



Daniel W. Gotshall

*Sebastes chrysomelas*, black-and-yellow rockfish

## **A Review of the Ecological Effectiveness of Subtidal Marine Reserves in Central California**

### **Part I: Synopsis of Scientific Investigations**

**U.S. Department of Commerce**  
National Oceanic and Atmospheric Administration  
National Ocean Service  
Office of Ocean and Coastal Resource Management  
**Marine Sanctuaries Division**

April 2004



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## **Part I: *Synopsis of Scientific Investigations***

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## INTRODUCTION

Marine reserves, often referred to as no-take MPAs, are defined as areas within which human activities that can result in the removal or alteration of biotic and abiotic components of an ecosystem are prohibited or greatly restricted (NRC 2001). Activities typically curtailed within a marine reserve are extraction of organisms (e.g., commercial and recreational fishing, kelp harvesting, commercial collecting), mariculture, and those activities that can alter oceanographic or geologic attributes of the habitat (e.g., mining, shore-based industrial-related intake and discharges of seawater and effluent). Usually, marine reserves are established to conserve biodiversity or enhance nearby fishery resources. Thus, goals and objectives of marine reserves can be inferred, even if they are not specifically articulated at the time of reserve formation.

In this report, we review information about the effectiveness of the three marine reserves in the Monterey Bay National Marine Sanctuary (Hopkins Marine Life Refuge, Point Lobos Ecological Reserve, Big Creek Ecological Reserve), and the one in the Channel Islands National Marine Sanctuary (the natural area on the north side of East Anacapa Island). Our efforts to objectively evaluate reserves in Central California relative to reserve theory were greatly hampered for four primary reasons; (1) few of the existing marine reserves were created with clearly articulated goals or objectives, (2) relatively few studies of the ecological consequences of existing reserves have been conducted, (3) no studies to date encompass the spatial and temporal scope needed to identify ecosystem-wide effects of reserve protection, and (4) there are almost no studies that describe the social and economic consequences of existing reserves.

To overcome these obstacles, we used several methods to evaluate the effectiveness of subtidal marine reserves in Central California. We first conducted a literature review to find out what research has been conducted in all marine reserves in Central California (Appendix 1). We then reviewed the scientific literature that relates to marine reserve theory to help define criteria to use as benchmarks for evaluation. A recent National Research Council (2001) report summarized expected reserve benefits and provided the criteria we used for evaluation of effectiveness. The next step was to identify the research projects in this region that collected information in a way that enabled us to evaluate reserve theory relative to marine reserves in Central California. Chapters 1-4 in this report provide summaries of those research projects. Contained within these chapters are evaluations of reserve effectiveness for meeting specific objectives. As few studies exist that pertain to reserve theory in Central California, we reviewed studies of marine reserves in other temperate and tropical ecosystems to determine if there were lessons to be learned from other parts of the world (Chapter 5). We also included a discussion of social and economic considerations germane to the public policy decision-making processes associated with marine reserves (Chapter 6). After reviewing all of these resources, we provided a summary of the ecological benefits that could be expected from existing reserves in Central California. The summary is presented in Part II of this report.

## CHAPTER 1

### **No-Take Reserves in Central California Kelp Forests: Metrics of Human Impact or the Tip of the Iceberg?**

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#### **Abstract**

Most evidence for the effects of marine reserves comes from tropical nearshore ecosystems. We attempted to evaluate the effects of marine reserves on temperate kelp forest systems by contrasting the population structure (density and size distribution) of 10 species of epibenthic fishes and several aspects of the associated ecosystems between three marine reserves and adjacent exploited areas in Central California. Densities of fishes were 12-35% greater within the reserves but this difference was not statistically significant. Habitat features explained only 4% of the variation in fish density and did not vary consistently between reserves and non-reserves. The average length of rockfish (genus *Sebastes*) was significantly greater in 2 of the 3 reserve sites, as was the proportion of larger fish. Population density and size differences combined to produce substantially greater biomass and therefore reproductive potential per unit of area within the reserves. The magnitude of these effects seems to be influenced by the reserve's age. While our results demonstrate that current levels of fishing pressure influence kelp forest rockfish populations, differences between the reserves and adjacent non-reserves are surprisingly small. We discuss a number of reasons why the influences of fishing on kelp forest ecosystems may be greater, or at least different, than our findings indicate. Potentially confounding influences include the very small size of the reserves, effects of historical fishing, poaching, spillover effects on adult and larval populations from reserve to non-reserve habitats, and the possibility that catastrophic phase shifts induced by human disturbances have altered both reserve and non-reserve areas.

