen.

The CVA experimental design module identifies promising experimental designs by examining a pool of potential CVA questions selected randomly from the design space. D-efficiency (Kuhfeld et al., 1994), a measure of the goodness of a specific experimental design relative to the ideal orthogonal balanced design, is calculated for a pool of questions five times larger than the desired survey size. Paired comparisons that contribute little to the overall statistical efficiency are eliminated one by one until an 18-question survey design was obtained. This procedure was replicated twenty times and the survey design with the highest D-efficiency was retained and used in the final survey instrument. A sample question is shown in Figure 3: for most people, Option 1 was strongly preferred to Option 2 because it was a dive with a smaller dive group, reef sharks were present, and there were more and larger grouper for only $5 more per dive.

Category	Option A	Option B
Size of the Dive Group	15-23 Divers per Group	24-30 Divers per Group
Presence of Other Animals	1 or more Reef Sharks	No turtle, shark, lobster
Grouper Abundance	6 Groupers per Dive	3 Groupers per Dive
Average Grouper Size	Large Grouper: 30-lbs	Medium Grouper Size: 15-lbs
Price of the Dive	$45 per Single Tank Dive	$ 40 per Single Tank Dive

1	2	3	4	5	6	7	8	9
A is much better		A is somewhat better		A & B are about equal		B is somewhat better		B is much better
Please circle a number from 1 to 9 that reflects your rating								

Figure 3: Paired Comparison Survey Question

A dummy variable OLS regression technique was used to estimate conjoint utilities as a function of the independent variables dive group size, presence of other animals, the abundance and average size of Nassau grouper, and the price of the dive. Based on expert opinion and pilot survey feedback, increasing grouper size and abundance were assumed to provide increasing utility while increasing dive price and group size were assumed to lead to a reduction in respondent utility. No *a priori* relationship was assumed for the presence of other animals.

The regression coefficients – known in marketing literature as part-worth's – were calculated and used in the CVA market simulator. Total utility for each of a variety of hypothetical dive profiles being simulated were calculated for each survey respondent. The product with the highest overall utility for each respondent is assigned a score of '1', while all other profiles are given a score of '0'. The market simulator averages the 'first choice' preference scores across all respondents and calculates percent market share for each hypothetical dive profile in a particular simulation.

The final paired comparison questions, along with questions about environmental attitudes, MPA knowledge, and demographics were distributed to visiting divers in the TCI via commercial dive operators and to student and non-student visitors at the Center for Marine Resource Studies. A total of 80 usable survey responses were used in this analysis; this represents an overall response rate of approximately 30%.

Survey Results

Female respondents accounted for 56% of the 80 usable survey responses. 65% of the respondents were under 30 years of age and 73% of respondents were American citizens. 94% of respondents had at least some college education and 33% had household incomes over US $125,000. This was the first visit to the TCI for 79% of respondents. The most important factors influencing the dive profile choice according to respondents was dive group size (46%) and overall species diversity (36%); only 9% of respondents stated that dive price was the most important factor in their comparison tasks. Individual utility regressions were conducted on the 80 individual survey responses. The overall fit of the regressions was high, with an average R^2 of 0.97. This is indicative of a high level of internal self-consistency in decision making for respondents.

The regression coefficients were then used to calculate utility for various scenarios and calculate market shares for competing dive profiles. Consider first a baseline scenario in which a survey respondent faced four alternative dive package profiles for their next SCUBA dive. They can choose between: (1) a dive with a very large group of 24 – 30 divers for the most economical price of $40; (2) a dive with a slightly smaller group of 15 – 23 divers for a slightly higher fee of $45; (3) a dive with a medium size group of 8 – 14 divers for $50; or (4) a dive with a small group of 3 – 7 divers for the most expensive price of $60. For all cases, assume that the environmental conditions are the same: the diver sees one small 2.3-kg (5-lb) grouper and no other animals (lobster, shark, sea turtle) during a 20-minute dive. Given the stated preferences of the current sample respondents, about 31% of these divers would choose the smallest and most expensive $60 dive profile. Another 34% would choose the medium size group for $50, while the remainder of respondents would choose one of the more economical, larger dive groups.

Now consider alternative dive profiles in which the abundance of Nassau grouper increases compared to the baseline scenario. This could be due to the effects of an MPA or because dive operators take their clients to more remote and pristine dive sites. From a theoretical perspective, variations in travel costs to a particular site are economically equivalent to a per trip entrance fee to the same location (Cameron, 1992). When we simulate an increase in grouper abundance for the most expensive $60 dive profile while holding grouper abundance in other dive profiles constant, we observe an increasing market share for the expensive dive with each increase in Nassau grouper abundance (Figure 4).

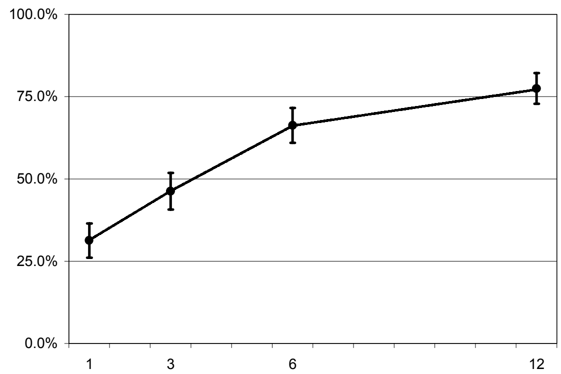

Figure 4: Small Group Dive Profile (p=$60) Market Share for Varying Levels of Grouper Abundance

An increase in abundance from one to twelve Nassau grouper per dive results in market share for the small, expensive dive profile rising from 31% to 78%. Increasing grouper abundance adds value to the dive experience. Dive operators could charge higher prices and increase revenue by taking clients to sites with higher grouper abundance. Alternatively, MPA entrance fees could be used to capture consumer surplus resulting from increased grouper abundance. An increase of $5 per dive, for instance, would have little impact on the number of dives if there were abundant Nassau grouper at a protected dive site. Note that we observe increasing market share but a flattening in the curve from six to twelve Nassau grouper per dive. This is indicative of decreasing average marginal utility for increasing grouper abundance.

Turning to Nassau grouper size, we observe an almost linear increase in market share for the expensive dive profile as average fish size increases from 2.3-kg (5-lbs) to 13.6-kg (30-lbs) average weight (Figure 5). Unlike the case for Nassau grouper abundance, there is no indication of diminishing marginal returns to size.

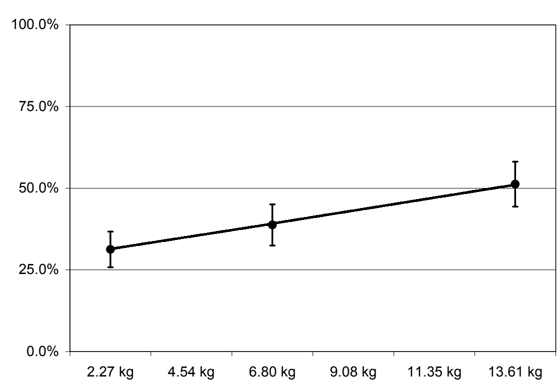

Figure 5: Small Group Dive Profile (p=$60) Market Share for Varying Levels of Grouper Size

The simulations in which the various other animals increased in abundance demonstrated that all the other reef species – spiny lobster, sea turtles and reef sharks – added value to the dive experience of TCI divers. As Figure 6 illustrates, the presence of one or more spiny lobsters increased the small group market share from 31% to 61% in this simulation. Market share for small group dive profiles increased even more, to 85%, with sea turtles and reef sharks.

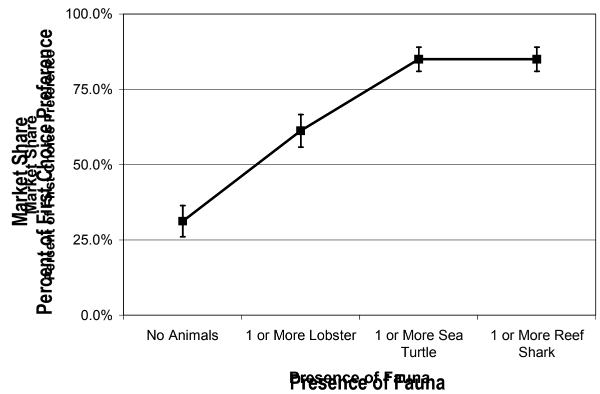

Figure 6: Small Group Dive Profile (p=$60) Market Share for Varying Levels of the Presence of Other Animals

Finally, we simulated the effect of price increases on market share for the small group profile in the absence of any other animals and with a baseline of 1-small 2.3-kg grouper. As the price of the small group dive profile increased from $40 to $60, there was a sharp decline in market share for the small group, from 91.3% (s.e. = 3.16) at $45 per dive, to 71.3% (s.e. = 5.06) at $50 per dive, to 31.3% (s.e. = 5.18) at $60 per dive. This high degree of price sensitivity in the absence of increasing environmental quality is in line with anecdotal information about dive clients from dive operators and is indicative of the price competitiveness of the dive industry in the TCI.

We can also analyze specific demographic groups using the Sawtooth market simulator. In this analysis, we conducted three market segmentation simulations: one of female versus male; one of divers under the age of 30 versus those 30 or older; and one of divers with basic SCUBA certification (resort or open water diver) versus those with advanced certifications (advanced open water or above).

We found no significant differences in market shares for divers with varying certification. Our results showed that there was a significant difference between male and female divers with regards to the price of the small group dive: when the price of the small group dive was $45, the small group market share for females was 86.7%

versus 97.1% for males (t=1.80). The difference in market shares for genders were not significant at higher prices for the small group dive.

When examining younger versus older divers, we found a significant difference in market share for the small group dive. Only 19.2% of the divers under 30 years of age chose the most expensive dive profile, compared to 53.6% of the older divers (t=3.15). In the absence of any special dive features, the older group was less price sensitive than the younger divers. In addition, there were significant differences between younger and older divers for all levels of grouper size (Figure 7). Increasing grouper size, however, added proportionally more value to the dive experience for the younger divers.

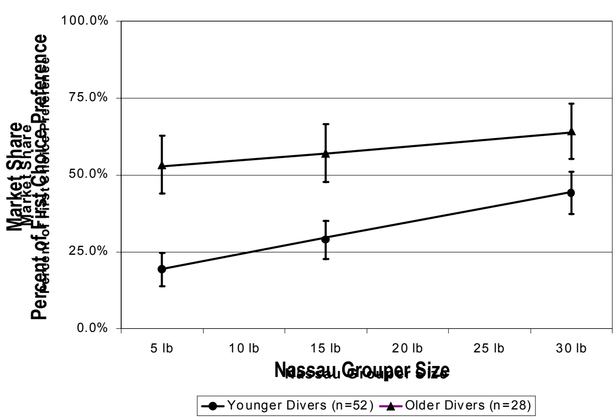

Figure 7: Grouper Size Influence on Small Group Dive Profile (p=$60) Market Share by Demographic Segment

Discussion

Lesson 1. Ecological Effectiveness of MPAs

An important lesson emerges from the South Caicos research – there can be no economic benefits of MPAs if MPAs are ecologically ineffective. This should be a straightforward truism but, unfortunately, MPA design is often guided by principles of geographic convenience or does not fully consider the complex organisms and ecological systems that MPAs are designed to protect. In South Caicos, we currently find little solid evidence of the ecological effectiveness of current MPAs due to the arbitrary design of the reserves.

In the case of spiny lobster, we believe that MPAs are likely to be an ineffective management option for the TCI. The long larval period of the spiny lobster (Lipcius and Cobb, 1994) makes it relatively unlikely that the Caicos Bank lobster population is self-recruiting. There are also complications in linking spawning and recruitment because of the myriad environmental factors that affect larval transport, settlement, and juvenile recruitment (Childress, 1997). If recruitment results from spawning in other areas of the Caribbean, there is little incentive for local fishers to invest in local conservation efforts. The participation of fishers in conservation efforts requires that they feel dependent on the resource and that they have the capacity to impact the state of the system (Mascia, 2000). In South Caicos, the first condition holds but, due to biological and institutional factors, the fishers do not seem to feel that their actions might help conserve the lobster stocks (Olguin et al., 1998). The solution for lobster conservation is likely to lie in more effective enforcement of existing size regulations rather than MPA expansion and, at a regional level, research and negotiations that implement policy at a scale appropriate to take into account the cross-border transport of spiny lobster larvae.

For queen conch, MPAs are much more likely to be effective fisheries management tools in the TCI. Conch have a short larval period (Appeldorn, 1997), so the Caicos Bank population is almost certainly self-recruiting. We suspect that queen conch disperse from core nursery grounds to deeper habitat dominated by seagrass as they grow older. If correct, this means that there should be good potential for designing MPAs that protect core nursery grounds and adjacent adult habitat, producing commercially valuable spillover to areas outside of reserves. When the link between the protection of nursery grounds and spillover benefits to the local fishery can be demonstrated, it is much more likely that fishers will actively support MPAs because it will be in their economic interest.

The ecological effectiveness of MPAs for local reef finfish is also uncertain. While evidence does suggest that size and abundance of key reef species does increase in MPAs (Murray et al., 1999), the effects of any particular MPA are highly dependent on site specific factors (Kramer and Chapman, 1999; Tupper and Juanes, 1999). Like lobster, marine fish species of commercial interest tend to have long larval phases and it is difficult to link larval output to juvenile recruitment. Nassau grouper do migrate from their home range to spawning sites and hence may spillover, for better or worse, to commercial fishing grounds along the migration path.

One issue that is clear in the TCI is the lack of protection for important spawning aggregations and migration corridors. Aggregations are highly vulnerable to fishing pressure (Russ and Alcala, 1989; Russ, 1991; Sluka et al., 1996; Sluka and Sullivan, 1997) and need protection to ensure the maintenance of long-term reproductive capacity. The larval output from aggregations may be valuable at both local and regional levels. MPAs and/or seasonal closures are thus likely to play an important role in the conservation of the stocks of important reef species like groupers, both in the TCI and beyond. For finfish, like queen conch, boundary changes to existing MPAs will be necessary to promote conservation and fishery production benefits. The exact nature of the boundary changes remains under investigation.

Lesson 2. Economic Analysis – Start with the Obvious.

A second lesson that arises from the TCI research is to start economic valuation with the obvious. We have seen that quantifying the ecological effects, and hence the consumptive use value, of MPAs is difficult for the tropical inshore environment. It is relatively easier, however, to assess any increase in size and abundance of animals within an MPA. In a country like the TCI, where tourism is the most important part of the economy, it makes logistical and economic sense to start valuation efforts by focusing on the non-consumptive use value that tourists hold for the natural environment.

Recreational activities such as snorkeling and SCUBA diving are amenable to a variety of valuation methodologies. An important aspect of tourism valuation is that it concentrates on localized spatial and temporal effects. The impacts of increased environmental quality on the reefs are of immediate economic value for snorkelers and SCUBA divers. Divers are also already paying for their experience, so the marginal value of increasing environmental quality has the very real potential of generating more revenue for dive operators or for government. Contrast this with the case of existence value: while people from around the world may hold some intrinsic value for knowing that TCI coral reef organisms are protected, the actual mechanism for transferring the consumer surplus from the beneficiary of conservation efforts (*i.e.*, people around the world) to those that bear the costs of conservation (*i.e.*, TCI residents, fishers, and dive operators) is highly problematic. The ability to relate costs and benefits at local scale is one of the key characteristics of successful renewable resource management institutions (Ostrom, 1990; Gibson et al., 2000).

Lesson 3. Non-Consumptive Use Value can be Significant.

Finally, we have learned from our analysis that the non-consumptive use value that divers hold for increased grouper size and abundance, and for the increased presence of other key reef species, is potentially large.

Increasing Nassau grouper abundance and mean size had positive effects on market share for the expensive dive group in the simulations. In the case of abundance, the net revenue increase was 13.0% for increasing from one to twelve grouper per dive (*i.e.*, the difference in total value between the baseline and target scenarios, □s_ip_i, for market share s_i and dive price p_i, i=1-5 dive group size options). Similarly, the increase in net revenue as a result of an increase from small to large Nassau grouper was 5.6%. Simulations showed diminishing marginal utility for increases in grouper abundance but we did observe a near linear increase in market share with increasing grouper size. Based on conversations with dive operators, it appears that divers may be correlating grouper size and overall dive quality as the rarer, large Nassau grouper are indicative of unfished (Sluka et al., 1998) and therefore superior dive sites

Both reef shark and sea turtles had a large impact on market share in the simulations. This is not unexpected, as both groups of animals are among the most popular with dive tourists. The strong impact of lobster on market share, on the other hand, was unexpected. While it is known that spiny lobster aggregations are an attraction for divers in New Zealand (Kelly et al., 2000b). Our general view in the TCI had been that lobsters were of value only for consumptive use. While our results are preliminary, they are the first of which we are aware that indicate Caribbean spiny lobsters may have a quantifiable non-consumptive value for tourists.

Our survey showed that divers are relatively price insensitive when animal abundance was high but that there was strong resistance to higher prices for dives that differed only by group size. This is consistent with the experience of dive operators in the TCI; they were very cognizant of the importance of dive price as a factor in client decision making. Dive operators fully realize that maintaining reef quality – coral diversity and fish abundance – is crucial to the success of their businesses. At the same time, however, they tended to be very wary of any increases in dive price that might be caused by MPA user fees. Their caution stems from a wariness of the government's ability to actually transform MPA revenue into concrete actions to protect the reefs.

Government, on the other hand, has a mandate to manage public goods for the citizens of the TCI. As with all public goods, there are incentives for users to free-ride on the contributions of others and a tendency for society to under-produce public goods as a result (Ostrom, 1990). The emerging NPS is now funded by a 1% value-added tax on hotel accommodation and meals. Revenue for 2000 is projected at approximately $550,000, but this amount is unlikely to finance the management of all TCI MPAs (F. Homer, personal communication). Our study suggests divers hold significant untapped non-consumptive use value for environmental quality that might be used to increase production of the public goods – ecological services – provided by MPAs.

The policy challenge in the TCI is to balance the needs of rural fishers and the growing dive-tourism industry while tapping the typically unrealized and remarkable WTP for nature-based tourism (Gössling, 1999). Conservation of Nassau grouper and other key finfish species could result in a loss of revenue for artisanal fishers in the TCI. However, in the TCI a $5 increase in the price of a dive might lead to revenue of up to $750,000. Thus, the income generated through premium pricing for access to MPA dive sites might be sufficient to compensate fishers for losses of fishing opportunities due to MPA implementation as well as cover the marginal costs of expanded park operations necessary for the protection for the ecological services crucial to the competitiveness of the dive industry.

What do we yet need to fill in the balance of the ecological and economic puzzle? This survey does not provide a quantitative estimate of consumer surplus, but is a first step in that direction and has already yielded results that lend support for the view that ecologically effective MPAs are economically valuable. The logical follow-up is to develop an expanded study that is distributed to a wider audience, possibly via an exit survey at the national airport. The goal of this survey would be to quantify WTP of tourists for specific marine attributes important to their TCI experience. A broader pre-trip survey (*e.g.*, Huybers and Bennett, 2000) would also be very useful in assessing how environmental quality enters into the decision of divers to travel to the TCI and if business and government support for MPAs have value as a signal to environmentally conscious consumers interested in high-quality dive experiences.

Finally, it is clear that an increased effort must be directed to understanding the ecological systems that MPAs are meant to protect in the TCI. Understanding these systems may allow the economic valuation of increased fishery production and ecological resilience. While a full economic calculus may not be necessary to justify protection – there are indications non-consumptive use values alone may be adequate justification for MPAs in the TCI – quantifying these values can only strengthen the case for the development of ecologically functional marine parks and fishing reserves in the TCI.

Literature Cited

Agardy, M. T. (1994). Advances in marine conservation: the role of marine protected areas. *Trends in Ecology and Evolution* 9 (7): 267-270.

Appeldoorn, R. S. (1997). Status of queen conch fishery in the Caribbean Sea. Paper presented at the International Queen Conch Conference, San Juan, Puerto Rico.

ARA Consulting Group Inc., Systems Caribbean Ltd., Ione Marshall, and KPMG Peat Marwick (Barbados). (1996). A study to assess the economic impact of tourism on selected CDB borrowing member countries. Consultant's Report prepared for Caribbean Development Bank, 30 May 1996.

Beets, J., and A. Friedlander. (1999). Evaluation of a conservation strategy: a spawning aggregation closure for red hind, *Epiniphelus guttatus*, in the U.S. Virgin Islands. *Environmental Biology of Fishes* 55: 91-98.

Bohnsack, J. A. (1993). Marine reserves: they enhance fisheries, reduce conflict, and protect resources. *Oceanus* Fall: 63-71.

Botsford, L. W., J. C. Castilla, and C. H. Peterson. (1997). The management of fisheries and marine ecosystems. *Science* 277: 509-514.

Boyle, K. J., and R. C. Bishop. (1987). Valuing wildlife in benefit-cost analyses: a case study involving endangered species. *Water Resources Research* 23: 943-950.

Brownell, W.N., and J. M. Stevely. (1981). The biology, fisheries, and management of queen conch, *Strombus gigas*. *Marine Fisheries Review* 43: 1-12.

Butler, M. J., and W. F. Herrnkind. (1997). A test of recruitment limitation and the potential for artificial enhancement of spiny lobster (*Panulirus argus*) populations in Florida. *Canadian Journal of Fisheries and Aquatic Sciences* 54: 452-463.

Cameron, T. A. (1992). Combining contingent valuation and travel cost data for the valuation of nonmarket goods. *Land Economics* 68: 302-317.

Carson, R. T., J. J. Louviere, D. L. Anderson, P. Arabie, D. S. Bunch, D. A. Hensher, R. M. Johnson, W. F. Kuhfeld, D. Steinberg, J. Swait, H. Timmermans, and J. B. Wiley. (1994). Experimental analysis of choice. *Marketing Letters* 5: 351-368.

Chapman, M. R. and D.L. Kramer. (1999). Gradients in coral reef fish density and size across the Barbados Marine Reserve boundary: effects of reserve protection and habitat characteristics. *Marine Ecology Progress Series* 181: 81-96.

Chiappone, M, R. Sluka, and K. Sullivan Sealey. (2000). Groupers (Pisces: Serranidae) in fished and protected areas of the Florida Keys, Bahamas and northern Caribbean. *Marine Ecology Progress Series* 198: 261-272.

Childress, M. J. (1997). Marine reserves and their effects on lobster populations: report from a workshop. *Marine and Freshwater Research* 48: 1111-1114.

Christian-Smith, J., and L. Darian. (2000). Analysis of the seafood market in the TCI: effects of the current import tariff. South Caicos: Center for Marine Resource Studies. Unpublished report.

Costanza, R., F. Andrade, P. Antunes, M. van den Belt, D. Boersma, D. F. Coesch, F. Catarino, S. Hanna, K. Limburg, B. S. Low, M. Molitor, J. G. Pereira, S. Rayner, R. Santos, J. A. Wilson, and M. Young. (1998). Principles of sustainable governance of the oceans. *Science* 281: 198-199.

Davis, D., and C. Tisdell. (1998). Tourist levies and willingness to pay for a whale shark experience. *Tourism Economics* 5: 161-174.

Davis, G. E., and J. W. Dodrill. (1985). Marine parks and sanctuaries for spiny lobster fisheries management. *Bulletin of Marine Science* 37: 194-207.

Dixon, J. A. (1993). Economic benefits of marine protected areas. *Oceanus* (Fall): 35-40.
Dixon, J. A., and S. Pagiola. (1998). Economic analysis and environmental assessment. Washington, D.C.: World Bank. Environmental Assessment Sourcebook Update 23.
Farber, S., and B. Griner. (2000). Valuing watershed quality improvements using conjoint analysis. *Ecological Economics* 34: 63-76.
Fletcher, J. J., W. L. Adamowicz, and T. Graham-Tomasi. (1990). The travel cost model of recreational demand: theoretical and empirical issues. *Leisure Sciences* 12: 119-147.
Gibson, C. C., E. Ostrom, and T. K. Ahn. (2000). The concept of scale and the human dimensions of global change: a survey. *Ecological Economics* 32: 217-239.
Gössling, S. (1999). Ecotourism: a means to safeguard biodiversity and ecosystem functions. *Ecological Economics* 29: 303-320.
Hanemann, W. M. (1984). Welfare evaluations in contingent valuation experiments with discrete responses. *American Journal of Agricultural Economics* 66: 332-341.
Hanley, N., R. E. Wright, and W. Adamowicz. (1998). Using choice experiments to value the environment. *Environmental and Resource Economics* 11: 413-428.
Herrnkind, W. F. (1980). Spiny lobsters: patterns of movement. In *The Biology and Management of Lobsters*, edited by J. S. Cobb and B. F. Phillips. New York: Academic Press.
Hesse, K. (1979). Movement and migration of the queen conch, *Strombus gigas*, in the Turks and Caicos Islands. *Bulletin of Marine Science* 29: 303-311.
Homer, F. (2000). *Management Plan for the Northwest Point National Marine Park and West Caicos Marine National Park, 2000-2004.* Providenciales, Turks and Caicos Islands: Coastal Resources Management Project, Ministry of Natural Resources.
Huber, J. (1997). What have we learned from 20 years of conjoint research: when to use self-explicated, graded pairs, full profile or choice experiments. Paper Presented at Sawtooth Software Annual Conference.
Hudson, E., and G. Mace. (1996). Marine fish and the IUCN red list of threatened animals. Report of the workshop held in collaboration with WWF and IUCN at the Zoological Society of London, 29 April - 1 May 1996.
Huybers, T., and J. Bennett. (2000). Impact of the environment on holiday destination choices of prospective UK tourists: implications for Tropical North Queensland. *Tourism Economics* 6: 21-46.
Johannes, R. E. (1998). The case for data-less marine resource management: examples from tropical nearshore finfisheries. *Trends in Ecology and Evolution* 13: 243-246.
Johnson, F. R., W. H. Desvousges, E. E. Fries, and L. L. Wood. (1995). Conjoint analysis of individual and aggregate environmental preferences. Durham, NC: Triangle Economic Research. TER Technical Working Paper T-9502.
Johnson, F. R., M. C. Ruby, W. H. Desvousges, and J. R. King. (1998). Using stated preferences and health-state classifications to estimate the value of health effects of air pollution: final report (revised October 1998). Durham, NC: Triangle Economic Research. Consultant's report prepared for Environment Canada, Health Canada, Ontario Hydro, Ontario Ministry of Environment and Energy, and Environnement et de la Faune Quebec, 31 March 1998.
Johnson, D. R., N. A. Funacelli, and J. A. Bohnsack. (1999). Effectiveness of an existing estuarine no-take fish sanctuary within the Kennedy Space Center, Florida. *North American Journal of Fisheries Management* 19: 436-453.
Kassakian, J. M. (1999). Finfish landings and abundance on South Caicos, TCI, BWI: current status and ideas for the future. South Caicos: Center for Marine Resource Studies. Unpublished report.
Kelly, S., D. Scott, A. B. MacDiarmid, and R. C. Babcock. (2000a). Spiny lobster, *Jasus edwardsii*, recovery in New Zealand marine reserves. *Biological Conservation* 92: 359-369.
Kelly, S., A. B. MacDiarmid, D. Scott, and R. Babcock. (2000b). The value and impact of a spill-over fishery for spiny lobsters around a marine reserve in northern New Zealand. Paper presented at Economics of Marine Protected Areas Conference, UBC Fisheries Center, Vancouver, 6-7 July 2000.
King, M. (1995). *Fisheries Biology, Assessment and Management*. Oxford: Blackwell Scientific Publications.
Kramer, D. L., and M. R. Chapman. (1999). Implications of fish home range size and relocation for marine reserve function. *Environmental Biology of Fishes* 55: 65-79.
Kuhfeld, W. F. (1997). Efficient experimental designs using computerized searches. Paper presented at Sawtooth Software Annual Conference
Lipcius, R. N., and J. S. Cobb. (1994). Introduction: ecology and fishery biology of spiny lobsters. In *Spiny Lobster Management*, edited by B. F. Phillips, J. S. Cobb and J. Kittaka. Oxford: Blackwell Scientific Publications.
Loomis, J., and D. M. Larson. (1994). Total economic values of increasing gray whale populations: results from a contingent valuation survey of visitors and households. *Marine Resource Economics* 9: 275-286.
Louviere, J. J. (1988). Conjoint analysis modeling of stated preferences. *Journal of Transport Economics and Policy* 22: 93-119.
MacDiarmid, A. B., and P. A. Breen. (1993). Spiny lobster population changes in a marine reserve. Paper presented at the Second International Temperate Reef Symposium, Wellington NZ.
Man, A., R. Law, and N. V. C. Polunin. (1995). Role of marine reserves in recruitment of reef fishes: a metapopulation model. *Biological Conservation* 71: 197-204.
Mascia, M. B. (2000). Institutional emergence, evolution, and performance in complex resource systems: marine protected areas in the Wider Caribbean. Paper presented at 8th Biennial Conference of the International Association for the Study of Common Property, Bloomington Indiana, 31 May - 4 June 2000.
Medley, P. A., G. Gaudian, and S. Wells. (1993). Coral reef fisheries stock assessment. *Reviews in Fish Biology and Fisheries* 3: 242-285.
Moberg, F., and C. Folke. (1999). Ecological goods and services of coral reef ecosystems. *Ecological Economics* 29: 215-233.
Moran, K. (1992). Descriptions of fishing locations near South Caicos, TCI, BWI. South Caicos: Center for Marine Resource Studies. Unpublished report.
Mulliken, T. A. (1996). Status of the queen conch fishery in the Caribbean. *TRAFFIC Bulletin* 16: 17-28.
Munro, J. L., and D. M. Williams. (1985). Assessment and management of coral reef fisheries: biological, environmental and socioeconomic aspects. *Proceedings of the 5th International Coral Reef Symposium* 4: 545-581.
Murray, S. N., R. F. Ambrose, J. A. Bohnsack, L. W. Botsford, M. H. Carr, G. E. Davis, P. K. Dayton, D. Gotshall, D. R. Gunderson, M A. Hixon, J. Lubchenco, M. Mangel, A. MacCall, D. A. McArdle, J. C. Ogden, J. Roughgarden, R. M. Starr, M. J. Tegner, and M. M. Yoklavich. (1999). No-take reserve networks: sustaining fishery populations and marine ecosystems. *Fisheries* 24: 11-25.

Nagelkerken, W. (1981). Distribution of the groupers and snappers of the Netherlands Antilles. *Proceedings of the Fourth International Coral Reef Symposium* 2: 479-484.

National Marine Fisheries Service. (1997). *Status of Fisheries of the United States*. Washington, D.C.: NMFS.

National Research Council. (1999). *Sustaining Marine Fisheries*. Washington D.C.: National Academy Press.

Ninnes, C. 1994. A review on Turks and Caicos Islands fisheries for *Strombus gigas* L. In *Queen Conch Biology, Fisheries, and Mariculture*, edited by R. S. Appeldoorn and B. Rodriguez. Caracas, Venezuela: Fudacion Cientifica Los Roques.

North, D. C. (1990). *Institutions, Institutional Change and Economic Performance*. Cambridge, UK: Cambridge University Press.

Olguin, J., W. Oppenheimer, K. Suter, and J. E. Todisco. (1998). The lobster "big grab" in South Caicos, Turks and Caicos Islands: institutional origins and biological consequences. South Caicos: Center for Marine Resource Studies. Unpublished report.

Ostrom, E. (1990). *Governing the Commons: The Evolution of Collective Action*. Cambridge, UK: Cambridge University Press.

Perrings, C., K.-G. Mäler, C. Folke, C. S. Holling, and B.-O. Jansson, eds. (1995). *Biodiversity Loss: Economic and Ecological Issues*. Cambridge, UK: Cambridge University Press.

Plan Development Team. (1990). The potential of marine fishery reserves for reef fish management in the U.S. southern Atlantic. Washington, D.C.: U.S. Department of Commerce.

Polunin, N. V. C., and C. M. Roberts. (1993). Greater biomass and value of target coral-reef fishes in two small Caribbean marine reserves. *Marine Ecology - Progress Series* 100: 167-176.

Rakitin, A., and D. L. Kramer. (1996). The effect of a marine reserve on the distribution of coral reef fishes in Barbados. *Marine Ecology Progress Series* 131: 97-113.

Randall, A. (1993). Passive-use values and contingent valuation - valid for damage assessment. *Choices* (Second Quarter): 12-15.

Roberts, C. M. (1997). Ecological advice for the global fisheries crisis. *Trends in Ecology and Evolution* 12: 35-38.

Roberts, C. M., and N. V. C. Polunin. (1993). Marine reserves: simple solutions to managing complex fisheries? *Ambio* 22: 363-368.

Roe, B., K. J. Boyle, and M. F. Teisl. (1996). Using conjoint analysis to derive estimates of compensating variation. *Journal of Environmental Economics and Management* 31: 145-159.

Ruitenbeek, H. J. (1999). Blue pricing of undersea treasures - needs and opportunities for environmental economics research on coral reef management in Southeast Asia. Paper presented at the 12th Biannual Workshop of the Environmental Economics Program for Southeast Asia, Singapore, 11-14 May 1999.

Russ, G. R. (1991). Coral reef fisheries: effects and yields. In *The Ecology of Fishes on Coral Reefs*, edited by P. F. Sale. San Diego, California: Academic Press.

Russ, G. R., and A. C. Alcala. (1989). Effects of intense fishing pressure on an assemblage of coral reef fishes. *Marine Ecology Progress Series* 56: 13-27.

Sawtooth Software. (1996). *CVA System, Version 2.0*. Sequim, Washington: Sawtooth Software.

Sluka, R. (1995). *Influence of habitat on density, species richness, and size distribution of groupers in the upper Florida Keys, USA and Central Bahamas*. Ph.D. Dissertation, University of Miami, Coral Gables, Florida.

Sluka, R., M. Chiappone, K. Sullivan, and R. Wright. (1996). Assessment of grouper assemblages. In *Habitat and Life in the Exuma Cays, Bahamas*. Nassau, Bahamas: Media Publishing Ltd.

Sluka, R., and K. M. Sullivan. (1997). The influence of spear fishing on species composition and size of grouper on patch reefs in the upper Florida Keys. *Fishery Bulletin* 96: 388-392.

Sluka, R., M. Chiappone, K. M. Sullivan, T. A. Potts, J .M. Levy, E. F. Schmitt, and G. Meester. (1998). Density, species and size distribution of groupers (Serranidae) in three habitats at Elbow Reef, Florida Keys. *Bulletin of Marine Science* 62: 219-228.

Stager, J.C., and , and V. Chen. (1996). Fossil evidence of shell length decline in queen conch (*Strombus gigas* L.) at Middleton Cay, Turks and Caicos, British West Indies. *Caribbean Journal of Science* 32: 14-20.

Stoner, A. W., and J. M. Waite. (1990). Distribution and behavior of queen conch, *Strombus gigas*, relative to seagrass standing crop. *Fishery Bulletin* 88: 573-585.

Stoner, A. W., and V. J. Sandt. (1992). Population structure, seasonal movements and feeding of queen conch, *Strombus gigas*, in deep water habitats of the Bahamas. *Bulletin of Marine Science* 51: 287-300.

Stoner, A. W., and M. Ray. (1996). Queen conch, *Strombus gigas*, in fished and unfished locations of the Bahamas: effects of marine fishery reserve on adult, juveniles, and larval production. *Fishery Bulletin* 94: 551-565.

Stoner, A. W., and M. Davis. (1997). Abundance and distribution of queen conch veligers (*Strombus gigas* Linne) in the central Bahamas. I. Horizontal patterns in relation to reproductive and nursery grounds. *Journal of Shellfish Research* 16: 7-18.

Tewfik, A., and C. Béné. In press. Densities and age structure of fished versus protected populations of queen conch (*Strombus gigas*) in the Turks and Caicos Islands. *Proceedings of the Gulf Caribbean Fisheries Institute* 51.

Tupper, M. (1999). A brief review of grouper reproductive biology and implications for management of the Gulf of Mexico gag grouper fisheries. Consultant's report prepared for Southeastern Fisheries Association Inc., Florida Offshore Fishing Consortium and Southern Offshore Fishing Association, December 1999.

Tupper, M, and F. Juanes. (1999). Effects of a marine reserve on recruitment of grunts (Pisces: Haemulidae) at Barbados, West Indies. *Environmental Biology of Fishes* 55: 53-63.

Turks & Caicos Islands Government. (1996). *Statistical Yearbook of the Turks and Caicos Islands*. Grand Turk: Office of the Accountant General.

Wooten, R. J. (1990). *Ecology of Teleost Fishes*. London: Chapman and Hall.

The Impacts of Marine Reserves On Limited Entry Fisheries

James N. Sanchirico[1] and James E. Wilen[2]

[1]Quality of the Environment Division, Resources for the Future
[2]Department of Agricultural and Resource Economics, University of California, Davis
All correspondence to: James N. Sanchirico, Quality of the Environment Division, Resources for the Future, *1616 P Street NW, Washington DC 20036, Email: sanchirico@rff.org*

Abstract

We utilize a spatial bioeconomic model to investigate the impacts of creating reserves on limited entry fisheries. We find that reserve creation can produce win-win situations where aggregate biomass and the common license (lease) price increase. These situations arise in biological systems where dispersal processes are prevalent and the fishery prior to reserve creation is operating at effort levels in a neighborhood of open-access levels. We also illustrate that using strictly biological criteria for siting reserves (e.g., setting aside the most biological productive areas) will likely induce the most vociferous objections from the fishing industry. In general, we find that the dispersal rate and the degree the patches are connected play a significant role on the net impacts on the fishing sector.

Keywords:
Fisheries; Limited-Entry; Marine Reserves.
JEL Classification: Q22 and R10.

Introduction

In June 2000, U.S. President Clinton signed an Executive Order directing agencies responsible for marine conservation to develop a plan to establish a comprehensive system of marine reserves within U.S. coastal waters. This was rightly hailed as a major victory by marine conservationists and scientists who have been vocal supporters of marine reserves for close to a decade. To long-time observers of the politics of fisheries management in the U.S., this action by the President is viewed by many as miraculous in a system more typically regarded as sluggish, provincial, and not prone to radical change. In many ways, the very rapid acceptance of the concept of marine reserves in U.S. policy is more reminiscent of radical executive order innovations in Canada thirty years ago establishing limited entry for the first time, and in New Zealand twenty years ago establishing individual transferable quota programs.

Despite the fanfare, it would be premature for proponents of reserves to declare victory before the details are worked out. With marine reserves, the devil will certainly be in the details, and whether the U.S. ends up with anything close to what is envisioned by marine conservationists will certainly depend upon a lengthy process of debate over different design options, scales, and visions. And this debate will reflect strengths of opinions among different stakeholder groups: from scientists, to representatives of the conservation community, to fishermen and others who depend upon current configurations of regulations for their livelihoods. Fishermen will clearly play an important role in the political process, since the costs and benefits to them are much more tangible than the equally important but less easily quantifiable conservation benefits held by the public at large.

This paper examines the question: how will various marine reserve options affect fishermen participating in limited entry fisheries? We ask the question from the fishermen's perspective since we believe that the fishing industry will be the most likely and most effective opponent of reserves, if any group emerges to oppose them. In an open political system like that present in the U.S., the political process will eventually craft compromises and tradeoffs that reflect perceptions of gains and losses, and industry opposition will mobilize to block plans that might have large negative impacts on fishermen and coastal communities. Alternatively, the process will favor those plans that seem to involve "win-win" situations in which fishermen can gain from reserves, or situations in which reserve costs to the industry are relatively low.[1]

We examine fisheries characterized by limited entry regulatory schemes for several reasons. First, most important fisheries in developed countries worldwide are either subject to some kind of limited entry program or are likely to have such programs in the near future. Second, we expand on some related work that addresses similar questions, but under the assumption of open-access institutions.[2] Third, there are some interesting issues related to the manner in which license prices themselves are good signals about the overall economic health of the fishing industry, and hence good gauges of the economic impact of reserves.

In the next section we develop a simple spatial bioeconomic model of a limited entry fishery harvesting a metapopulation. The model allows us to depict various behavioral characteristics of both the fishing industry and the population biology. The model is used to simulate the introduction of a marine reserve in a patchy biological system. We focus on license prices and on aggregate industry rents to characterize the impacts of reserves. The last section summarizes and discusses further issues for investigation.

The Model

The foundations of the model employed here is developed in more detail in Sanchirico and Wilen [1999a, 2000]. The basic structure combines a standard biological metapopulation model with a reformulation of economic models developed by both H.S. Gordon [1954] and V. Smith [1968]. The complete integrated bioecono mic model can be written as:

$$\dot{x}_i = f_i(x_i)x_i + d_{ii}x_i + \sum_{\substack{j=1 \\ j \neq i}}^{n} d_{ij}x_j - h_i \qquad \forall i = 1,\ldots,n \tag{1}$$

$$\dot{E}_i = s_i R_i(E_i, x_i) + \sum_{\substack{j=1 \\ j \neq i}}^{n} s_{ij}[R_i(E_i, x_i) - R_j(E_j, x_j)] \qquad i = 1,\ldots,n \tag{2}$$

The first component is the biological system, which depicts the evolution of biomass levels in n separate biological patches where x_i is the biomass level in patch i, $f_i(x_i)$ is the per capita growth rate in patch i, h_i is the harvest rate in patch i, d_{ii} is the rate of emigration from patch i ($d_{ii}<0$), and d_{ij} is the dispersal rate between patches i and j. The biological system depicted here is a standard linear metapopulation model in which there are n discrete patches in space, each of which is characterized by "own" patch dynamics as well as linkages to other patches.[3] In this formulation, own growth is separable from dispersal, and the dispersal process can be flexibly modeled via appropriate choice of the coefficient d_{ij}.[4] In this paper, we follow the long tradition in the ecology literature and depict the dispersal processes as density dependent.[5] In density dependent dispersal processes, biomass flows between patches in a manner dependent upon the relative densities of each patch. The simplest representation of a density dependent dispersal process depicts the dispersal mechanism between patch one and two as $d_{11}x_1+d_{12}x_2 \equiv b(x_2/k_2-x_1/k_1)$, and between patch two and one as $d_{22}x_2+d_{21}x_1 \equiv b(x_1/k_1-x_2/k_2)$.