## Introduction

Marine resources worldwide are showing signs of degradation (Dayton et al. 1995, Botsford et al. 1997, Lauck et al. 1998, Pauly et al. 1998). The problem may be even worse than we imagine, owing to the largely unappreciated importance of top-down forcing processes (Pace et al. 1999, Estes et al. 2001), past losses of large vertebrates and other consumers (Jackson et al. 2001), and the tendency of many ecological systems to behave in chaotic and non-linear ways (Scheffer et al. 2001). Given these potential difficulties and their associated uncertainties, how do we assess and redress the impacts of fisheries and other anthropogenic effects on marine ecosystems? In response to the increasing evidence that management has failed to achieve fishery sustainability (Dayton et al. 1995, Botsford et al. 1997, Ralston 1998), there is growing interest in the use of no-take areas (marine reserves) as a fisheries management tool (Bohnsack 1993, Castilla and Fernandez 1998, Dayton 1998). Marine reserves may serve as buffers against catastrophic declines caused by the synergistic interactions between exploitation and environmental extremes (Bohnsack 1993), as well as protect against the inherent risk of uncertainty in fisheries management (Lauck et al. 1998). Additionally, they may aid in sustaining and possibly enhancing stocks (Murray et al. 1999).

Beneficial effects of marine reserves on fish and invertebrate populations have been demonstrated in numerous studies (National Research Council 2001, Halpern in press). These effects include increased abundance and increased individual size and age in targeted populations. No take reserves may also enhance habitat quality, such as recovery of corals (Roberts and Polunin 1993, McClanahan 1997a), species diversity (Cole et al. 1990, Russ and Alcala 1996), and community stability (Castilla and Durán 1985, Roberts and Polunin 1993, Dayton et al. 1995). Reserve effects may extend beyond reserve boundaries through spillover of adults and/or larvae to fishing grounds (Attwood and Bennett 1994, Holland and Brazee 1996, Castilla and Fernandez 1998).

The evidence that marine reserves enhance population abundance and individual size of exploited species is unequivocal. However, it really shouldn't surprise us that protection results in measurable effects, given the large number of fisheries that have declined or collapsed worldwide (Botsford et al. 1997, Pauly et al. 1998). The absence of any such effects would be far more surprising and remarkable. The more important question is whether or not marine reserves, in their present forms, can be used to achieve the goals of marine conservation and fisheries management? The answer depends on the extent to which measured effects of marine protected areas capture the full impacts of human exploitation. If marine protected areas in their current forms return these systems to near pristine conditions, then the course of effective conservation and management is relatively simple. If, on the other hand, pristine conditions were far different from those achieved or achievable by marine reserves, then the proper course of conservation and management is far less certain.

There are 103 marine protected areas along the California coast, only 11 of which receive protection from all take (McArdle 1997). Unfortunately, most of California's marine protected areas lack baseline biological information, without which their effects are difficult to evaluate. Understanding these effects is crucial if marine reserves are truly going to be incorporated into fisheries and ecosystem management. Thus, we set out in the mid 1990s to assess the effect of