The second component is a behavioral model of a harvesting industry operating over a heterogeneous environment that depicts the fleet responding to economic variables over both time and space. Let E_i denote the patch-specific levels of effort in each patch i, and let $R_i(E_i,x_i)$ be corresponding rents (or profits) expected in patch i. Then we can hypothesize a simple sluggish adjustment process in which the level of effort, E_i in patch i, changes according to equation 2. In this specification, effort in patch i changes in response to the level of rents $R(E_i, x_i)$ vis a vis outside opportunities (captured in the first term), and net dispersal (depicted by the second term). The second term consists of a sum of pairwise spatial dispersal rates, each proportional to rent differentials across space between the patch in question and alternative patches. Hence there will be dispersal from patch j into patch i if rents in i exceed those in j, and dispersal to j from i if the net difference is negative. At any point in time, patch i may be contributing to a subset of patches experiencing higher relative rents and drawing from another subset experiencing relatively lower rents. For the system as a whole, these spatial forces tend to redistribute effort over space in a manner that, in the long run, equalizes net rents across all patches.[6]

The above system (equations 1 and 2) is capable of addressing a variety of questions about how fishing efforts will distribute itself over time and space, and how that distribution of absolute and relative effort will affect the biological system through own growth and dispersal. To close the model we need to specify the rent functions in ways that characterize institutional features of the fisheries that are of interest. In Sanchirico and Wilen [1999a] we assumed that the fishery is an open-access fishery in which effort flows into the fishery and across space in a way that ultimately dissipates rent in equilibrium. In particular, if we assume a Schaefer production function ($h_i=q_iE_ix_i$, where q_i is the catchability coefficient), constant patch-specific costs per unit effort (c_i), and a common vessel capital opportunity cost (π), we have

$$R(E_i, x_i) = pq_iE_i\,x_i - (c_i + \pi)E_i \tag{3}$$

as the aggregate rent function for patch i, where p is the ex-vessel price. With these patch-specific rent functions inserted into the system above, we can examine both transition paths and equilibria of various systems with different biological and economic characteristics.[7]

The spatial and intertemporal bioeconomic system outlined here is particularly useful for examining the impacts of reserve formation (Sanchirico and Wilen [2000]).[8] If we begin, for example, with a system in which harvesters freely move across all patches in a biological system, we can characterize the nature of the exploited equilibrium that would emerge, as well as the nature of the adjustment process to that exploited equilibrium. In this (pre-reserve) equilibrium, the level of own biological growth in each patch will be exactly offset by the total net dispersal between the patch and other linked patches, and the harvest in the patch in question. In addition, net rents will be identically equal to zero in each patch, leading to an economic equilibrium over time and space. The pre-reserve bioeconomic steady-state can be formally written as

$$\dot{x}_i \overset{set}{=} 0 \Rightarrow \quad f_i(x_i)x_i + d_{ii}x_i + \sum_{\substack{j=1 \\ j\neq i}}^{n} d_{ij}x_j = h_i \qquad \text{i=1,..,n}$$

$$\dot{E}_i \overset{set}{=} 0 \Rightarrow \quad R_i(E_i, x_i) = 0 \qquad \text{i=1,..,n} \tag{4}$$

Note that while the biological dispersal coefficients (d_{ii} and d_{ij}) affect the equilibrium vector of biomass and effort levels in each patch, the economic response parameters (s_i and s_{ij}) only affect the speed of response to equilibrium. This occurs because the economic system equilibrates when net rents in each patch are zero, and the conditions that generate zero rents are independent of the response rates, as in the Vernon Smith model of a single patch.

Reserve Creation in a Limited Entry Fishery

With straightforward modifications of the rent functions, the system discussed above can be used to depict a fishery that is regulated by a **limited entry licensing** system, with the licenses placed on vessels. The simplest way to see this is to note that the licensing system will create rights that have some value to existing and potential participants. In equilibrium, the price of licenses will rise until the opportunity cost is equal to the anticipated production rents in the fishery. Let L be the equilibrium license **lease** price in a limited entry system, that is, the amount that a potential entrant would be willing to pay an existing participant to lease his/her vessel-specific license for a period. In equilibrium, we will have

$$R(E_i, x_i) = pq_iE_i\, x_i - (c_i + \pi)E_i - LE_i = 0 \tag{5}$$

across all patches, where the (common) license lease price is endogenously determined and is a function of the total number of licenses outstanding.[9]

Thus, the license lease price rises to eliminate rents at two margins. The first margin is related to outside opportunities for the vessel capital in question. In equilibrium, owners of vessel capital will be indifferent between participating in the alternative fishery and earning π per unit capital, or participating in the limited entry fishery by paying the lease price L and earning $pq_ix_i - (c_i + \pi)$ per vessel in the fishery. The second margin exists between any patch i and another alternative patch j. In the full spatial equilibrium, a vessel owner facing lease price L will be indifferent between fishing in patch j and in patch i, so that $pq_ix_i - (c_i + \pi) = L = pq_jx_j - (c_j + \pi)$.[10]

To examine the implications of reserve creation within this system of limited entry licensing, we develop a simple three-patch exercise. In this exercise, we compare two systems, one without a reserve and one with a reserve located in patch three.[11] Assuming that the own growth functions in each patch are quadratic, the system of equations that define the steady-state equilibrium for each of the two cases are in Table 1.

Table 1: Two Systems of Equations Defining the Steady-State Equilibrium

EQUATION (6): WITHOUT A RESERVE	EQUATION (7): WITH A RESERVE
$r_ix_i(1 - \frac{x_i}{K_i}) - d_{ii}x_i + \sum_{j=1}^{2} d_{ij}x_j - q_ix_iE_i = 0$, i=1,2,3	$r_ix_i(1 - \frac{x_i}{K_i}) - d_{ii}x_i + \sum_{j=1}^{2} d_{ij}x_j - q_ix_iE_i = 0$, i=1,2
$pq_ix_i - (c_i + p) - L = 0$ i=1,2,3	$r_kx_k(1 - \frac{x_k}{K_k}) - d_{kk}x_k + \sum_{j=1}^{2} d_{kj}x_j = 0$ k=3
$E^{TOT} = \sum_{i=1}^{3} E_i$	$pq_ix_i - (c_i + p) - L = 0$ $i = 1,2$
	$E^{TOT} = \sum_{i=1}^{2} E_i$

We use these systems to compute equilibrium values for biomass, harvest, effort in each patch and the (common) license price. Understanding the impacts of reserves on license prices is important, because license values are indicative of the total rents in the system. Any policy that increases aggregate rents in the industry will also increase license values; policies that decrease rents will decrease license values.

In the remainder of this section, we discuss the impacts of reserve creation within closed, fully-integrated, and cascade systems (see Figure 1). As it turns out, we can analytically derive the results for the closed system. In the more complex and linked systems however, the systems of equations are non-linear. We numerically simulate the impacts of reserve creation in these cases.[12]

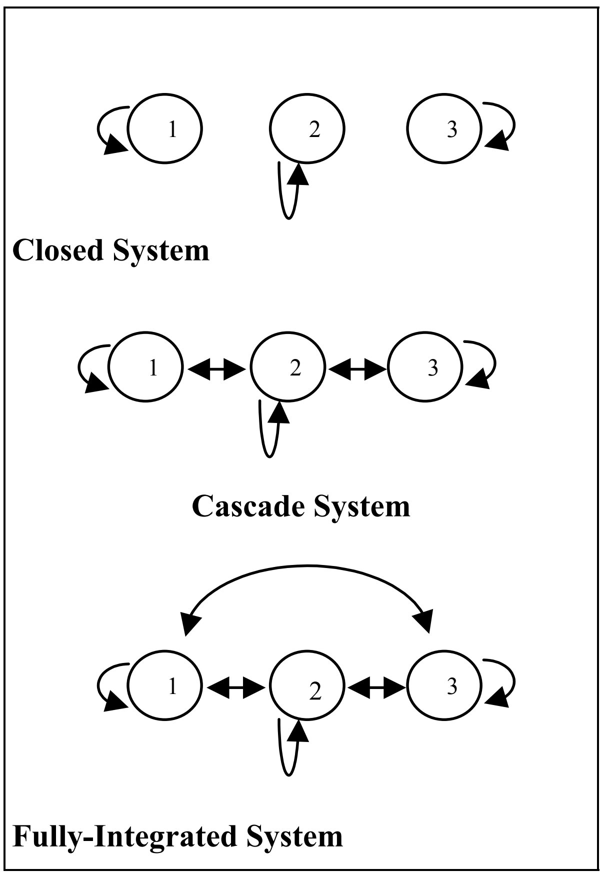

Figure 1: Biological Systems*

* The arrows represent the own and dispersal feedback mechanisms.

Closed Biological System

As Sanchirico and Wilen [2000] have shown in the case of an open-access fishery, reserve creation in a **closed** metapopulation system increases aggregate biomass but fails to provide any benefits to the industry. In fact, due to the absence of spillover from the reserve, the reduction in harvest is greater than in the fully-integrated and cascade systems. Intuitively, we would expect that in the limited entry setting, the open-access results would hold qualitatively, because there still is no mechanism for biological spillover effects. In fact they do. The steady-state levels for the closed case are provided in Table II. While this case is very restrictive, we find that it illustrates rather clearly the necessary mechanisms for the industry to benefit from closures.

We solve for the steady-state equilibrium levels of biomass, effort, and license price in this setting by using the fact that $pq_i x_i - (c_i + \pi) = L = pq_j x_j - (c_j + \pi)$ and the steady state equations (equations 6 and 7). As is evident in Table II, the results illustrate not only the impact of a reserve, but also differences between equilibria in the open-access system and a limited-entry licensing system. Of course when the amount of effort limited in the fishery (E^{TOT}) is equal to the aggregate open-access levels, then the biomass and effort levels equal the open-access levels. In this case, all rents are dissipated and the license value is zero. Thus, the open-access aggregate effort level provides a natural upper bound to the total amount of effort. As the amount of total system effort is constrained to be smaller than the open-access equilibrium level, the "shadow price" on the constraint (the license value) rises. For any given amount of effort in the system, there will be a unique license value of L, and that value will be equal to the (common) rent level in all patches.

As it turns out, the implication of shutting down patch three in this setting is rather straightforward. The biomass density level there will equilibrate at its carrying capacity and the effort level will be zero. In the open patches, the more constrained the total effort, the higher the biomass and the system-wide license price, and the lower the patch-specific effort levels in remaining open areas. As discussed earlier, in a licensed limited-entry system, the license price provides a signal on the impact of policies on fishery rents. If the license price increases after patch three is closed, then fishery rents have increased due to the closure. Of course, if the license price decreases, then rents have decreased.

Table 2: Equilibrium Levels with and without a reserve located in patch three

EQUATION(8): WITHOUT A RESERVE	EQUATION (9): WITH A RESERVE
$x_i = x_i^{OA} + \frac{1}{q_i \sum_{i=1}^{n} \frac{r_i}{q_i^2}}[\sum_{i=1}^{3} E_i^{OA} - E^{TOT}]$ $E_i = \frac{r_i}{q_i}(1 - x_i^{OA} - \frac{1}{q_i \sum_{i=1}^{3} \frac{r_i}{q_i^2}}[\sum_{i=1}^{3} E_i^{OA} - E^{TOT}])$ $L^{ss}_{d_{ij}=0} = \frac{p}{\sum_{i=1}^{3} \frac{r_i}{q_i^2}}[\sum_{i=1}^{3} E_i^{OA} - E^{TOT}]$	$x_i = x_i^{OA} + \frac{1}{q_i \sum_{i=1}^{2} \frac{r_i}{q_i^2}}[\sum_{i=1}^{2} E_i^{OA} - E^{TOT}],\ i = 1,2$ $x_k = K_k,\ k = 3$ $E_i = \frac{r_i}{q_i}(1 - x_i^{OA} - \frac{1}{q_i \sum_{i=1}^{2} \frac{r_i}{q_i^2}}[\sum_{i=1}^{2} E_i^{OA} - E^{TOT}]),\ i = 1,2$ $E_k = 0,\ k = 3$ $L^{ss}_{d_{ij}=0} = \frac{p}{\sum_{i=1}^{2} \frac{r_i}{q_i^2}}[\sum_{i=1}^{2} E_i^{OA} - E^{TOT}]$
where $E_i^{OA} = r_i(1 - \frac{c_i + p}{pq_i})$ and $x_i^{OA} = \frac{c_i + p}{pq_i}$	

We can illustrate these results either in terms of aggregate fishery rents or equilibrium license prices. The function depicting aggregate license values (equilibrium license price times the number of licenses outstanding) has a maximum at the total effort levels that maximize rents.[13] In the pre-reserve system, for example, the optimal number of licenses is $E^{TOT*} = \frac{1}{2}\sum_{i=1}^{n} E_i^{OA}$.

Post reserve creation, the optimal number of licenses decreases by $1/2 * E_3^{OA}$. In a biologically closed system, generally the post-reserve license value function is everywhere lower than the pre-reserve function (see Figure 2 panel A), suggesting that reserves in a closed system do not improve economic conditions for fishermen. An implication of this is that the rent-maximizing level of effort after reserves are created is at a lower aggregate effort level, suggesting that a reserve policy designed to maximize industry rents may have to employ a buy-back or other effort-reducing scheme to eliminate excess effort.

Linked Metapopulation Systems

Upon reflection, it should not be surprising that creating a reserve in a complete decoupled system does not increase aggregate rents to fishermen. This is because there is no direct fisheries payoff to the closure in a biological system in which there is no dispersal. For fishermen to benefit from a closure, the closed patch must contribute some spillover of biomass to the remaining open patches, and that requires some dispersal. Accordingly, in this section, we simulate a closure in patch 3 but in **fully-integrated** and **cascade** systems in which there is density-dependent biological dispersal between the patches. In the cascade system, closing patch 3 corresponds to closing a patch on the edge of the system where the biomass only directly interacts with patch 2. Both of these biological structures illustrate circumstances that can lead to potential "win-win" scenarios.

For the fully-integrated and cascade systems, we numerically simulate the derived demand for licenses and the maximum aggregate rent curves pre and post reserve creation. We continue to assume that the own growth functions in each patch are quadratic with equal intrinsic growth rates and carrying capacities (r=0.8, K=1), and with dispersal rates (b=0.4) equal to half the own growth rate. We initially introduce heterogeneity into the system by assuming that patch 3 is a relatively higher cost patch to fish than the other patches (c_3=13.0 and c_1=c_2=11.0). The price is assumed identical across all patches (p=65) and catchability coefficients are normalized to 1.0.

The simulation finds values of effort and biomass across all patches that bring the system into a bioeconomic equilibrium, given some fixed amount of effort (E^{TOT}). The results are presented in Figures 2 and 3. For comparison purposes, we simulate a reserve in the biologically closed case, which complements our earlier analytical results. For all cases, Figures 2 and 3 can be read as follows. Suppose that we have a limited entry system with some reasonably restricted level of effort such as 1.08. Then, before a reserve is implemented in the biologically closed case (Panel A, Figures 2 and 3), license prices would have equilibrated at a level of 21.52 per unit of effort, and aggregate license values will be 23.31. After patch 3 is closed, license values per unit effort fall to 7.85, and aggregate license values (rent in the system) fall to 8.51.

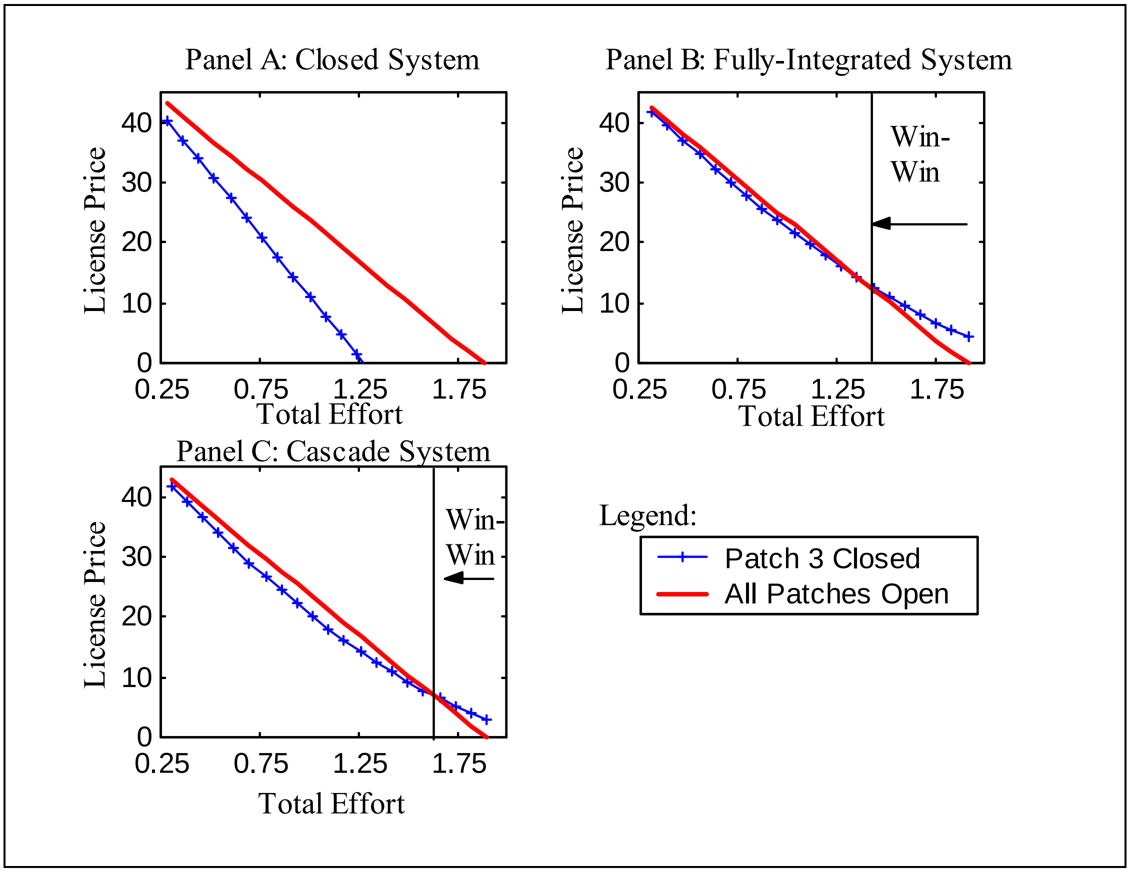

Figure 2: Derived demands for effort with and without a reserve in patch **3**

We first discuss the impacts of closing patch 3 within a **fully-integrated** system in which there is density-dependent biological dispersal between the patches. Panel B in Figures 2 and 3 shows the impact of reserve creation in this kind of biological system. It is important to emphasize several important results, including some comparisons of the fully-integrated system with the closed case. First, the difference between system rents before and after the creation of a reserve is smaller than in the closed system. Second, over most of the range of total effort, a marine reserve costs the industry some rents. Thus while dispersal from the closed patch helps make up for the closure, on net the increase in yields after the closure is not sufficient to compensate for the lost opportunities in the closed patch. However, and interestingly, at high initial levels of effort, a reserve actually increases license values, suggesting possible "win-win" scenarios. This is consistent with Sanchirico and Wilen [2000] who show that "win-win" reserve designs are more probable when the closed patch is over-depleted, and when dispersal occurs after closure.

While the closed and fully-integrated cases define the bounds in terms of the degree of biological connectedness, the cascade system illustrates the impacts of siting reserves in intermediate cases. In addition, this case illustrates how the placement of a reserve can affect the overall outcome. Recall that in this case, patch three is only directly connected to patch two, which is the center patch. From panel C of Figure 2, it is evident that the range of effort levels corresponding to the "win-win" scenario is smaller than in the fully-integrated setting, everything else equal.

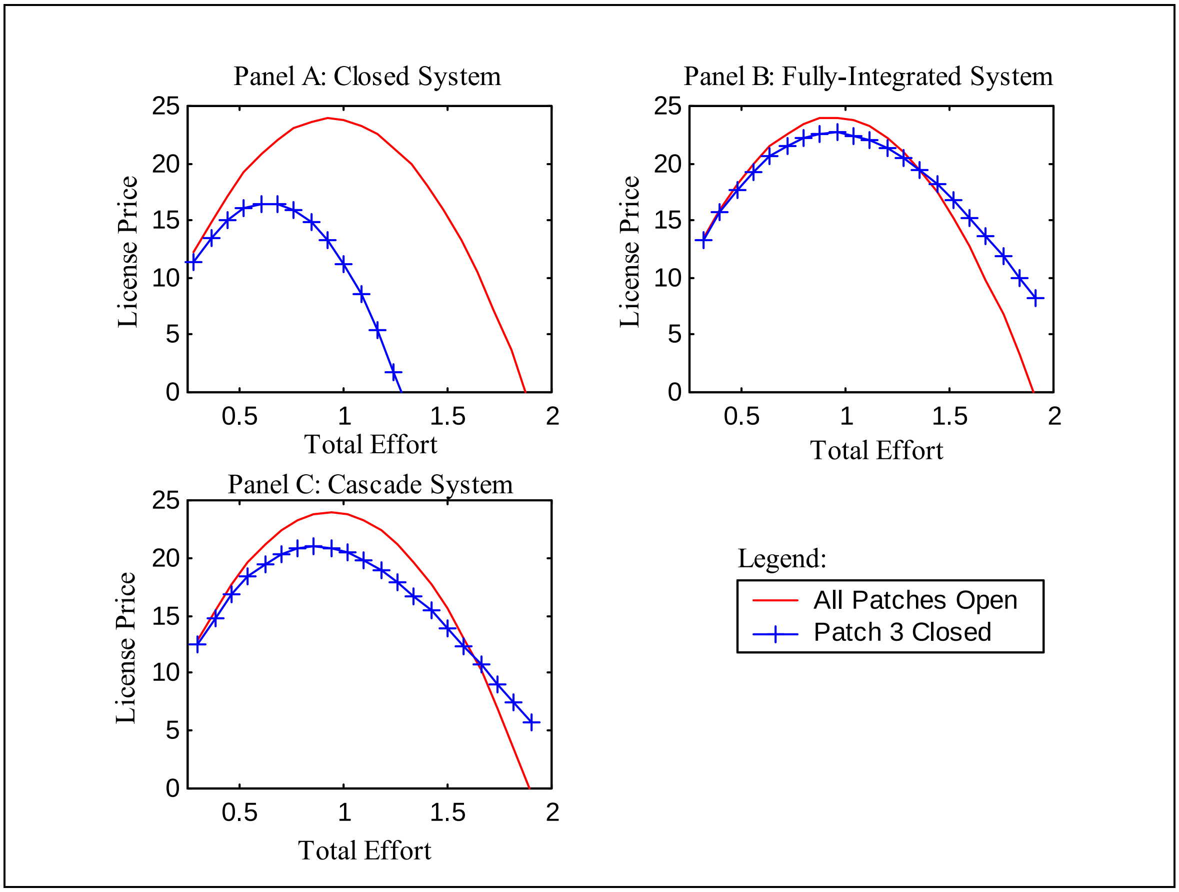

Figure 3: Maximum Aggregate Rents with and without a Reserve

Therefore, while locating a reserve on an edge of the system could yield positive economic and biological benefits, the magnitude of the benefits is less than what would arise when an area that is more connected to the rest of the system is set aside. This result stems from a spatial Le Chatelier effect, in which the cascade system is more constrained than the fully-integrated system but less constrained than the closed system. As Figures 2 and 3 illustrate, the expected effects of reserves depend not only on the dispersal rate, as many have noted, but also on the ecological structure of the system where reserves are sited.

Which Patch Should Be Closed?

It is clear that whether a reserve sited in a particular location or fishery will yield a win-win situation depends on the current biological and economic conditions. It is also clear that opposition to reserves by fishermen will be greatest in cases where they have the most to lose. Given the political-economy realities of siting reserves, a worthwhile exercise is to try to determine characteristics of patches where reserves are most likely to have the least objection.

In this section, we illustrate the impacts of siting reserves when there exists a patch that is more biologically productive and when there exists a lower cost patch. We focus on these two cases for a couple of reasons. First, many proponents of reserves treat the reserve selection issue as if it were one of simply finding and closing inherently high productivity areas. If this was the case, then we would expect that closing high productivity areas will yield a "win-win" scenario, or at least dominate (higher license prices) closing lower productivity areas, everything else equal. Second, under open-access conditions, Sanchirico and Wilen [2000] found that closing the most profitable patch provided the greatest chance for aggregate harvests to increase. This result was due to the fact that under open-access, the most profitable area is over-exploited (lower harvests) and shutting it down results in the lowest opportunity cost. In a limited-entry setting however, both the amount harvested and the net returns per unit of harvest in the fishery determine whether aggregate license prices might increase.

To investigate the impacts of closing high productivity areas and low cost areas, we simulate the closure of patch 3 in a fully-integrated system, but with a new assumption that the patch has a growth rate or cost 1.5 times greater than patches 1 and 2. We then compare the results to a closure in patch 2, which is assumed to have a lower growth rate and cost, everything else equal.[14] To keep the analysis simple, when we simulate the biological case, we assume that there exists no economic heterogeneity so that, in terms of the economic parameters, each patch is identical. And when we simulate the economic case, we assume that there exists no biological heterogeneity. This allows us to focus specifically on the questions at hand. Of course, in practice these conditions are most likely intertwined. For example, in many fisheries, there is anecdotal evidence that the areas with highest biologically productivity are also areas where the cost per unit of harvest is the lowest.

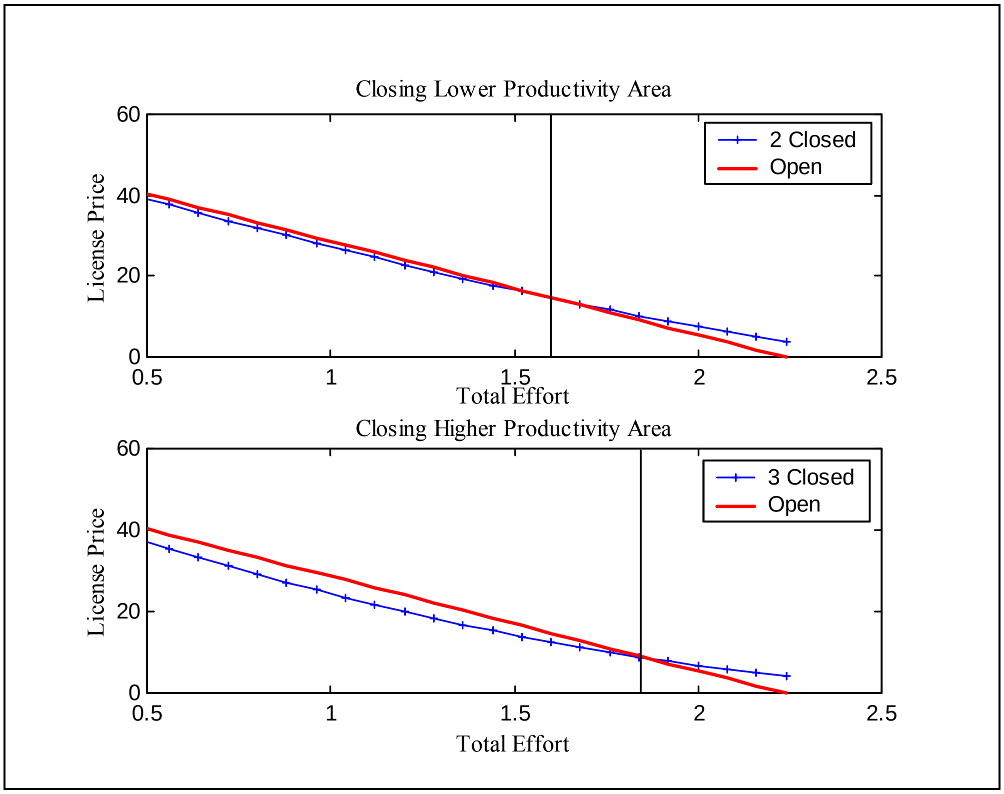

Figure 4: Closing High vs. Low Productivity Areas

The results on the impacts of closing high versus low productivity areas are presented in Figure 4. Under the chosen parameters, both cases result in win-win scenarios over ranges of total effort whereby the systems are severely overexploited. Interestingly, the range of effort levels over which this occurs is **larger** when the lower productivity area is closed. In fact, over the whole policy-relevant range of effort levels, the license price is higher when the lower productivity area is closed. Since license prices represent the value of a policy in terms of fishery rents, these results suggest that closing low productivity areas might provide greater returns to the fishermen than higher productivity areas, everything else equal. This result was also found in Sanchirico and Wilen [2000], and was attributed to the fact that high productivity areas provide the highest pre-reserve returns to fishermen and hence the highest opportunity costs of closures. As it turns out, in many cases similar biological gains can be accrued by closing off lower productivity areas, while at the same time lowering the opportunity costs of the closure to the fishermen.

As evident in Figure 5, the range over which the win-win occurs is larger when the higher cost area is closed than when the lower cost area is closed, everything else equal. Also, the differences between the two cases are smaller over the entire range of effort levels. We find, therefore, that it is no longer necessarily the case that closing the low cost area will yield a greater

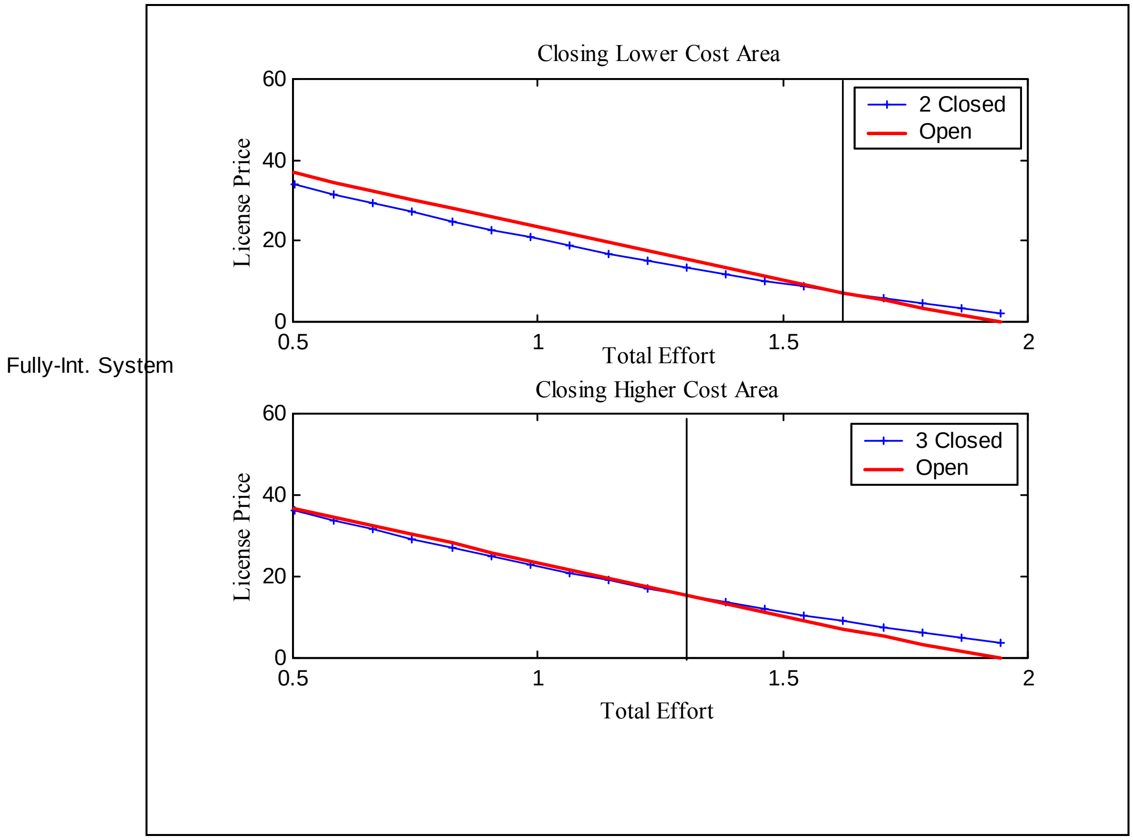

Figure 5: Closing High vs. Low Cost Areas

chance of a win-win situation.[15] In fact, in both cases, aggregate catches actually decrease under the parameters assumed. If total catches decrease, then how does the license price increase? Recall that the license price summarizes all of the available information in the fishery including ex-vessel prices, growth rates, harvest levels, and costs per unit of effort. In this case, the reserve shifts the spatial distribution of the catch to more profitable areas. The benefits of this shift, as measured by the license price, more than compensate for the loss in total catch, at least when total effort is in a neighborhood of open-access levels.[16]

These results suggest three important points. First, economic factors should be considered in siting decisions, if for no other reason than to reduce the opposition to sensible reserve plans. Second, some of the conventional wisdom that might be suggested from purely biological objectives (e.g., to close high productivity patches) may be reversed when one considers the economic costs to the industry of reserve siting. Third, if placed appropriately, reserves can improve the economic conditions of a fishery that is managed with non-spatially explicit instruments.

Conclusions and Discussion

This paper considers the manner in which the establishment of a marine reserve in a limited entry licensing system might affect fishermen. We utilize a simple metapopulation-based biological model and append a behavioral model of fishermen that hypothesizes spatial movement in response to rent differentials. We simulate a limited entry system in which there is a fixed amount of total effort that then distributes itself over space, determining an endogenous license price reflecting the shadow value of another unit of effort.

There are several qualitative conclusions that can be drawn from this simple analysis. First, license values are a gauge of aggregate system profits and hence, in principle, they can be used as a measure of the impact of reserves on the fishing industry. If a reserve is implemented that makes fishermen worse off in the aggregate, that outcome will be signaled with a drop in aggregate equilibrium license values and the reverse will happen when fishermen's incomes are improved. Given the almost infinite range of design options possible for a system of reserves, it seems politically expedient to look for options that have few negative or even positive impacts on fishing profits.

Second, we show that the nature of the spatial dispersal system is important to the success potential of reserves. Closed systems increase aggregate biomass, but they cannot increase aggregate catch because there is no larval dispersal from the reserves. Open systems with linkages fare better. Generally in open systems, a reserve decreases rents and causes license values to fall. However, there are some circumstances in which a reserve actually increases rents. These are when the initial pre-reserve equilibrium is close to the open-access equilibrium (such as when the limited entry program barely limits entry), and when the high cost patch is designated the reserve. Thus a "win-win" situation requires special biological and economic preconditions.

Third, we illustrate the intricate relationship between possible "win-win" scenarios and the location choice of the reserve. In systems where there exists biological heterogeneity, closing the lower productivity area reduces the opportunity cost of the closure to fishermen while still providing positive biological benefits. In open-access fisheries, earlier evidence suggested that closing the most profitable patch would likely produce win-win situations. In a limited-entry fishery however, both the amount harvested and the net returns per unit of harvest in the fishery determine whether aggregate license prices might increase. As a result, we find that closing the high cost area provides the most opportunity for a win-win result in a licensing limited entry setting.

Fourth, we also find that getting the most out of reserves may call for additional policies. In particular, in many cases it may be necessary to reduce the limited entry fleet by buy-back or other means to tailor the fleet to the new bioeconomic post-reserve conditions. Alternatively, regulators might consider implementing area licensing rather than fishery-wide licensing in order to further optimize effort, catches, and biomass distribution.

References

Allison, G. W., J. Lubchenco, and M.H. Carr (1998), *Marine Reserves are necessary but not sufficient for marine conservation*. Ecological Applications **8** (1): s79-s92.

Arnason, R. (2000), *Marine Reserves: Is there an economic justification?,* Paper presented at the International Conference on the Economics of Marine Protected Areas at the University of British Columbia Fisheries Centre, Vancouver, July.

Boersma, P. Dee and Julie K. Parrish (1999), Limiting abuse: marine protected areas, a limited solution, *Ecological Econ.* **31:** 287-304.

Carr, M.H. and D.C. Reed (1993), Conceptual Issues Relevant to Marine Harvest Refuges: Examples from Temperate Reef Fishes, *Canadian Journal of Fisheries and Aquatic Sciences* **50**: 2019-2028.

Carr, M. H. and P. Raimondi (1998), Concepts Relevant to the Design and Evaluation of Fishery Reserves. In *Marine Harvest Refugia for West Coast Rockfish: A Workshop,* edited by M. M. Yoklavich. Washington, DC: National Oceanic Atmospheric Administration, Technical Memorandum National Marine Fishery Service.

Dugan, J. E. and G. E. Davis (1993), Applications of Marine Refugia to Coastal Fisheries Management. *Canadian Journal of Fisheries and Aquatic Sciences* **50**: 2029-2042.

Hanneson, R. (2000), *The Economics of Marine Reserves,* Paper presented at the International Conference on the Economics of Marine Protected Areas at the University of British Columbia Fisheries Centre, Vancouver, July.

Hanski, I. (1999), *Metapopulation Ecology*. Oxford Series in Ecology and Evolution, Oxford Press, England.

Farrow, S. (1996), Marine Protected Areas: Emerging Economics. *Marine Policy* **20** (6): 439-446.

Gordon, H. S. (1954), *The Economic Theory of a Common-Property Resource: The Fishery*, Journal of Political Economy **62:** 124-142.

Hastings, A. and S. Harrison (1994), Metapopulation dynamics and genetics, *Annual Review of Ecology and Systematics* **25:** 167-188.

Levin, S. A. (1976), *Population dynamic models in heterogeneous environments*, Annual Review of Ecological Systems **7**: 287-310.

Palumbi, S. R. (1999), *The ecology of marine protected areas*, Working paper from the Department of Organismic and Evolutionary Biology, Harvard University, Massachusetts.

Pulliam, H. R. (1988), *Sources, Sinks, and population regulation*, The American Naturalist **132**: 652-661.

Roberts, C. M. and N.V.C. Polunin (1991), *Are Marine Reserves Effective in the Management of Reef Fisheries?*, Reviews in Fish Biology and Fisheries **1:** 65-91.

Sanchirico, J. N. (2000), *Marine Protected Areas as Fishery Policy: An Analysis of No-Take Zones*, Discussion Paper 00–23, Resources for the Future, Washington, DC.

Sanchirico, J. N. and J. E. Wilen (1999a), *Bioeconomics of Spatial Exploitation in a Patchy Environment*, Journal of Environmental Economics and Management **37**: 129-150.

Sanchirico, J. N. and J. E. Wilen (1999b), *Managing Renewable Resource Use With Market-Based Instruments: Matching Policy Scope to Ecosystem Scale*, Paper presented at the Association of Environmental and Resource Economists workshop on "Market Based Instruments", Boston, July.

Sanchirico, J. N. and J. E. Wilen (*forthcoming* 2000), *A Bioeconomic Model of Marine Reserve Creation*, Journal of Environmental Economics and Management.

Smith, V. L. (1968), *Economics of Production from Natural Resources*, American Economic Review 409-431.

Thomson, C. (1998). Evaluating Marine Harvest Refugia: An Economic Perspective, In *Marine Harvest Refugia for West Coast Rockfish: A Workshop,* edited by M. M. Yoklavich. Washington, DC: National Oceanic Atmospheric Administration, Technical Memorandum National Marine Fishery Service.

Tuck G. N. and H. P. Possingham (1994), *Optimal Harvesting Strategies for a Metapopulation*, Bulletin of Mathematical Biology **56**(1): 107-127.

Endnotes

[1] Note that we are taking a political economy perspective rather than a welfare economics perspective here. We are looking for circumstances in which it is likely that natural proponents of reserves (marine ecologists and managers) and natural opponents of reserves (fishermen) might find themselves in agreement over a particular reserve formation plan.

[2] The literature on the potential impacts of marine reserves is expanding rapidly, and an in-depth literature review is beyond the scope of this paper. Instead we point the interested reader to the following biological review articles: Davis and Dugan [1993]; Roberts and Polunin [1991]; Carr and Reed [1993]; Allison, Lubchenco and Carr [1998], Carr and Raimondi [1998]; Palumbi [1999], Boersma and Parrish [1999]; and the following economic review articles (Farrow [1996]; Thomson [1998]; Sanchirico [2000].

[3] See, for example, the review articles by Levin [1976]; Hastings and Harrison [1994]; and Hanski [1999].

[4] This lumped parameter representation is itself very stylized since it ignores important aspects of real population growth and dispersal dynamics including age- and size-specific mechanisms, selectivity issues, and more complicated spawner/recruit processes. However, it is analytically tractable whereas richer models must be analyzed using simulation methods.

[5] The literature on reserves also discusses other formulations that depict uni-directional flow, generally assumed to be the result of oceanographic processes such as currents, winds, and temperature. These models are often referred to as sink-source models (Pulliam [1988]; Tuck and Possingham[1994]), and they characterize dispersal flow as a process that is independent of population densities in the sinks.

[6] This will not be the optimal way to distribute effort over space, of course, since it is the outcome of a myopic, open-access process. It will also be the case that too much effort will be drawn into the whole system since effort will be responding to average and not marginal rents.

[7] For example, Sanchirico and Wilen [1999a] show how the spatial distribution of effort and the total amount of effort in an open-access system changes with different dispersal mechanisms, and with different kinds of biological and economic heterogeneity.

[8] Hannesson[2000] extends this framework to investigate the effects of reserve creation in the presence of random environmental shocks.

[9] As it is currently modeled, the license price is equivalent to a charge per unit of effort.

[10] This particularly simple structure is due in part to the fact that the rent function is linear in E. This makes marginal rents equal to average rents, which simplifies the nature of the rent-dissipating relationships.

[11] We assume that the biological and economic system parameters are unchanged after an area is set aside. This assumption might not hold in practice, however. For example, the intrinsic growth rate of the stock in the reserve might increase after the area is set aside due to the elimination of harmful harvesting practices. Another possibility is that the costs of harvesting in the open patches might increase due to an increase in congestion. While these circumstances can be incorporated here in a structural way, they are incorporated in a reduced form model by Arnason [2000]. In that paper, Arnason provides a complementary framework to the model developed here, which illustrates the biological and economic trade-off inherent in reducing the amount of fishable area.

[12] In the cases where the n-patch biological system is fully integrated, for example, the equation for the license price is a n+1 degree polynomial.