marine reserves on populations of exploited fish species and their habitat in Central California kelp forests. We did this by contrasting three existing marine reserves with nearby areas in which fishing was permitted. This approach has been used elsewhere to evaluate the influence of fishing on reef fish populations (see for example, Buxton and Smale 1989, Cole et al. 1990, Grigg 1994, McClanahan 1994), although relatively few of these have been conducted in cold-temperate regions (Bohnsack 1998) (for cold-temperate examples, see -- South Africa: Buxton and Smale 1989, Bennett and Attwood 1991, Buxton 1993, Attwood and Bennett 1994; Chile: Castilla and Durán 1985, Castilla 1996, Castilla and Fernandez 1998; USA: Palsson and Pacunski 1995, Rogers-Bennett et al. 1995, Palsson 1998), and only a small number of other studies have considered kelp forest fishes (Cole et al. 1990, Palsson and Pacunski 1995, Palsson 1998, Babcock et al. 1999). Cold water/kelp forest systems differ from tropical reefs in numerous ways, several of which may influence their response to fishing. Compared with coral reefs, kelp forests are more productive (Duggins et al. 1989), have a lower fish species diversity (Ebeling and Hixon 1991), and support fish faunas with higher overall trophic status because of the paucity of herbivorous species (Gaines and Lubchenco 1982), at least in the northern hemisphere (Choat 1982).

The purpose of this paper is two-fold. First, we contrast three marine protected areas with nearby non-reserve areas in the Monterey Bay National Marine Sanctuary. These findings are a synopsis of the results reported by Paddock and Estes (2000). Second, we consider the implications of these findings to fisheries management and the conservation of kelp forest ecosystems in Central California and elsewhere. We do this in two ways: by evaluating the potential influences of adult fishes and their reproductive products both within the reserves and in nearby non-reserve areas; and by identifying a variety of historical and modern processes that could have led to substantial differences between the measured effects of the existing marine reserves and the structure of pristine ecosystems.

## **Materials and Methods**

Field research was done from 1994 through 1996 in or near three marine reserves (referred to hereafter as areas--Fig. 1) within the Monterey Bay National Marine Sanctuary: Hopkins Marine Life Refuge (all fishing prohibited since 1984); Point Lobos State and Ecological Reserve (designated as a State Reserve in 1963 which prohibited invertebrate collection but allowed commercial and recreational fishing, all fishing prohibited since 1973), and Big Creek Marine Ecological Reserve (all fishing prohibited since January 1994). Non-reserve areas consisted of at least two sites nearby each reserve (Table 1). All sampling was done in habitats with a giant kelp (*Macrocystis pyrifera*) surface canopy at about 14 m depth over rocky substrates with moderate rock relief.