[13] In this case, the program for determining the level of aggregate effort that maximizes aggregate rents is

$$\max_{E^{TOT}} L^{ss}_{d_{ij}=0} E^{TOT} = \frac{p}{\sum_{i=1}^{n} \frac{r_i}{q_i^2}} [\sum_{i=1}^{n} E_i^{OA} - E^{TOT}] E^{TOT}$$

.

[14] Note that both closures are simulated with the following growth rates r3=1.5*r, r2=r1=r.

[15] In Sanchirico and Wilen [2000], a win-win situation occurred when both aggregate catches and biomass increased. Aggregate catches are used as indicators for the impact of reserves on the open-access fishery because there is no long-run economic change from reserves; rents are dissipated both before and after reserve creation.

[16] It is important to point out that the effort distribution with a common license price does not correspond in general to the optimal distribution. As a result, there is an opportunity for a reserve to shift effort towards a more "economically favorable" distribution than that which occurred prior to the reserve. In order to arrive at an optimal spatial distribution, policymakers would need to introduce spatially explicit license prices (Sanchirico and Wilen [1999b]). In this case, it would not be possible for a reserve to redistribute effort in a manner that would increase the license price.

MPAs: Process, Privilege and Participation: a sociological discussion

Victoria Silk

Simon Fraser University
E-mail: Victoria.silk@gems4.gov.bc.ca

Abstract

As we start the new millennium, it is apparent to all that fish resources and their habitats are in peril globally. The establishment of marine protected areas (MPAs) has gained a profile in fisheries management as having the potential to play a major role in the restoration of marine ecosystems. The current dialogue regarding their implementation includes a focus on scientific concerns, appropriate process, and potential long-term benefits of MPAs for fish ecosystems and humans. The consensus that human communities have an investment in healthy fish stocks begs discussion of the following: the importance of identifying and including different interest groups in the decision making process, determining both for whom fish stocks will be rebuilt and by what means they will be exploited, and revisiting notions of how we define the "experts". Access to the playing field of fisheries management has been the exclusive privilege of scientists and policy makers, giving these groups an obvious advantage when it comes to the construction of new knowledge systems. The inclusion of stakeholders is not a new idea in fisheries management, however there is still a notable absence of them in the decision-making arenas (Gerrard, 2000; Neis, 1992). It is important that policy makers, and the researchers who inform them, seek out and include less visible user groups. Their participation in the initial stages of planning MPAs should help facilitate balanced discussions on what future fisheries and their technologies within the designated areas will look like. The concept of reinventing fisheries management is gaining momentum (Pitcher, Hart, and Pauly, 1998). A component of this should include challenging the way we conceptualize and legitimize the "experts", thus ensuring that the inclusion of stakeholders results in more than just a physical presence in the negotiations. We need a new and inclusive respect for the different ways that people know the world of fish, one that will embrace participation, mentorship, co-authoring, and the interdisciplinary work of academics.

Introduction

"*dans le vrai*" the notion that positions formulated within an existing legitimated, discursive system, are more likely to be accepted or recognized as credible versus those offered from positions outside of "*le vrai*"

(Foucault, 1976, p 224)

The intention of this paper is threefold: 1) to highlight the need for both identification and inclusion of different user groups in the initial planning stages of MPAs, 2) to present preliminary ideas on what future activities and methods of exploitation within these areas will look like, and 3) to emphasize the importance of challenging our notions of "experts". The concerns being raised in this paper are of a sociological nature with a particular focus on issues relevant to those living in coastal communities that rely on small-scale fisheries. This discussion will be supported in part with illustrations from the Newfoundland fishery.

Identification and Inclusion of Stakeholders

Evidence suggests that marine protected areas have the potential to become an effective tool in resource management for the restoration of marine ecosystems (Sumaila, 1998; Weru, 1998). While it is recognized that not all species benefit from protected reserves, it has been established that over time there is generally an increase in species abundance and diversity within these areas (Weru, 1998). There is a growing body of literature that documents the effectiveness of MPAs on conservation of habitat and aiding with the recovery of overexploited species through the provision of a more efficient ecosystem based approach to management of coastal oceans (Anon., 2001).

We now see increased support for the implementation of MPAs from two of society's most powerful institutions: the international science community and national governments. The International Conference on the Economics of Marine Protected Areas, held at the University of British Columbia in July of 2000, which brought together academics from several countries to discuss MPAs, is one example of this. President Clinton's recent announcement of the Executive Order that called for the establishment of a comprehensive national program to provide greater protection of American waters reflects the growing concern of governments (news release, May 26, 2000). While this support has evolved out of a genuine commitment to effecting positive change in the condition of the world's marine environments, the track record to date of these two institutions with regards to fisheries management raises some concerns.

The scientific community, working on the premise that nature behaves in a predictable manner, has been guilty of assuming that there can be accuracy in fisheries management (Rose, 1996; Steele et al., 1992; Costanza et al., 1997), of offering incorrect advice on stock size (Walters, 1998; O'Boyle, 1993), and in some cases, of offering advice that reflects poor science (Steele et al., 1992). Governments, for their part, have based management decisions on economic models that favor industrial fisheries whose goal is accumulation of wealth, to the detriment of fishers involved in livelihood or subsistence harvesting (Davis, 1996). The last several decades have seen governments implement restrictive legislation that has resulted in loss of common access to, and privatization of, fishery resources (Kirby, 1982; Marchak, 1987; Matthews, 1993; Copes, 1998; Copes, 1999). Privatization in today's global economic environment has meant fewer companies with greater wealth and the ability to transcend national boundaries for the purposes of increasing profits (Kurien, 1995). The actions of these two institutions have contributed to the unfortunate state that now exists for both marine ecosystems and those who depend on fish resources. Those who have been most impacted by the collapse of fish stocks, have had the least input into the management schemes that determine their futures.

If all stakeholders are to have equitable representation of their interests through participatory action, they first need to be identified and this can prove to be a difficult process. There is a tendency to accept the myth of liberal democracy that states all voices are equal in society. Little attention is paid to the reality that some voices are significantly louder than others, thus having more access to, and clout within, the political and social arenas. While it may appear that different interest groups are being represented, upon closer examination, this is often not the case. O'Boyle (1993), in discussing the importance of defining membership of fishing groups, points out that while it is assumed that existing fishery associations represent the broader membership, often the protagonists in fisheries debates are discovered to be representing their own interests rather than those of the broader membership. An example of questionable representation by an officially recognized and legitimated organization of fishers, could be seen with the Fish Food and Allied Workers Union (FFAW), the official union of all fishers in Newfoundland. During the mid-1980s when fish stocks were perceived by many to be in great danger due to the serious offshore exploitation, inshore fishers attempted to establish a separate inshore union, believing that it was not possible for the FFAW to effectively represent the conflicting interests of the offshore and inshore fishers. While the attempt to join forces with a new union was unsuccessful, the FFAW was obliged to recognize the concerns of the inshore sector, and an inshore local was formed in the aftermath of the challenge.

The identification of stakeholders does not always guarantee that their concerns will be addressed in a meaningful way. For example, there is a tendency for policy advisers to view the fishing industry as a predominately male occupation (Rowe, 1991) in spite of extensive documentation that highlights women's involvement in fisheries (Gerrard and Groenbach, 1987; Caddigan, 1991; Antler, 1977; Silk, 1995). The refusal to acknowledge women in their roles as harvesters, processors, fish sellers or for their familial support services, has led to biased policies that promote gender inequalities (Rowe, 1991; Wright, 1992; Neis, 1993). The fishing industry globally is comprised of men, women, and children in extended families and communities that engage in harvesting, marketing, gleaming, processing, preparation for home consumption and administrative work (Fraga, 2000; Acejas et al., 2000).

Future activities and methods of exploitation

If one of the intentions of developing MPAs is to restore stocks for commercial consumption, there needs to be a discussion regarding what future activities and methods of exploitation will take place within them. In order to put a face on the problem of fisheries degradation from a sociological perspective, one needs to look at who has taken control of the resource, what they have done with that privilege, and how traditional stakeholders have been affected. Globally there are 1000s of coastal communities with a historical attachment to near shore fish resources whose catches have always been dependent on natural cycles. The migration of Canada's east coast cod and capelin stocks from the offshore to the inshore, or the hydrological seasons in the Gulf of Guinea that generate fluctuations in species assemblages (Binet and Marchal, 1993) are two examples of this. Severe exploitation of fish stocks has led to the situation where these natural cycles are no longer producing viable fisheries for coastal communities. Given the tenuous state of both marine ecosystems and coastal communities, the importance of determining who will be the future benefactors of restored stocks, and through what activities and methods of exploitation, is self-evident.

One component of this requires an in-depth analysis of why certain frequently cited assumptions remain relatively unchallenged within fisheries management despite the fact that many well-known and respected scholars have critiqued them. Gordon's theory of common property (1954), expanded on by Hardin (1968), is one of them. This theory focuses on the perceived inability of individuals and communities to effectively manage the 'commons' and the belief that there are too many fishers chasing too few fish (for an extended critique of Hardin's theory see McCay and Acheson, 1987; Matthews, 1993; Fairlie et al., 1995). With reference to Canada, Marchuk (1987) presents a convincing argument that suggests the tragedy of the commons might be more aptly described as "the tragedy of mismanaged state property" (p. 5) given that the present failure of fisheries has evolved during a time of extensive government regulation and restriction.

Overcapitalization is another frequently cited cause of problems (Sissenwine and Rosenberg, 1993), however the question is rarely posed in terms of overcapitalization of what sectors, and by whom? The drawbacks of subsidization of offshore fleets and privatization schemes such as ITQs have been critiqued by many (Marchuk, 1987; Sinclair, 1988; Davis, 1996; Copes, 1998; Copes, 1999), yet they continue to stand in the forefront as justifications for a fishery that can take credit for the destruction of marine resources globally (Safina, 1995). These include the wholesale strip-mining of resources by inappropriate technologies, predominately owned by transnational companies who have no loyalties to local communities.

There is also the myth of economic efficiency, a term that is invoked by governments as a rationale to justify offshore industrial technologies that employ fewer numbers of fishers (Storey, 1993). The technologies being used in commercial fisheries were initially developed with the intention of providing better access to unexploited stocks on a year round basis (Kodera, 1971). They have continued to evolve to the point where we now see purse seiners that catch thousands of kilograms of fish in a single haul, draggers with carrying capacities upwards of hundreds of metric tons of fish (Kodera, 1971), monofilament gillnets that when lost at sea become ghost nets which can continue to fish for years (Martin, 1997), fish finding sonar that can track fish anywhere, and deep water trawls that are able to exploit fish at increasingly greater depths (Junquera et al., 1992). While there have always been natural fluctuations in stock abundance (Sherman et al., 1993), there is now the ability, in conjunction with other anthropogenic pressures, to drive stocks worldwide into extinction. Economic efficiency is part of a powerful ideology used to justify on-going state support for corporate fisheries.

The activities of industrial fishing have led to a situation that is described by Pauly et al. (1998) as "fishing down the food web". This refers to the pattern of systematically fishing our way through the piscivorous stocks of higher trophic levels, and then turning our efforts to smaller invertebrates of the lower trophic levels. The ecological impact is the degradation of marine resources, and there is a concern about whether stocks of the larger fishes can be rebuilt when their primary food sources are being commercially fished to the brink of collapse.

This pattern of exploitation has serious social consequences in addition to the ecological ones. Complex social relations and inequities further exacerbate the loss of income that results from a shortage of fish. Using Newfoundland outport communities as an example, one can witness a concentration of benefits through limited licencing schemes that result in a small elite of fishers accessing high value stocks such as crab, shrimp and lobster.[1] The shift to commercial exploitation of species such as shellfish sees fewer workers employed due to the highly mechanized nature of the fishery. In Catalina, Newfoundland, shrimp plant workers have decreased in numbers from twelve hundred to one hundred and fifty as a result of mechanization (Dr. Barb Neis, personal communication). Looking at the industry from a global perspective, shrimp is a product that can be harvested in one part of the world, and then

[1] I witnessed this process first hand both as a full-time commercial fisher in Petty Harbor, Nfld. and also as a member of the Atlantic Fisheries Licencing Appeal Board in the mid-1980s.

shipped to another country in order to access a cheap labor pool. Once it is processed and packaged, it is then returned to first world markets where profits can be maximized. The creation of a large niche in first world markets for shrimp, has fueled third world fish farming that displaces people from their land, destroys mangrove swamps (Acejas et al., 2000), and ignores poor working conditions for plant workers both in developed and in developing countries (Diaz, 2000; Ramos et al., 2000).

Those who have the socially endorsed privilege to both legitimate and disseminate new knowledge need to be aware of the pervasive bias that continues to benefit the corporate elite within the fishing world. Globalization enables the exploitation of fish stocks worldwide by large industrial factory freezer trawlers that Kurien (1995) describes as global predators. The end result is fragmentation of communities and of lifestyles. Developing countries such as India have seen thousands of displaced workers in artisanal fisheries lose the ability to feed themselves and their families (Kurien, 1995). This raises the question: where is the viability of an industrial offshore fishery that requires unprecedented millions of dollars in government subsidies in order to stay afloat (Sutton, 1998), while using technologies which destroy stocks, ecosystems, untargeted species, and ocean bottoms? At this point in time, fish resources can either be placed back in the hands of coastal communities that have historical resource attachments to them or, the trend towards privatization that has seen an increasing transfer of fishing rights to corporate ownership can continue. MPAs may contribute to the restoration of degraded marine ecosystems however, implementation of them could result in final closure of the commons if consideration is not given to whom the future benefactors will be and acceptable models of exploitation.

Redefining the Experts

The recognized experts in fisheries management do not have a particularly notable track record to date, which suggests that it is time to reevaluate how we define "expert". While it is not uncommon to encounter reference to the concept of traditional ecological knowledge (TEK) and acknowledgement that it represents a specific way of viewing the world (Neis, 1992; Salas et al., 1998), there does not appear to be wide spread support within management regimes for recognition of TEK as an equally legitimate source of data to be used for management purposes. Traditional knowledge differs from the scientific method of research in that it is a highly deductive, experiential way of creating a working knowledge base. In contrast, the epistemological method of Western science research involves objectivity, isolation of minutiae, and intense observation of discrete pieces of the whole (Harding, 1986; Stanley and Wise, 1990; Fee, 1983). Examples of Western research would be single species stock assessment; fish science studies on fish fecundity or, the study of a single haemoglobin in an arctic fish (Kunzmann et al., 1992). From a sociological perspective, the problem that arises with research that derives from objective, quantitative methodologies is the risk of portraying a world view which is highly abstract in light of day to day lived realities. This kind of abstraction can produce a somewhat skewed vision of what actually takes place in the communities of people who are impacted by the loss of fishing, and the implementation of new resource management schemes.

Another aspect of this debate is to question who are the benefactors of research funding in the wake of resource failures. The destruction of Canada's East Coast cod stocks resulted in an influx of world-renowned fisheries scientists and academics, and likewise, generous amounts of grant money to study everything from the dynamics of cold oceans and seal diets, to analyzing the social impacts of the economic collapse of single industry towns. This can be witnessed in the plethora of research that has focused on the East Coast fishery, most of which has been funded by government grants. Much of the research that was conducted in the aftermath of the Grand Banks collapse, confirmed what fishers had been arguing for years, yet their concerns were not taken seriously until they were legitimated by researchers and policy advisors.

The question of where the research dollars go was raised recently at the Gender, Globalization and Fisheries conference, held in Salmoner, Newfoundland (May, 2000). This conference brought together an eclectic group of people that included fish workers from Atlantic Canada, researchers who work with NGOs and academics from eighteen countries. Many of the non-academic participants voiced concerns about the lack of responsible research and the unwillingness of academics to acknowledge the contributions of community expertise to the research process. Likewise, many of the fish workers felt that researchers in general fail to acknowledge the contributions of local knowledge or provide appropriate feed back of research results. They expressed the desire that some of the funding that goes to academics be directed towards communities that may have the ability and expertise to do their own brainstorming on solutions.

The proceedings of the International Coral Reef Regional Symposium (Anon., 2000) confirm that this problem is a global one. In a lengthy discussion on MPAs, two issues of importance that are often overlooked were identified. One is that local people need to be provided with the opportunity to take a leadership role in making decisions, and the other is that researchers often neglect to provide crucial feedback to the communities in which they conduct their research. A recent Canadian study has identified that there are no effective policies in place to enable community participation in the creation of MPAs (Wallace and Boyd, 2000).

Conclusion

Critical to this discussion is the fact that there is no consensus on what does or doesn't work within fisheries management. Neither is there consensus on the costs and benefits of large scale versus small-scale fisheries, on the ethics of privatization of resources, nor the analysis of the impacts of globalization. While there is considerable debate regarding the conflict between the demands of corporate interests to maximize their profits, and the need for communities to retain employment (O'Boyle, 1993), the fact is that decisions in management have been weighted very much in favor of corporate interests (Matthews, 1995). If the process of designing and implementing new management schemes proceeds without appropriate consultation, and does not give equal weight to determinations of for whom, and by what means, resources will be accessed, then what little power and control coastal communities are presently struggling to maintain, will be seriously threatened.

Researchers and managers are privileged to actively affect the knowledge production process. Given this, they have a moral and ethical responsibility to be aware of the repercussions of their work. With the creation of MPAs, this means being aware of who is excluded, whose voices aren't being heard, and who stands to lose. There needs to be assurance that, should fish stocks be revitalized, appropriate methods of exploitation in the right hands will occur. It is a task only half done to figure out if the fish can be brought back, the other half is determining for whom the fish will be on reserve. Learning to create decision-making forums that encourage the broadest participation will be the key to successful results. Different ways that this can be accomplished are through conferences and workshops that encourage participation of those who are outside of traditional academic disciplines. The failure of centralized management regimes to protect fish stocks (Brown, 1998) has led to pressure from community groups to take back some control over resource management. Co-management and community management schemes are starting to show promise as new measures that can bridge the gap between policy makers and those involved in fisheries (McCay, 1989). Co-authorship, mentoring, interdisciplinary projects and respect for traditional ecological knowledge can also provide a platform for a broad range of concerns. We need to brainstorm ideas for discussion that will help ensure that the implementation of MPAs will proceed as the result of a most thorough, thoughtful, and inclusive process that will have given recognition to the concerns of all who stand to be impacted.

References

Acejas, A., Heinan, A., Quist, C., and J. Fraser. (2000). "I Leave You The Fish". Unpublished paper. PIPULI, Mindanao, Philippines.

Anon. (2001). Marine Protected Areas: Tools for Sustaining Ocean Ecosystems. Washington, D.C.: National Academy Press.

Anon. (2000). International Coral Reef Initiative Regional Symposium. Coral Reefs in the Pacific: Status and monitoring; Resources and management. Workshop Report, May 23, 2000, The Institute for Research in Development (IRD) and Secretariat of the Pacific Community (SPC), Noumea, New Caledonia.

Antler, E. (1977). "Women's Work in Newfoundland Families". Atlantis: A Women's Studies Journal: 106-113.

Binet, D. and E. Marchal. (1993). The Large Marine Ecosystem of Shelf Areas in the Gulf of Guinea: Long-Term Variability Induced by Climatic Changes. In Large Marine Ecosystems: Stress, Mitigation, and Sustainability, eds. K. Sherman, L.M. Alexander, and B.D. Gold, 104-138. Washington, DC: AAAS Publications.

Brown, R.C. (1998). Community-based cooperative management: renewed interest in an old paradigm. In Reinventing Fisheries Management, eds. T.J. Pitcher and D. Pauly, 185-193. London: Kluwer Academic Publishers.

Cadigan, S. (1991). Unpublished Ph.D. thesis. St. John's, Nfld.: Memorial University.

Copes, P. (1998). Adverse Impacts of Individual Quota Systems in Small-Scale Fisheries: What are the Positive Alternatives? In Managing Our Fisheries, Managing Ourselves, eds. L. Loucks, A.T. Charles, and M. Butler, 23-30. Proceedings of symposuim on Fisheries that work: New directions in the North Atlantic.

Copes, P. (1999). Common-property fishing rights: Coastal Resources for whom? Samudra Report, 23: 14-19.

Costanza, R., J. Cumberland, H. Daly, R. Goodland, and R. Nargaard. eds. (1997). Ecological Economics. Florida, USA: CRC Press LLC.

Davis, A. (1996). Barbed wire and bandwagons: a comment on ITQ fisheries management. Reviews in Fish Biology and

Fisheries 6: 97-107.
Diaz, E.A. (2000). Chile*:* Trabajadoras de plantas pesqueras, riesgos laborales en sus puestos de trabajo. Unpublished paper. Santiago, Chile.
Fairlie, S., M. Hagler, and B. O'Riordan, (1995). The Politics of Overfishing. The Ecologist 25, 2/3, 46-73.
Fee, E. (1983). Women's Nature and Scientific Objectivity. In Woman's Nature, eds. M. Lowe, and R. Hubbard, 9-28. USA: Paragon Press..
Foucault, M. (1976). The Archeology of Knowledge. New York: Pantheon Books.
Fraga, J. (2000). Conservation Policy and Coastal Management in Yucatan, Mexico. Unpublished paper. Department of Human Ecology, Center for Research and Advanced Studies, I.P.N., Mexico.
Gerrard, S. and D. Gronbech, eds. (1987). Women in the Norwegian Fisheries. Norway: Finmark Regional College.
Gerrard, S. (2000). Collaborative Partners in Small-Scale, Global Fishery Research: Reflections and Challenges. Unpublished paper. Institute of Planning and Community Studies, Faculty of Social Science, University of Tromso, Norway.
Gordon, H.S. (1954). The Economic Theory of a Common-Property Resource: The Fishery. Journal of Political Economy 62 (2): 124-142.
Hardin, G. (1968). "The Tragedy of the Commons". Science 162: 1243-1248
Harding, S. (1986). The Science Question in Feminism. Ithaca, NY: Cornell University Press.
Junquera, S., S. Iglesias,. and E. de Cardenas. (1992). Spanish fishery of Greenland halibut (*Reinhardtius hippoglossoides*). In 1990-91 NAFO SCR document 92/28.
Kirby, M.J.L. (1982). Navigating troubled waters. A new policy for the Atlantic Fisheries. Supply & Services, Ottawa.
Kodera, K. (1971). Aimed Bottom and Midwater Trawling Techniques of Japanese Factory Stern Trawlers. Fish finding, purse seining and aimed trawling. In Modern Fishing Gear of the World 3*,* ed. Hilmar Kristjonsson, 411-419. London, England: Fishing News (Books) Ltd.
Kunzmann, A., A. Fago, R. D'Avino, et al. (1992). Haematological Studies on *Aethotaxis mitopteryx DeWitt*, a high-Antarctic fish with a single haemoglobin. Polar Biology 12: 141-145.
Kurien, J. (1995). Joint Action Against Joint Ventures. The Ecologist 25, 2/3: 115-119.
Marchak, P. (1987). Uncommon Property. Canada: Methuen Publications.
Martin, B. (1998). A fisherman's tale. Our Planet 9 (5): 25-26.
Matthews, D.R. (1993). Controlling Common Property: Regulating Canada's East Coast Fishery. Toronto: University of Toronto Press.
Matthews, D.R. (1995). Commons versus Open Access: The Collapse of Canada's East Coast Fishery. The Ecologist 25: 2/3: 86-104.
McCay, B.J. and J.M. Acheson, eds. (1987). The Question of the Commons: The Culture and Ecology of Communal Resources. Tucson: University of Arizona Press.
McCay, B.J. (1989). Co-Management of a Clam Revitalization Project: The New Jersey "Spawner Sanctuary" Program. In Cooperative Management of Local Fisheries: New Directions for Improved Management and Community Development, ed. E. Pinkerton, 103-124. Vancouver: University of British Columbia Press.
Neis, B. (1992). "Fishers Ecological Knowledge and Stock Assessment in Newfoundland," Newfoundland Studies 8(2): 155-178.
Neis, B. (1993). Context Paper on Fisheries. Unpublished paper. Department of Sociology, Memorial University, St. John's Newfoundland.
O'Boyle, R. (1993). Fisheries Management Organizations: A Study in Uncertainty. Fishery Aquatic Science 120: 423-436.
Pauly, D., V. Christensen, E. Froese, and E. Torres. (1998). Fishing down marine food webs. Science 279: 860-863.
Pitcher, T.J., P. Hart, P., and D. Pauly, eds. (1998). Reinventing Fisheries Management. London: Kluwer Academic Publishers.
Ramos, A.B., E.F. Rosel, and G. Rodriguez. (2000). Community Management and Integrated Management in the Coastal Zones, A Cuban Example. Unpublished paper. University De Pinar Del Rio, Cuba.
Rose, G. (1997). The trouble with fisheries science! Reviews in Fish Biology and Fisheries 7: 365-370.
Rowe, A. (1991). Effect of the Crisis in the Newfoundland Fishery on Women Who Work in the Industry. St. John's, Nfld.: Women's Policy Office, Government of Newfoundland and Labrador.
Safina, C. (1990). The World's Imperiled Fish. Scientific American 273 (5): 46-53.
Salas, S., J.A. Archibald, and N. Haggan. (1998). Aboriginal Knowledge and Ecosystem Reconstruction. In Back to the Future: Reconstructing the Strait of Georgia Ecosystem. Fisheries Centre Research Reports, 6 (5): 22-28.
Sherman, K., L.M. Alexander, and B.D. Gold. (1993). Large Marine Ecosystems. Washington, DC: AAAS Publications.
Silk, V. (1995). Women and the Fishery. In Their Lives and Times. eds C. McGrath, B. Neis, and M. Porter, 264-269. St. John's, Nfld: Killick Press.
Sinclair, P.R. (1988). A Question of Survival. St. John's, Newfoundland: Institute of Social and Economic Research.
Sissenwine, M.P. and A.A. Rosenberg. (1993). Marine Fisheries at a Critical Juncture. Fisheries 18 (10): 6-14.
Stanley, L. and S. Wise. (1990). Method, Methodology and Epistemology in Feminist Research Processes. In Feminist Praxis: Research, Theory and Epistemology in Feminist Sociology*,* ed. L. Stanley, 20-60 London: Routledge.
Steele, D.H., R. Andersen, and J.M. Green. (1992). The Managed Commercial Annihilation of Northern Cod. Newfoundland Studies 8 (1): 34-68.
Storey, K. ed. (1993). Defining the Reality. St. John's, Newfoundland: Institute for Social and Economic Development.
Sumaila, U.R. (1998). Protected marine reserves as hedges against uncertainty: an economist's perspective. In Reinventing Fisheries Management, eds. T.J. Pitcher, and D. Pauly, 303-309. London: Kluwer Academic Publishers.
Sutton, M. (1998). Harnessing market forces and consumer power in favor of sustainable fisheries. In Reinventing Fisheries Management, eds. T.J. Pitcher, P. Hart, and D. Pauly, 125-135. London: Kluwer Academic Publications.
Tisdall, S. "Canadians overdraw on the Grand Banks," Guardian Weekly, 17 October 1993, International News, p. 7.
Wallace, S. and D.R. Boyd. (2000). Out of Sight, Out of Mind and Almost Out of Time, a brief to the Sierra Club of British Columbia. April, 2000.
Walters, C. (1998). Designing fisheries management systems that do not depend upon accurate stock assessment, In Reinventing Fisheries Management. eds. T.J. Pitcher, P. Hart, and D. Pauly, 279-288. London: Kluwer Academic Publishers.
Weru, S.D.M. (1998). Marine Conservation in Kenya. In Large Marine Ecosystems of the Indian Ocean: Assessment, Sustainability, and Management. eds. K. Sherman, E.N. Okemwa, and M.J. Ntiba, 353-359. USA: Blackwell Science Inc.
Wright, M. (1992). The Smile of Modernity: The State and the Modernization of the Canada Atlantic Fishery 1945-1970. Unpublished thesis, Queen's University.

MARINE PROTECTED AREA PERFORMANCE IN A GAME THEORETIC MODEL OF THE FISHERY

Ussif Rashid Sumaila

Fisheries Centre, UBC, Vancouver, Canada and Chr. Michelsen Institute, Bergen, Norway.

Abstract
What bio-economic benefits can be expected from the implementation of marine protected areas (MPAs) in a fishery facing a shock in the form of recruitment failure, and managed cooperatively compared to non-cooperatively? What are the optimal sizes of MPAs under cooperation and non-cooperation? I explore these questions in the current paper by developing a computational game theoretic model, which incorporates MPAs using the North East Atlantic cod fishery as an example. Results from the study indicate that MPAs can protect the discounted economic rent from the fishery if the habitat is likely to face a shock, and fishers have a high discount rate. The total standing biomass increase with increasing MPA size but only up to a point. The study also shows that the economically optimal size of MPA for cod varies between 50 – 70% depending on (i) the exchange rate between the protected and unprotected areas of the habitat; (ii) whether fishers behalf cooperatively or non-cooperatively; and (iii) the severity of the shock that the ecosystem may face.

1. Introduction

Marine protected areas (MPAs) are parts of the marine habitat in which fishing is controlled or prohibited entirely for all or part of the time [see Bohnsack 1990 and Sumaila et al. 2000]. The interest in MPAs as a tool for fisheries and ecosystem management has now gone past marine researchers and conservation groups to policy makers. Evidence of this is the May 2000 Executive Order issued by the President of the USA calling for "appropriate actions to enhance or expand protection of existing MPAs and establish or recommend, as appropriate, new MPAs"[1].

Among the groundwork recommended to guide how to go about implementing the Executive Order is the "assessment of the economic effects of the preferred management solution".[2] The objective of this paper is precisely to provide an assessment of the economic performance of MPAs: Will the establishment of an MPA bring about significant biological and economic benefits if the management objective is to maximize the joint profits of fishers? What sizes of MPAs may be considered optimal under cooperation and non-cooperation?

Published economic models that study the potential economic benefits of MPAs can be group into (i) single species/non-spatial/single agent (sole owner) models, for example, Holland and Brazee [1996], Hannesson [1998] and Sumaila [1998]; (ii) single species/spatial/single agent models, e.g., Holland [1998], Sanchirico and Wilen [1999]; (iii) multispecies or ecosystem/spatial/single agent models, for instance, Walters [2000] and Pitcher et al. [2000]; and (iv) multispecies or ecosystem/non-spatial/single agent, e.g., Sumaila [1998]. To my knowledge, there are no multi-agent models that explore the economic potentials of MPAs in the literature. The current paper fills this gap by developing a two-agent game theoretic model for the assessment of MPA performance. With a two-agent model, 1 address an important question, which until now has not been addressed in the literature, namely, how will MPAs perform when participants in a fishery cooperate, resulting in efficient management versus when they do not cooperate, leading to competitive and wasteful management.

The North-East Atlantic cod fishery is used to demonstrate the workings of the model developed. This cod stock is highly migratory, working its way through both Norwegian and Russian Exclusive Economic Zones (EEZs), as well as international waters. Norway and Russia together determine the total allowable catch (TAC), giving each country approximately 45% of the TAC, with the remainder harvested by other countries, such as Iceland, the Faroe Islands and some EU countries. The Russian and other-country catch is mainly harvested by trawlers offshore, while the Norwegian share of the TAC is divided between two vessel groups; trawlers and coastal vessels [see Armstrong and Sumaila 2000]. Thus, the fishery is presently managed cooperatively [see Nakken et al. 1996], which makes the current model relevant for studying the fishery.

I present the model in the next section. The results of the study are given in section 3, while the concluding remarks are presented in section 4.

2. The Model

Biological aspects

Let recruitment of age 0 fish to the whole habitat in period t $(t=1..T)$, R_t, be represented by the following Beverton-Holt recruitment function.[3]

$$R_t(B_{t-1}) = \frac{\alpha B_{t-1}}{1+\gamma B_{t-1}} \qquad (1)$$

where $B_{t-1} = \sum_{a=1}^{A} p_a w_{s,a} n_{a,t-1}$ represents the post-catch spawning biomass of fish; p_a is the proportion of mature fish of age a $(a=1..A)$; $w_{s,a}$ is the weight at spawning of fish of age a; $n_{a,t-1}$ is the post-catch number of age a fish in period t-1; and α and γ are constant biological parameters. The α and γ values determine the recruitment for a given spawning biomass, which again determines the pristine stock level.

Initially, it is assumed that the stock and recruits are homogeneously distributed, and randomly dispersed at a constant density. The fish population is split into two distinct components, i = 1,2 where 1 and 2 denote the protected and unprotected areas, respectively. There is net movement from the protected to the unprotected area, due to fish density being high relative to the carrying capacity in the protected section of the habitat (see the Basin model, MacCall, 1990). This movement is captured by the *net migration rate*, which tells us the net proportion of a given age group of fish that is transferred from the protected to the unprotected area in a given fishing period.

The division of the habitat is done by first, dividing the initial stock size between the protected and unprotected areas in proportion to these areas' respective sizes. Hence, an MPA consisting of 20% of the habitat, results in a split of the initial stock size into a 2:8 ratio in favor of the unprotected area. Second, it is assumed that recruitment takes place separately in the two areas defined as in equation 1 above, each area with its own B^i_{t-1} and γ^i, i=1,2. The α parameter, being an intrinsic element of the stock under consideration, is kept equal for fish both in the reserve and in the fished area. Finally, the respective γ parameters are set such that (i) the sum of recruitment from both areas satisfies

$$R^1_t + R^2_t = R_t \quad \text{for } B^1_{t-1} + B^2_{t-1} = B_{t-1} \qquad (2)$$

and (ii) the recruitment into the protected and unprotected areas is directly related to the quantity of the biomass in them. These conditions are enforced by giving γ^i values from 1 to 10 depending on the MPA size, with a value of 1 depicting a large MPA and a value 10 depicting small MPA.

For the protected area, the stock dynamics in numbers, $n^1_{a,t}$, is described by

$$n^1_{0,t} = R^1_t, \qquad (3)$$

$$n^1_{a,t} + \psi n^1_{a,t} = s n^1_{a-1,t-1}, \qquad \text{for } 0 < a < A,$$

$$n^1_{A,t} + \psi n^1_{A,t} = s\left(n^1_{A-1,t-1} + n^1_{A,t-1}\right), \quad n^1_{a,0} \quad \textit{given.}$$

where the parameter s is the age independent natural survival probability of cod; $\psi n^1_{a,t}$ is the net migration of age a (where A is the last age group) cod from the protected to the unprotected area in period t, and ψ is as defined earlier; $n^1_{a,0}$ denotes the initial number of age a cod in the protected area. Recollect that there is no harvesting in the protected area.
The stock dynamics in the unprotected area are expressed as

$$n^2_{0,t} = R^2_t, \qquad (4)$$

$$n^2_{a,t} + h^2_{a,t} = s n^2_{a-1,t-1} + \psi n^1_{a,t}, \qquad \text{for } 0 < a < A,$$

$$n^2_{A,t} + h^2_{A,t} = s\left(n^2_{A-1,t-1} + n^2_{A,t-1}\right) + \psi n^1_{A,t}, \quad n^2_{a,0} \quad \textit{given}$$

where $h^2_{a,t}$ is the total harvest function.
The total harvest is defined in the traditional way as

$$h^2_{a,t} = q_a n^2_{a,t} e_t$$

where q_a is the age dependent catchability coefficient, e_t, is the effort employed in the harvesting of cod in period t.

I introduce a shock in the natural system (see Sumaila, 1998) by incorporating a recruitment failure (zero recruitment) that occurs in each of the years 5 to 15 of the 28 year-time horizon model. It is important to note that the shock is assumed to occur only in the fished area, an assumption which follows Lauck (1996), where it is assumed that true uncertainty occurs due to human intervention in the natural environment, leading to over-fishing and habitat degradation. Sensitivity analysis is performed to study the effects of changes in the degree of shock and the exchange rate.

Economic aspects

A dynamic game theoretic model is applied to describe the cooperative and non-cooperative management of the Northeast Atlantic cod fishery in which there are two participants, namely, the coastal vessel group (*cf*) and the trawler gear group (*tf*). These are the two main vessel types used to harvest cod. The single period profit from harvesting fish, $\prod_m(.)$, is defined as

$$\prod_m(n^2,e) = v\sum_{a=0}^{A} w_a q_a n_{a,t}^2 e_t - \frac{k}{1+b}(e_t)^{1+b} \tag{5}$$

where m= *cf, tf* (*cf* stands for coastal fleet, and *tf* is the trawler vessel group)[4]. The variable e_t (t = 1,2,..,T=28) denotes the profile of effort levels employed by the particular player; n^2 is the age and time dependent stock size matrix in the fished area; v is the price per unit weight of cod; w_a is the average weight of age a cod; k is a cost parameter, and $b>0$ is a parameter introduced to ensure strict concavity in the model, which is required to ensure convergence (see Flåm, 1993 and Sumaila, 1997).

I assume that under cooperation, the objective of the participants in the fishery is to find a sequence of total effort levels, e_t (t = 1,2,..,T=28) that would maximize their joint benefits. Using the effort level as the control variable, the vessel groups jointly maximize their present value of profit, *Prof*

$$Prof_j = \sum_{t=1}^{T} \delta_j^t \prod_{j,t} \tag{6}$$

where $\delta = (1+r)^{-1}$ is the discount factor, and r denotes the interest rate. The optimization is carried out for given sizes of the MPA, subject to equation (2), (3) and (4), and the obvious non-negativity constraints.

Under non-cooperation, I assume each agent wishes to maximize own profits, that is, $\prod_{cf}$ and $\prod_{tf}$, respectively, for the coastal and trawler fleets. The non-cooperating agents must therefore choose their own effort levels in each fishing period in order to maximize own discounted profit, given that the other agent does the same. This is done without regard to the consequences of their own actions on the other agent's payoff. For the coastal fleet this translates into choosing own effort level to maximize

$$Prof_{cf} = \sum_{t=1}^{T} \delta_{cf}^t \prod_{cf,t} \tag{7}$$

Modified Lagrangian functions in the sense of Flåm (1993) and Sumaila (1997) are set up and computed using the simulation package known as Powersim. The computational procedure is resorted to because it is difficult to try to solve the current multi-cohort model analytically (see Conrad and Clark 1987).

The solution procedure (algorithm) is from nonsmooth convex optimization, in particular, subgradient projection and proximal-point procedures (see for example, Flåm, 1993). This class of algorithms is intuitive because they are of "behavioristic" type: they model out-of-equilibrium behavior as a "gradient" system driven by quite natural incentives.

The Data

The parameters α and γ are set equal to 3 and 1 per billion kilograms, respectively, to give a billion age zero fish (assuming negligible weight at age zero) when the spawning biomass is half a million tons.[5] Based on the reported survival rate of cod, s is given a value of 0.81 for all a. The price, v= NOK 6.78 and 7.46[6] per kilogram of cod landed by trawlers and coastal vessels, respectively (Sumaila, 1997). The cost parameter k_m,which denotes the cost of engaging a fleet of vessels (10 and 150, respectively for *tf* and *cf*) for one year, is calculated to be NOK 210 and 230 million, respectively, and b is set equal to 0.01. The discount factor is given a value of 0.935 as recommended by Norway Bank. The initial number of cod of age groups 1 to 8 are obtained by taking the average of the initial numbers from 1984 to 1991 reported in Table 3.12 of the ICES (1992). For the other age groups, I assume the same number as for age group 8. This gives (460,337,298,223,117,61,33,9,9,9,9,9,9,9,9) for a=1..15, resulting in an estimated initial stock size of 2.24 million tons. The parameter p_m=0 for $a<7$ and 1 otherwise; $q_{a,tf}$=0 for $a<5$; $q_{a,tf}$= 0.032,0.062,0.075 for a= 4,5,6, respectively and $q_{a,tf}$ =0.084 otherwise. $q_{a,cf}$ =0 for $a<7$; and $q_{a,cf}$ = 0.056,0.14,0.191,0.255,0.217,0.153,0.089,0.051,0.0255, for a=7..15, respectively. w_a = (0.1,0.3,0.6,1.0,1.4,1.83,2.26,3.27,4.27,5.78,7.96,9.79,11.53,13.84,15.24,16.34) for a=0..15; and w_{sa} is assumed to be 90% of w_a (see Sumaila, 1995).

3. The Results

Plots of the economic rent and standing biomass as a function of the MPA size are presented in Figure 1 for both the cooperative and the non-cooperative management scenarios. The figure shows that total economic rent from the fishery is strongly related to the size of the MPA. The rent increases with the MPA size until an optimal size is reached at 60% and 70% under non-cooperative and cooperative management, respectively. With regards to standing biomass, we see a similar pattern, total standing biomass in both the protected and fished areas, increase with increasing MPA size. But contrary to what one would have expected, it peaks at the same MPA sizes as in the case of the economic rent. One would have expected the standing biomass to keep increasing linearly with size but this is not the case. The reason for this counterintuitive result is that after 60% and 70% of the habit has been protected under non-cooperative and cooperative management, respectively, optimal fishing in the unprotected area requires a much lower standing biomass in this part of the habitat, which is low enough to more than compensate for the higher biomass in the protected area.

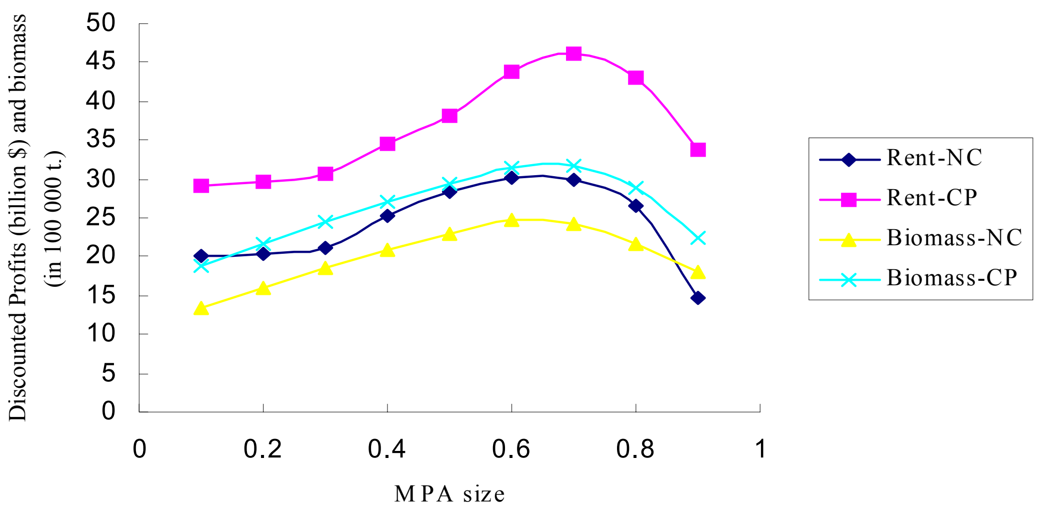

Figure 1: Rent and standing biomass as a function of MPA size

The base case results for key parameters of the model (discounted profits, standing stock biomass and MPA size) are presented in Table 1. The table reports the outcomes for the 'with' and 'without' an MPA under both non-cooperative and cooperative management. In the case of the 'with' MPA, the MPA size that gives the highest discounted profits are reported.