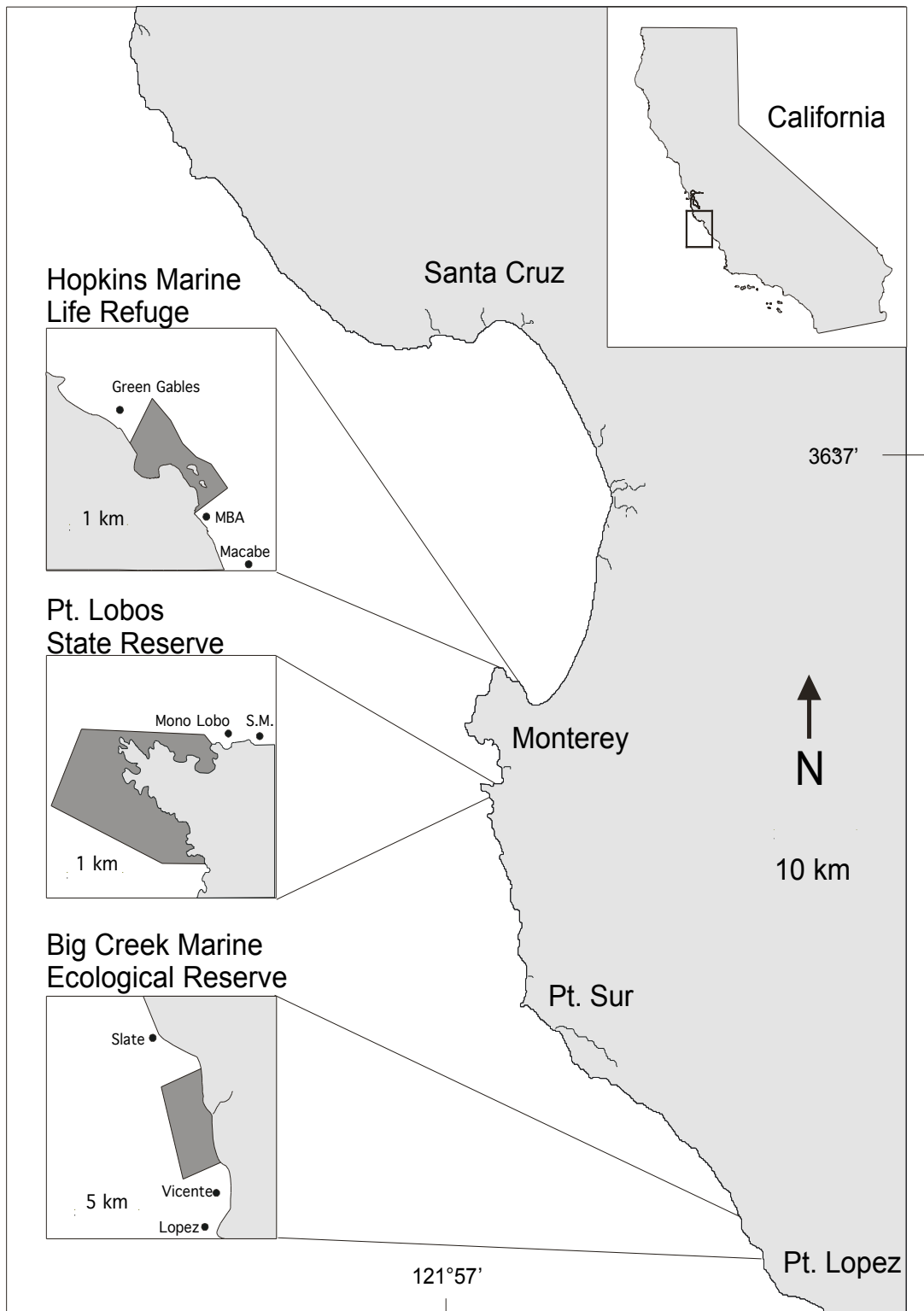


Figure 1. Location of study areas. Non-reserve sites (listed in Table 1) marked as dots. Reserves shaded.

Table 1. List of Sites for each Reserve/Adjacent Non-Reserve (From Paddock and Estes 2000).

| <b>Area</b>                         | <b>Site</b>                 | <b>Location/ Depth Average and Range</b>  |
|-------------------------------------|-----------------------------|---|
| Hopkins Marine Life Refuge          | “Lead line” (HMLR-LD)       | Mid-reserve (off Bird Rocks)<br>9.1 m / 5.5-12.1 m  |
|                                     | Hopkins East (HMLR-E)       | At eastern edge of reserve<br>10.3 m/9.7-11.8 m   |
|                                     | Hopkins West (HMLR-W)       | Western-facing side of reserve<br>10 m/8.8-12.1 m   |
| Hopkins: Adjacent Non-Reserve       | Monterey Bay Aquarium (MBA) | Reef in front of Monterey Bay Aquarium (0.2 km from eastern reserve boundary) 10.6 / 7.6-12.1 m |
|                                     | Macabe Beach                | SW off Macabe Beach (0.9 km from eastern reserve boundary)<br>11.2 m / 7.3-13.3 m               |
|                                     | Green Gables                | In front of Green Gables B&B (0.6 km from western reserve boundary) 12.1 m / 9.1-13.0 m         |
| Pt Lobos Marine Reserve             | Cypress Cove                | toward western point of Reserve<br>12.1 m / 8.5-18.5 m  |
|                                     | Whalers Cove                | near the mouth of Whalers Cove<br>10.6 m / 10.6-17.0 m  |
| Pt Lobos: Adjacent Non-Reserve      | Mono Lobo                   | NE of Whalers Cove (0.2 km from reserve boundary)<br>12.7 m / 12.7-18.2 m                       |
|                                     | South Monastery             | off southern end of Monastery Beach (0.5 km from reserve boundary) 13.6 m / 7.0-17.0 m          |
| Big Creek Marine Ecological Reserve | Big Creek Cove              | Off main beach<br>13.9 m / 11.8-15.2 m  |
|                                     | Square Black Rock           | 1 mile north of cove<br>15.2 m / 8.2-20.9 m   |
| Big Creek : Adjacent Non-Reserve    | Slate Rock                  | Off Esalen (3.2 km from northern reserve boundary)<br>13.6 m / 12.1-23.3 m                      |
|                                     | Vicente Creek               | 1st creek south of Big Creek (1.2 km from southern reserve boundary)<br>13.6 m / 10.3-21.2 m    |
|                                     | Lopez Point                 | south of Vicente Creek (3.2 km from southern reserve boundary)<br>19.1 m / 17.0-21.2 m          |