Table 1: Base case: Total discounted profits (in billion NOK), the average annual standing biomass (in million tones) and MPA size in percentage of habitat area.

		Non-cooperative	Cooperative
Discounted profits (NoMPA)	Trawlers	13.93	18.15
	Coastal	12.60	16.82
	Total	26.53	34.97
Discounted profits (BestMPA)	Trawlers	13.77	23.70
	Coastal	16.50	22.37
	Total	30.27	46.06
Average stock biomass	NoMPA	1.15	1.81
	BestMPA	2.48	3.16
	MPA-size (%)	60	70

We see that under the assumptions of the model, (i) MPAs are likely to give higher discounted profits from a fishery that is likely to face a shock. Under non-cooperative management fishers make a total of about NOK 30.27 billion with an MPA, compared with NOK 26.53 billion without an MPA. This is achieved with an MPA size of 60% of the habitat. The equivalent numbers under cooperation are NOK 46.06 and 34.97 billion, respectively. In this case the optimal MPA size is 70%. To reveal the insurance value of MPAs under the two management regimes, I compared these numbers to the discounted profits that would be obtained when the habitat is assumed not to face a shock. This comparison showed that (i) MPAs manage to protect about 62% of the no shock returns to the fishery under cooperation, and 68% under non-cooperation.

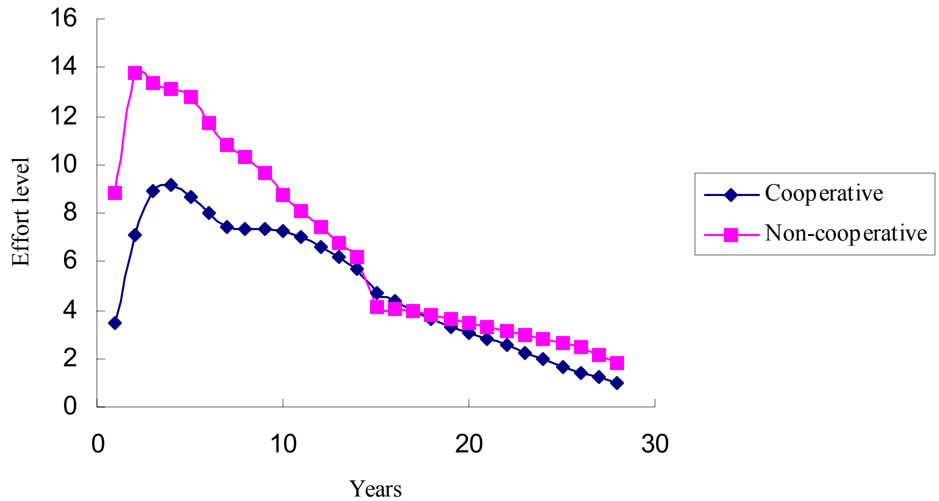

Figure 2: Effort profile under cooperative and non-cooperative management

It should be noted that in general higher economic benefits are achieved under cooperation than under non-cooperation. This is because fishers in a cooperative setting allow the resources to build to higher levels after the shock has occurred, by employing less fishing effort than under non-cooperation, especially during the initial periods of the time horizon of the model (see Figure 2). On average between 28 and 35% more fishing effort is employed under non-cooperative than under cooperative management.

More fish is left in the sea “with” than “without” an MPA (see Table 1). Hence, the implementation of MPAs can protect and enhance the stock biomass by helping maintain high standing fish biomass under the scenarios explored. More fish is left in the sea under the cooperative management regime because fishers here already have an efficient management policy in place; hence, they are in a better position to reap benefits from the insurance cover that MPAs provide. This result leads to two interesting observations. First, fisheries with good management plans can, under certain situations, benefit from implementing MPAs. Second, MPAs are no panacea – they need to be implemented as complements to other traditional management tools.

The discount factor, the exchange rate between the protected and unprotected areas, and the degree of shock introduced in the model were varied to examine how sensitive the model results are to changes in these parameters. The optimal MPA sizes remain the same except when a milder recruitment failure is assumed, and only under non-cooperative management (see Table 2). In which case, the optimal MPA size changes from 60 to 50%. An interesting result from the sensitivity analysis is that at a high discount factor (98%), MPAs do not appear to enhance economic benefits. This is an indication that MPAs are a possible means by which to mitigate the negative effects of high discount rates in fisheries. This means that when fishers are very impatient, e.g. in developing countries because of the pressures of meeting basic needs, or when a fishery is operating under open access, MPAs could be employed as a tool to protect the stock, and mitigate economic waste.

Table 2: Sensitivity analysis: Total discounted profits (in billion NOK), the average annual standing biomass (in million tonnes) and MPA size in percentage of habitat area.

Discount factor of 0.98

		Non-cooperative	Cooperative
Discounted profits (NoMPA)	Trawlers	23.57	48.91
	Coastal	29.18	54.52
	Total	52.74	103.41
Discounted profits (BestMPA)	Trawlers	24.32	36.00
	Coastal	25.84	41.61
	Total	50.17	77.60
Average stock biomass	NoMPA	0.91	2.50
	BestMPA	2.12	2.91
	MPA-size (%)	60	70

Lower migration rate of 0.4 of biomass in protected area

		Non-cooperative	Cooperative
Discounted profits (NoMPA)	Trawlers	13.93	18.15
	Coastal	12.60	16.82
	Total	26.53	34.97
Discounted profits (BestMPA)	Trawlers	11.90	17.80
	Coastal	12.79	16.46
	Total	24.69	34.26
Average stock biomass	NoMPA	1.15	1.81
	BestMPA	2.79	3.43
	MPA-size (%)	60	70

Milder shock – Recruitment failure from year 5 to 9

		Non-cooperative	Cooperative
Discounted profits (NoMPA)	Trawlers	13.07	17.05
	Coastal	11.30	15.28
	Total	24.37	32.33
Discounted profits (BestMPA)	Trawlers	14.77	24.09
	Coastal	16.10	22.32
	Total	30.87	46.41
Average stock biomass	NoMPA	1.36	2.02
	BestMPA	2.50	3.17
	MPA-size (%)	50	70

4. Concluding remarks

Using the model developed in this article, I have demonstrated that MPAs can help protect losses in economic rent from a fishery in a real world situation, where shocks to the habitat are bound to happen from time to time. The establishment of MPAs could help maintain high fish biomass in the marine habitat. This is the case whether fishers behave cooperatively or not. Hence, this study brings to the fore the insurance value of MPAs, as argued by, among others, Clark [1996] and Lauck [1996].

The paper also shows that for the full economic benefits of reserves to be realized they have to be implemented as part of an efficient management package. The article isolates the differences in economic and biological outcomes depending on whether the fishery is managed cooperatively or non-cooperatively. Finally, it is demonstrated that MPAs could serve as useful fisheries management tool when fishers have high discount rates, and are therefore very impatient.

Reference

Armstrong, C.W and U.R. Sumaila (2000). *Optimal allocation of TAC and the implications of implementing an ITQ management system for the North-East Arctic cod.* Land Economics, 77(2001): 350-359.

Bohnsack, J.A. (1990). *The potential of marine fishery reserves for reef fish management in the U.S. Southern Atlantic.* Miami: NOAA tech. Memo NMFS-SEFC-261. 40 pp.

Clark, C. (1996). *Marine Reserves and the Precautionary Management of Fisheries*, Ecol. Applic., 6(2): 369-370.

Conrad, J.M. and C.W. Clark (1987). *Natural resource economics: Notes and problems.* Cambridge University Press, Cambridge.

Flåm, S.D. (1993). *Path to constrained Nash equilibria.* Applied Mathematics and Optimisation 27: 275-289.

ICES (1992). *Report of the Artic Fisheries Working Group*, Copenhagen.

Hannesson, R. (1998). *Marine Reserves: What Would They Accomplish*? Marine Resource Economics, 13: 159-170.

Holland, D.S. (1998). *The use of year around closed areas for the management of New England trawl fisheries.* PhD Thesis, University of Rhode Island, Kingston, RI, USA. 329 pp.

Holland, D.S. and R.J. Brazee (1996). *Marine Reserves for Fisheries Management.* Marine Resource Economics, 3(11): 157-171.

Lauck, T. (1996). *Uncertainty in fisheries management.* pp. 91-105. In Fisheries and uncertainty: a precautionary approach to resource management, pp. 91-105. Ed. by D. V. Gordon and G. R. Munro. University of Calgary Press, Calgary, Canada,195 pp.

MacCall, A.D. (1990). *Dynamic geography of marine fish populations.* Washington Sea Grant Program, University of Washington Press, Seattle, 153 pp.

Nakken, O.P., P. Sandberg and S.I. Steinshamn (1996). *Reference points for optimal fish stock management.* Marine Policy 20(6): 447-462.

Pitcher, T.J., R. Watson, N. Haggan, S. Guénette, R. Kennish, U.R Sumaila, D. Cook, K. Wilson and A. Leung (2000). *Marine reserves and the restoration of fisheries and marine ecosystems in the South China Sea.* Bulletin of Marine Science, 66(3): 543-566.

Sanchirico, J.N. and J.E. Wilen (1999). *Bioeconomics of Spatial Exploitation in a Patchy Environment.* Journal of Environmental Economics and Management, 37: 129-150.

Sumaila, U.R. (1998). *Protected marine reserves as fisheries management tools: a bioeconomic analysis.* Fisheries Research, 37: 287-296.

Sumaila, U.R. (1997). *Cooperative and non-cooperative exploitation of the Arcto-Norwegian cod stock in the Barents Sea.* Environmental and Resource Economics, 10: 147-165.

Sumaila, U.R. (1995). *Irreversible capital investment in a two-stage bimatrix game model.* Marine Resource Economics 10(3): 163-183.

Sumaila, U.R., S. Guénette, J. Alder, and R. Chuenpagdee (2000). *Addressing ecosystem effects of fishing using marine protected areas.* ICES Journal of Marine Science 57(3): 752-760.

Walters, C. (2000). *Impacts of dispersal, ecological interactions and fishing effort dynamics on efficacy of marine protected areas: How large should protected areas be*? Bulletin of Marine Science 66(3): 745-757.

Endnotes

[1] I thank Scott Farrow for alerting me to this information.

[2] Executive Order 13158, May 26, 2000 available at www.whitehouse.gov.

[3] This function is chosen because recent biological studies have shown that it is more realistic than the Ricker recruitment function for species such as cod (Pitcher and Guénette, pers. comm.).

[4] Clearly harvest costs may be affected by the MPA size, making for longer travel distance. However, this would depend on the structure and positioning of the MPA, as well as the fisher's alternatives, issues that are beyond the scope of this paper.

[5] This is the minimum spawning biomass recommended to ensure the long term sustainability of the North-east Atlantic cod (Nakken et al., 1996).

[6] A US$ is equal to about NOK8.80 in October 2001.

Questions:

Scott Farrow: Is your non-cooperative solution like open access, or like a monopoly?

Ussif Rashid Sumaila: It is not as extreme as open access, so rent is not completely wasted.

Yvonne Ortiz: All the graphs show declining biomass at the end regardless of whether MPAs were there or not.

Ussif Rashid Sumaila: This is because by the time we get to the end, it is like the end of the world is here because we are approaching the last period of the model. It does not matter whether we have MPAs or not.

PAPERS IN ABSTRACT

This section reports the abstracts of papers, and their discussion, that were delivered orally at the conference, but that were not submitted as papers for this publication.

MARINE PROTECTED AREAS AND UNCERTAINTY

Louis W. Botsford1, James E. Wilen2 and Alan Hastings3
1Dept. of Wildlife, Fish, and Conservation Biology
2Dept. of Agriculture and Resource Economics
3Dept. of Environmental Science and Policy, University of California, Davis, CA, USA

Marine protected areas have been proposed to increase fishery yield and reduce uncertainty in fishery management. Several studies have indicated that they provide the former primarily when the fishery would otherwise be overfished We focus here on the latter. We show that while MPAs may reduce the uncertainty in specifying removals from the population, uncertainty in the level of removal tolerable for sustainability remains, and a new source of uncertainty associated with larval dispersal is introduced. These affect the relative economic advantage of MPAs over classical management by effort control.

ESTABLISHING MARINE PROTECTED AREAS: SOCIOECONOMIC AND BIODIVERSITY CONSIDERATIONS

Anthony T. Charles
Management Science / Environmental Studies, Saint Mary's University, Halifax, Nova Scotia, Canada

The use of marine protected areas (MPAs) has become a widely recognized tool for maintaining and enhancing marine ecosystem health, and for helping to achieve sustainable fishery systems. However, the establishment of MPAs is clearly not a straightforward task. This paper examines socioeconomic and biodiversity issues arising in MPA design, with emphasis on the implications of heterogeneity within the human system and the natural ecosystem. On the human side the one hand, an MPA can have differential impacts on the various players involved, possibly leading to a lack of acceptance and a loss of potential benefits. This highlights the importance of considering matters of process and distribution in MPA design. At the same time, the possibility of spatial heterogeneity in fish stocks (for example, involving an uncertain distribution of genetically-distinct sub-stocks) implies that MPAs can impact on biodiversity in both desirable and undesirable ways. Simulation models are developed and utilized to aid in the analysis of these socioeconomic and biodiversity implications.

Questions

Rögnvaldur Hanneson: Is there a feedback link between profits and fishing efforts?

Anthony Charles: This is a case where institutional assumptions are important. This model is not open access in the traditional sense. It assumes a degree of territorial rights and much more of a natural limitation to available fishing time, which seems to be the case in many artisanal fisheries.

Ussif Rashid Sumaila: How does uncertainty figure in your model?

Anthony Charles: At the moment, it is not included, although I have another model that focuses on uncertainty and impacts on uncertainty. The key here was to focus on distribution. Uncertainty will complicate the model, but it will not affect the results so much as the perception of results. Are there ecosystem benefits? That is the big uncertainty.

FISHERIES AND NATURE CONSERVATION – TWO OPPOSITE DIRECTIONS ?

Ralf Döring
Botanical Institute – Dept. of Landscape Economics, University of Greifswald, Greifswald, Germany

The discussion about creating marine protected areas is controversial. Nature conservationists want to protect parts of the marine ecosystem. They are emphasising the need for protection of spawning grounds, rare marine habitats, rare bird species etc. Fishermen declining these need. They are stressing that although they use the ecosystem valuable habitats exist. However no-take-zones (often spawning grounds) were established by fishermen. So, obviously no-take-zones are useful for protection of fish stocks and habitats as well.
But a well protected area needs a buffer zone. To bring this two positions together here it is to ask whether the aim of conservation there could be reached with some sort of ‚environmental acceptable fishing practice'.
The paper describe the controversies about a possible protected area at the Baltic Sea coast of Germany. Some parts of this area are out of use voluntarily to preserve juvenile fish stocks. From a nature conservationist point of view this area should be a reserve to protect a feeding ground for rare bird species in winter. It seems that both positions could be reached with a no-take-zone for parts of the ecosystem. Around this full protected area a buffer-zone could be established

were an environmental friendly fishing technique might be used to support the aim of conservation. The markets for fish don't honour an environmentally sound fishing practice at the moment. So, to create more acceptance the fisherman should be paid for the use of environmental acceptable fishing gear in this area as long as this fisheries must compete with other fisheries on the same markets. In Germany products from organic farming reached good market prices and market positions. It seems possible that fish products from environmental sound fisheries will be part of this market in the future.

Keywords: Nature Conservation, Acceptance of no-take

Questions

Darwin Hall: In California, there is an organization of farmers that started in the 1980s, to give certification to organically grown foods. They started out self-certified, and then introduced the certification in the legislation in the state of California. Are there groups of fishers in Germany that can take on that role?

Ralf Doering: No, it is different in Germany. There is a big organization, and farmers can go into this organization and agree to its rules to be certified. I hope fishermen can have these organizations, but it is hard to say that a particular fish is "organic", so there has to be a way to determine certification. Also, fishermen do not operate in groups – certification is individual.

Darwin Hall: In agriculture, they can go to the sites and see how the crops are grown, as well as records that farmers use for tax records. An analogous scenario with fishermen would be observing equipment or going on trips with fishers to see if they are familiar with the site they claim to fish in.

NATIONAL SURVEY OF AMERICANS' ATTITUDES TOWARD PROTECTED AREAS IN THE OCEAN

Vikki Spruill[1], Susan Boa[1], Lisa Dropkin[1] and Mark Mellman[2]
[1]SeaWeb, 1731 Connecticut Ave, NW, Washington, DC, USA
[2]The Mellman Group, Washington, DC, USA

The current U.S. policy discussion of protected ocean areas encompasses both changes to the National Marine Sanctuaries program and the establishment of new Marine Protected Areas. In fall 1999 SeaWeb, funded by the Goldman Fund, commissioned a public opinion poll to examine Americans' attitudes toward protected areas in U.S. ocean waters. The survey was designed to test public support for establishing protected areas and strengthening protections within existing U.S. Marine Sanctuaries. The poll also measured perceptions of the condition of the ocean, perceived problems facing the ocean, and attitudes toward human activity and use of resources within protected areas. Finally the survey tested persuasive messaging to determine how to best to communicate about protected areas.

The survey found that Americans express affinity for and concern about the oceans, although they continue to hold misconceptions about threats to ocean health. Most view oil pollution as the most significant ocean threat, while fewer understand the impact of run-off and overexploitation of ocean resources. Despite some confusion as to how specific areas in the ocean can be distinguished for protection, Americans express strong support for establishing protected areas that limit or prohibit damaging human activities. Americans are largely unaware of the existing Marine Sanctuary program and believe this program should be modified to prohibit those commercial and recreational activities that are harmful to wildlife and habitat. The study shows that there is opportunity to create a new debate about ocean protected areas, but that a successful campaign must reconcile differences between Americans' concerns and the science of protected areas. The survey suggests successful public communications will reinforce a theme of human dependence on the ocean and the damage already done.

The survey interviewed 802 adult Americans nationwide. The margin of error is +/- 3.5 percentage points at the 95% confidence level.

Questions

Saudiel Ramirez-Sanchez: Would it not be more important to assess people's understanding on how science works rather than assessing people's opinions about specific environmental issues?

Nancy Baron: It is hard to do it in a short piece, but a feature piece can get into detail. For instance, I did a feature on the Fisheries Centre, and it was possible to get into details of Ecopath and other modeling techniques. It depends on the media and the audience as well.

Lisa Dropkin: A public education campaign is massive and expensive, with very little return. We compete with lots of other information. Americans see, on average, more than 30,000 commercials in a year. The factual data that people assimilate is very little.

Michael Murphy: With regards to terminology, I noticed that you were talking about MPAs in this talk, but in surveys you referred to “ocean protected areas”, “coastal protected areas”, “marine sanctuaries”, etc. In the scientific community we do not use these terms. It seems that using “coastal protected areas” got a better response from the public.

Lisa Dropkin: Language is indeed important. We did a follow up survey where we looked at the effects of using “marine reserves” versus “ocean protected areas” versus “marine sanctuary”, and it turned out that using “Marine Protected Areas” did the best in getting the gist of the concept across. There is a lot of language work to do. We need to be careful when going out and emphasizing “coastal protected areas” because we do not want to burn ourselves further down the road when we put MPAs in other areas. What the public consider “coastal” and what we consider “coastal” are different.

CAN TRADITIONAL FISHERMEN AFFORD NOT TO GUIDE? TRADEOFFS OF WHALE SHARK TOURISM VS. FISHING ON SPAWNING AGGREGATIONS IN A PROPOSED MARINE RESERVE

Rachel T. Graham[1,2] and William D. Heyman[2]
[1] Environment Department, University of York, York
[2] The Nature Conservancy, Belize Marine Program, Punta Gorda, Belize

The wisdom of fishing spawning aggregations has increasingly come under fire from conservationists following the collapse and elimination of many key aggregations worldwide. Fully-protected marine reserves, areas closed to all fishing, are seen as possible means of protecting vulnerable fish stocks and critical sites such as spawning aggregations. The sustainability and successful local ownership of proposed marine reserves often hinges on the development of economic alternatives with former users of the areas protected. This transition period often requires additional capital outlays and training which can be can be facilitated and co-financed by outside groups. We present the case for tradeoffs between whale shark guiding and commercial fishing in the proposed no-take zone of the Gladden Spit Marine Reserve by comparing catch per unit effort and revenue of the fishery to effort and revenue from guiding. This site harbours the last commercially fished snapper spawning aggregation on the Belize Barrier Reef and a seasonally predictable aggregation of whale sharks. Over the past 20 years snapper landings have declined significantly while tourism interest in the whale sharks has increased. Declaration of a reserve at this site will displace fishers, and some perceive a reserve as threatening their livelihoods (even though a reserve would almost certainly enhance fishery sustainability). Economic alternatives such as whale shark guiding and recreational fishing are being developed to facilitate the shift away from unsustainable fishing practices and promote local ownership and stewardship of the reserve’s resources. Several fishers have already made the transition to tour guides, and tourist demand appears to outstrip guide supply and the pull of this lucrative activity outweighs the increased effort and declining income associated with fishing the spawning aggregation.

Keywords: whale sharks, spawning aggregation, economic alternatives, marine reserve, guiding.

Questions

Jean Boncoeur: On the island on the northern shore of the Honduras, the main season for mutton snapper spawning is October to November. Is that the same stock as the one you refer to in your paper, since they have a different spawning season?

Rachel Graham: We are seeing different spawning seasons for snappers. We are working with people from the Honduras to see if they come from the same stock. There are some tagging experiments going on, and we are seeing large-scale migrations in the snappers.

Daniel Holland: You said there were some small areas within the marine reserve. If these small areas are no-take areas, what is controlled in the other areas of the marine reserve?

Rachel Graham: In those areas, only recreational fishing is allowed. They are trying to determine the regulations on it – do they put a limit on catch per day? How would they enforce it? They are still trying to decide on things like that. The procedure for setting protection on this area got raced through because the Minister of Agriculture and Fishing really likes this area and he knows how slowly these things take place. Usually going through the bureaucracy to set up a protected area takes six to seven years, but this one only took two.

Sean Hastings: In the community consultations, who actually drew the lines for the areas, and when were the lines drawn?

Rachel Graham: The lines were drawn by three fishermen and Will Heyman in April 2000. They have been circulated to a certain degree, but there are lots more consultations to do. Will went out with the fishermen and set the nursery areas as a no-take zone. It was the fishermen who suggested that the nursery areas be set as a no-take zone. We need more input from other fishers, but it is a start.

Jackie Alder: You presented whale shark tourism as an alternative economic industry for the fishers, but this only happens for two months of the year. What do fishermen do for the other ten months?

Rachel Graham: Although fishermen fish year-round, they only spend 4 months of the year fishing in that area anyway, so the trade-off is not so bad. Whale shark tourism will not be the only thing they do. If they did recreational fishing tours as well, they could go right through the general use area all year.

STAYING AFLOAT: DEVELOPING ECONOMIC ALTERNATIVES WITH FISHERMEN, TO SUPPORT THE DECLARATION, MANAGEMENT, AND LOCAL OWNERSHIP OF MARINE RESERVES IN BELIZE

William D. Heyman[1] and Wil Maheia[2]
[1]The Nature Conservancy, Belize Marine Program, Belize
[2]Toledo Institute for Development and Environment (TIDE), Punta Gorda, Belize

Scientific and technical studies have demonstrated how marine reserves can effectively improve fish stocks over time by promoting emigration and larval dispersal. However, fishermen still have to feed their families every day. As fisheries resources are steadily declining in southern Belize and more marine reserves are declared to protect marine resources and accommodate tourism, fishermen are increasingly seeking economic alternatives to “stay afloat”. Belizean non-government organization, TIDE, along with The Nature Conservancy, have embarked on a comprehensive program retraining fishermen in Southern Belize to work in new industries that are consistent with the sustainable management of marine reserves. The alternatives were chosen based on criteria including socio-cultural acceptance, economic potential, and compatibility with the local environment. Retraining has focused primarily on tour-guiding, and includes catch and release fly fishing guiding, kayak guiding, and scuba dive guiding. As a result of this two-year participatory process, tour guides have become active stewards and supporters of three marine reserves in southern Belize. The economic implications of both commercial fishing and the alternatives will be discussed and compared.

Questions

Tanya Dobrzynski: You say that tourist demand for tours does not exceed supply. Is there a possibility that in the future, it will exceed supply and tourism will become unsustainable?

Rachel Graham: The guides take tourists out in small boats, with only a few people in one boat, so the impact on the ecosystem is low. Tourists do not want to be packed into little boats. They want to have a private, personal experience. I do not think tourism will grow too quickly in the future.

GENERAL DISCUSSION

Trond Bjorndal
Let me start with the topic that Rögnvaldur Hanneson brought up at the beginning of the conference, that he believes that MPAs are of limited economic benefit. Do you still feel this way after hearing today's empirical studies, and can you comment on using MPAs as a part of optimal fisheries management?

Rögnvaldur Hanneson
I do not think any of the papers that were presented refute the idea that MPAs do not much improve upon open access fisheries. As for the second question on how MPAs can supplement the current management system, there are many cases in the world where it is not possible to implement ideal management, like ITQs. What can you do then? In such cases MPAs might compensate for things you cannot do. I think I mentioned reef fisheries as an example of how MPAs could compensate for the impossibility of putting ITQs in place. Some of the examples we heard today support that contention. It was very interesting to hear from Rachel where fishermen have found it in their best interest to switch from fishing to tourism. Fisheries and fish resources differ greatly from one part of the world to another, and we are all influenced by the reality we live in. I myself am most familiar with fisheries that produce material benefits and so tend to view fisheries in that light. I will not apologise for that. In many parts of the world, ocean resources have uses other than for food, and in that case MPAs can be useful to allocate fish resources to the most preferential use or to reach a compromise between conflicting uses.

Anthony Charles
I was pleased with the blend of models with case studies and practical work so far in this conference.
I think there is a place for both approaches. Meta-analyses with case studies combined with models would be useful. I was impressed that both theoretical and practical applications dealt with both spatial and human distribution. Both methods are fascinating from a research point of view, and also practically important, like in the case of Belize. This heterogeneity is at the forefront of the MPA issue.

Trond Bjorndal
Does anyone want to elaborate on the benefits from MPAs?

Callum Roberts
Mallory King did her thesis on looking at the benefits of MPAs and who would benefit, with a case study in Kasidy National Marine Park. She started out thinking that there would not be much benefit to MPAs, but the problem with many marine reserve studies is that they drew measures of benefits much too narrowly, so instead she looked at measures of security. She looked at how food security of households was connected with distance to MPAs. Those near MPA boundaries were more secure; not that fish catch increased, but the CPUE decreased so the benefits coming from alternative jobs that the people could take on are feeding back into local communities. The people who benefited least were fishermen. They are the least able to take advantage of MPAs, so you have to give them access to alternatives.

Nina Mollett
This is a follow-up on Trond's question. I am working in Juneau, for the National Marine Fisheries Services. Alaskan fisheries are already highly regulated with ITQs. We are looking at marine reserves as an option amongst many. What do you do if already have good management system in place? What benefits would MPAs bring in then?

Gordon Munro
Marine reserves are important if some kind of management procedure is already in place, so that MPAs act as a supplement. The quote I read this morning in the newspaper stresses the degree of risk with the current management system, and it looks at marine reserves as enhancing the current system. If you are looking for benefits, it is not good enough to say that the expected returns are greater with open access. The real tradeoff is between the expected return and the safety of the fishermen's jobs.

Ussif Rashid Sumaila
There will be papers presented tomorrow that will address these questions, so perhaps we should wait and see what they have to say.

James Sanchirico
For MPAs to present a win-win situation, where both the industry and the biological system will benefit, requires that the ecosystem in question is already over-exploited. In an over-exploited situation, setting aside a place will not affect a fishery much since there is not much there to be fished anyway, but in an optimal fishing setting, fisheries will lose because they will be losing fishing grounds.

Rachel Graham
We found that for our study in Belize, promoting fishermen exchanges is highly useful. We are not getting people to set aside areas or to change fishing methods; what we are doing is getting people to go to Guatemala, Jamaica, and the Honduras to see what the effect of not having a marine reserve can do to the ecosystem, and they come back and say that they are convinced that they need a marine reserve. If you get them to go to very depleted places, they can see for themselves. This will facilitate fishermen for joint discussions to garner support for marine reserves, and it is very effective. It is a community-based decision making process. This is partly the cause for such a rapid designation of the marine reserve.

Anthony Cox
This is a follow-up to Gordon's comments. An issue that economics tend to duck in this debate is that it is not only the returns that matter, but also distribution. Traditionally, we say that as long as the aggregated net benefits are positive, then we assume that the political system will take care of the distribution, but that is not what happens. Also, in a public assessment poll, it was found that risk assessment matters as well. Some points need to be made about the results that Lisa shows: there is a distinct difference between what scientists see and what the public sees as key issues. There is also a big gap in the opinion polls on the actions that people think need to be done, and what they are willing to pay. The distribution and perception issues are linked, and that is another thing that needs to be worked on.

Tony Pitcher
It seems to me that since 1997, which is the last time we talked about this, there has been a shift in scientific opinion as well. I am not hearing the same arguments as I heard last time, like the contention that fishing causes no damage so MPAs are not necessary. People now seem to accept that MPAs are there now and that there are benefits to be had. From the economic side, the same argument that MPAs can offer no benefit is still in place, but that is something we could work on. Social benefits are another issue that was hardly mentioned in 1997 meeting, but people can now mention these other, non-monetary benefits that can come from MPAs. So, on the ecological and social sides, there has been a shift in opinion. We still need to hear from economists on the second day of the meeting.

Eric LeBrun
In many of the models that have been presented, there has been a very strong assumption that MPAs have a positive impact due to migration of fish from MPAs to fishing grounds. Have there been any cases proven or observed that show the positive effects of fishermen on fishing grounds?

Callum Roberts
In one instance in St. Lucia, biomass has tripled inside the MPA and doubled outside over a period of three years. Here they have established a large percent of coral reef habitat (35-40%) as a reserve. There is a study in the Philippines which shows that there is a higher density of fish in reserves - that over a number a years a spill-over problem is created. However, there is not enough data to assess this yet, and more collaborative studies/data sets are needed.

David Whitmarsh
There is evidence, a measurable effect, in northwestern Sicily of increased biomass leading to higher fishing incomes in areas around MPAs that started ten years ago.

Darwin Hall
For many different kinds of fish or marine animals, size affects fecundity, and we would expect an increase in growth rate for fish in MPAs. Which of the three models that we have just seen best takes that into account?

Ragnar Arnason
I certainly included it in my model – not explicitly, but implicitly as a shift in the growth factor.

As a comment on the earlier discussion, we have to show that a shift in the biomass growth function takes place (second-order effect), not just a shift in the biomass itself (first-order effect).

Ussif Rashid Sumaila
In my model, I use a Beverton-Holt recruitment function whereby changes in fecundity can be taken into account.

Darwin Hall
Does the fecundity change with the age of fish in the Beverton-Holt model?

Ussif Rashid Sumaila
The function works like this: if the fish is one year old, it is not in the function because it has not reached reproductive maturity yet; if the fish is four years old, it is 50% included in the function; if the fish is eight years old, it is 100% included in the function.

Darwin Hall
There is a bioeconomic flipside: there is talk about MPAs and fishers going to the borders of MPAs. Although going out to the borders of MPAs can take a longer time, the amount of time necessary to look for fish once there may become lower. Do the models account for that?

Ragnar Arnason
Yes, it is included in my model.

Rachel Graham
I do not understand how these models take into account the geographical and ecological spawning aggregations, and how MPAs affect that.

Ragnar Arnason
The model suggests a simple shift in the biomass growth function, and that covers aggregations, changes in fecundity, etc. However, in spite of that, it is still difficult for me to show the benefits of MPAs.

James Sanchirico
People take too much from the models. Remember, models are only illustrative of the things going on in the world. Sometimes people forget that. You have to ask yourself what spawning aggregations do, and ask what the models do to simulate the effect and whether they are sufficiently robust.

Louis Botsford
I do not think any of our models are spatially heterogeneous or include full life-history considerations.

SUMMARY OF THE CONFERENCE

THE "NEW" EMERGING ECONOMICS OF MARINE PROTECTED AREAS

Scott Farrow
Carnegie Mellon University

Policy interest and research in Marine Protected Areas has grown rapidly in recent years, even attracting the attention of the U.S. President who issued an executive order calling for "appropriate actions to enhance or expand protection of existing MPAs and establish or recommend, as appropriate, new MPAs." Among the tools and guidance recommended for use in this effort is an "assessment of the economic effects of the preferred management solution.[1]

The research for the conference is consequently both more focused and advanced than the eclectic survey in Farrow (1996) on the emerging economics of marine protected areas. In the following pages, the conference presentations are mapped into key topics from the earlier review and new topics are identified both from conference presentations and from more general areas of economics.

As a result of work on MPA related projects with the U.S. Man and Biosphere program, Farrow (1996) raised six questions that he thought would help link the interests of MPA managers, often trained in the natural sciences, with those of economists. After reviewing the role of benefit-cost analysis and economic efficiency, the questions asked regarding the emerging economics of MPAs were:

- Are MPAs economically justifiable?
- Where to draw boundaries ?
- Can diversity and species survival be included?
- Is every MPA an island?
- Is passive use value relevant?
- Are MPAs politically supportable?

The papers of the conference can be mapped reasonably well into these topics. Table 1 below provides the mappings of this author, using the first author from the conference.

While no such mapping is exactly accurate, the focus of the conference papers tended to emphasize two areas: 1) whether MPA's can be economically justified, often stated solely in terms related to the fishing industry, and 2) what activities generate or sustain political support for the distributional impacts of MPA's[2].

Topics receiving attention at the conference that were largely absent in the earlier survey included uncertainty and the advances in the spatial modeling of fisheries. The papers at the conference were an impressive advance over the state of the art in the early 1990s, with the analyses getting both more focused and integrating theoretical and empirical insights. In general, the economic papers at the conference tended to be skeptical of aggregate benefits (still focused on the fishing sector) that could result from the creation of Marine Protected Areas.

An outsider looking in

Some perspective on the economics of MPAs can be gained from matching its issues to advances outside this specialized area.. In such a spirit, Farrow mentioned several approaches that may be more favorable to the creation of MPAs. The discussion was divided into three parts: 1) concentration versus spreading of economic activity, 2) uncertainty and the precautionary principle, and 3) the hidden role of the Kaldor-Hicks potential compensation rule in economics. Each is discussed in turn.

The flip side of limiting access in order to create an MPA is the act of increasing the concentration of marine production activity in the MPA. This has parallels to research on the location of industry in which calls for a "new economic geography" has highlighted the role of increasing returns to scale in production, whether through organizational inter-dependencies or the technology of production itself (e.g. Krugman, 1995). This research outside the marine realm reinforces work by Helfand and Rubin (1994) that asks when externality generating activities should be concentrated in one area or dispersed. This line of thought has been largely ignored by marine economists (although echos appear in some work as by Pezzey, Roberts, and Urdal (2000)). For instance, if standard biomass growth models of a logistic sort are used, the biological "return to scale[3]" is mathematically confined to be less than or equal to 1 which cannot exhibit increasing returns to scale (a value greater than 1). Age structured models, allowing

[1] Executive Order 13158, May 26, 2000 available at www.whitehouse.gov.

[2] Sanchirico (2000) provides a more recent review focusing on research hypotheses.

[3] Often measured by elasticities of production, the ratio of percentage change in output to percentage change in an input.

Table 1: Conference Papers and Topic Mapping

	Are MPA's Economically Justifiable?	**How does econ help define the boundary**	**Role of species diversity/survival**	**Is every MPA an island?**	**Is passive use value relevant?**	**Distribution and political support**
Hannesson	√					
Rodwell	√	√				
Reithe	√					
Holland			√	√		
Alder, et al						√
Doering					√	√
Ortiz			√			√
Pitcher	√					
Beattie	√	√		√		
Graham	√					√
Heyman	√					√
Msiska						
Dropkin						√
Charles			√			√
Boncoeur	√		√			
Cox	√	√		√		
Nsiku	√		√			
Chuengpagdee						√
Hall					√	√
Rudd		√	√			
Silk						√
Sumaila	√					
Arnason	√					
Botsford		√		√		
Sanchirico	√	√				
Roberts	√			√		
MacDiarmid		√				
Fauzi						√
Farrow	√		√			√
Number of papers	**16**	**7**	**9**	**6**	**2**	**13**

different returns to scale in different patches (e.g. Guénette and Pitcher, 1999), would be more consistent with what has led to economic concentration in other fields of study. Economists and marine biologists may be talking past each other until the models capture this central element.

The theme of stock and other uncertainty received more, although not dramatic, emphasis at the conference. To the extent that the economic models address uncertainty, they tend to stay within a framework of asking if expected benefits exceed expected costs. Recent advances in analyzing investment synthesized by Dixit and Pindyck (1994) suggest that the standard economic decision rule is incorrect when uncertainty and irreversibility exist. Depending on who has the right of the status quo between maintaining the level of fish stocks or the level of fish harvesting, a formal precautionary rule can emerge in the new paradigm such that one should establish an MPA unless the costs to fishers are significantly higher than the expected benefits of the MPA, not just equal to or greater than the benefits. The outline of the approach is to consider that there are stochastic and irreversible costs from overfishing (shut-down or extinction) as well as stochastic benefits from continuing to fish. In an analysis parallel to that of investing in safety by Hayakawa and Farrow (2000) a quantitative and empirically based precautionary decision rule results from expected value maximization. Analyses that omit uncertainty, delayability, and irreversibility may be basing marine management decisions on the wrong benefit-cost criteria.

Finally, the conference interest in distributional impacts has a parallel in recent work in benefit-cost analysis. Distributional analysis has long been the weak point of standard analysis, typically using the Kaldor-Hicks potential compensation criteria. Recent emphasis on environmental justice and sustainability suggest that the potential criteria should be replaced with actual compensation criteria (Farrow, 1998). Concerns for both intra and inter-generational fairness suggest stronger grounds for compensating those who lose from a policy action than is typically considered in economic analysis. Thus case studies in local benefits and costs can be linked to larger studies of the economics of MPAs.

New "emerging" economics of MPAs

While not an official requirement for a degree, economists implicitly take a modified Hypocratic Oath along the lines of "Above all, do not reduce efficiency." Economists are rightly skeptical that many proposals that sound good to some stakeholders will fail a broader test. At the same time, new approaches in economics are questioning the universal application of: standard interior (convex) solutions to real world problems, decision criteria that ignores uncertainty and irreversibility, and distribution.

The research at the conference represented large strides over just a few years ago although fundamental issues remain. The exciting aspect of this evolving area is that for the moment, a good slogan still beats good analysis. However, advances in the synthesis of the natural and social sciences may yet lead to the compelling, empirically consistent analysis that provide a rational framework for managing our marine assets wisely.

References

Dixit, A. and R.(1994), Pindyck, Investment Under Uncertainty, Princeton University Press.

Farrow, S. (1996), "Marine Protected Areas: Emerging Economics," Marine Policy 20(6):439-446.

Farrow, S. (1998), "Environmental Equity and Sustainability: Rejecting the Kaldor-Hicks Criteria," Ecological Economics 27:183-188.

Guénette, S. and T. Pitcher, "An Age-structured Model Showing the Benefits of Marine Reserves in Controlling Overexploitation," Fishereries Research, 39:295-303.

Hayakawa, H. and S. Farrow, (2000) "An Implementable Precautionary Principle: Investing in Safety," Discussion Paper, Center for the Study and Improvement of Regulation, Carnegie Mellon University, Pittsburgh, Pennsylvania.

Helfand, G. and J. Rubin (1994), "Spreading versus Concentrating Damages: Environmental Policy in the Presence of Non-Convexities," Journal of Environmental Economics and Management, 17(1):84-91.

Krugman, P. (1995) Development, Geography, and Economic Theory, MIT Press, Cambridge, MA.

Pezzey, J.C.V., C. Roberts and B. Urdal (2000) "A Simple Bioecconomic Model of a Marine Reserve," Ecological Economics, 33:77-91.

Sanchirico, J. (2000), "Marine Protected Areas as Fishery Policy: An Analysis of No-Take Zones," Resources for the Future Discussion Paper 00-23, Washington, D.C.

Comments

Reg Watson

I would like to strongly support what Scott said about treating uncertainty. Fisheries modelers tend to use biomass dynamic (simple production) models in an effort to get biological modeling out of the way so they can get to the more complex economic modeling they wish to address, but this leaves out important things like recruitment, the influence long runs of bad (autocorrelated) years, and overfishing. These sorts of approaches cannot handle these things so their results naturally show that MPAs cannot offer protection against these risks. If these factors are added into the models, it will make a big difference.

Sean Hastings

I appreciate the distinction you made regarding approaches to the presentations. When we take our models to the policy makers, it becomes an issue of why we should not set MPAs as opposed as to why we should, and we still have a long way to go with our persuasion. We should integrate as many factors as we can in MPAs, not just talk about them in relations to fisheries.

Scott Farrow

What do economists add to this? I think it is our fixation on externalities. I think bringing passive values to this issue is one approach. There are other values involved outside of fisheries. In terms of a benefit/cost analysis, we should carry these other values along in every analysis. Set them at zero if you do not believe they have much impact on MPAs, but do not ignore them.

Louis Botsford

The talk I gave on persistence analysis, though I applied it to fisheries, also applies to other species and habitats associated with MPAs.