## *Species*

We focused on fishes that are both common in Central California kelp forests and exploited by commercial and recreational fisheries. This included six rockfish species --*Sebastes atrovirens* (kelp rockfish), *S. carnatus* (gopher rockfish), *S. caurinus* (copper rockfish), *S. chrysomelas* (black and yellow rockfish), *S. nebulosus* (china rockfish), and *S. miniatus* (vermilion rockfish); and four others --*Ophiodon elongatus* (lingcod); *Hexagrammos decagrammus*, (kelp greenling); *Semicossyphus pulcher* (sheephead), and *Scorpaenichthys marmoratus* (cabezon). Population analyses focused on the rockfishes. Young-of-the-year (YOY) were not surveyed as their numbers are seasonal, they utilize different habitats than adults (often recruiting into the giant kelp canopy; Carr 1989), and are difficult to identify to species in the field.

## *Fish counts*

The density of targeted fish species was estimated from fish counts by scuba divers within 50 m x 4 m transects. All transects were located from randomly selected origins and compass bearings. Two divers simultaneously counted the number of fish encountered in a swath 2 m wide and 1 m above the bottom on either side of the transect line.

## *Habitat surveys*

Habitat surveys were conducted to determine the degree of similarity among sites in topography, turf algal species composition and abundance, and kelp density (Fig. 2). This information was in turn used to 1) discern whether habitat variation influenced fish populations, and 2) help assess whether the differing assemblages of exploited species had any influence on their associated ecosystems. Algal abundance and cover were measured because algae provide recruitment habitat for rockfish, shelter fishes from predators, and provide a substrate for prey (Love et al. 1991).

Divers counting fish were followed along the transect by two habitat surveyors. One surveyor measured the percent cover of turf algae by placing a 1 m<sup>2</sup> quadrat at a randomly chosen distance along each 5 m segment of the transect line. Turf algae were categorized as fleshy reds (e.g., *Gigartina* spp., *Gelidium robustum*, *Rhodomenia californica*), articulated corallines (*Corallina* spp., *Bosiella* spp., *Calliarthron* spp.), encrusting corallines (*Lithothamnium* spp. and *Lithophyllum* spp.), and epibenthic browns (*Dictyopteris* spp., *Dictyoneurum californicum*, *Desmarestia ligulata*, *Cystoseira osmundacea*), and the percent cover for each category was visually estimated (Dethier et al. 1983). The second diver counted sporophytes >1 m high of giant kelp (*Macrocystis pyrifera*) and other stipitate brown algae (*Laminaria* spp., *Pterygophora californica*, *Eisenia arborea*) in a swath 1 m to either side of the transect. The second habitat surveyor also classified the substrate in each 5 m segment as being predominately sand, cobble, flat rock, low boulders (<1 m high), medium boulders (1-3 m high), or pinnacles (>3 m high). Bottom depth was recorded at 5 m intervals along the transect line. The relationship between fish density and each of the habitat factors was evaluated using a stepwise linear regression.

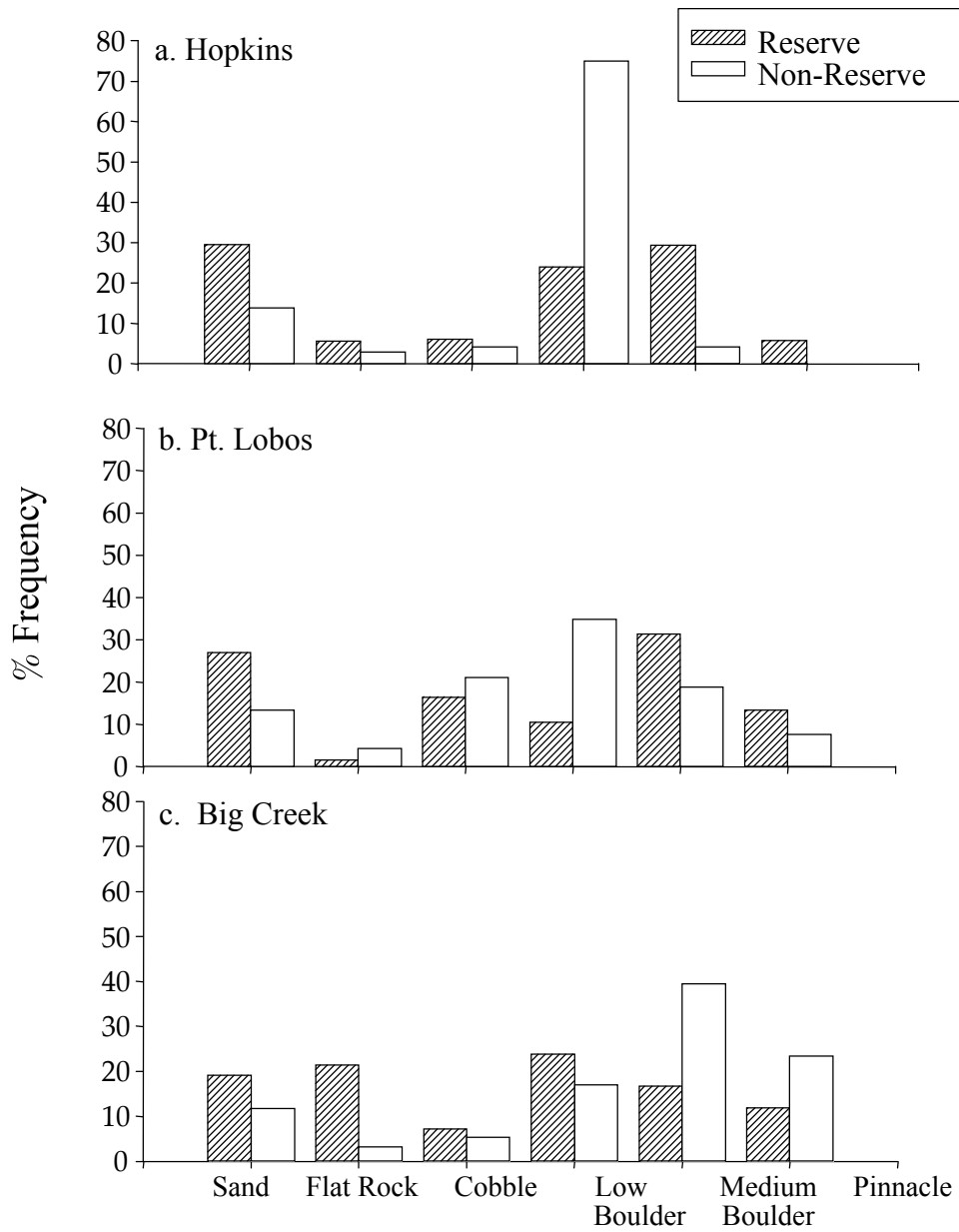


Figure 2. Substrate comparisons between reserve and non-reserve areas.

## *Fish sizes*

Total length of individual fishes was estimated *in situ* to the nearest centimeter with a diver-held Plexiglas ruler mounted perpendicularly at the end of a 1 m long rod (the rod and flexible ruler allowed divers to measure fish in crevices). Transects to measure fish sizes were conducted in the same areas as fish counts, but on different days in order to not bias either the count or size data. Two divers swam a square course starting at the boat's anchor, measuring every targeted fish species that was encountered within 1 m of the sea floor. The accuracy of this method was evaluated by collecting some of the fish with a pole spear in non-reserve areas after they had been measured *in situ*. This was done for 32 individual fish of 3 species (*S. carnatus*, *S. chrysomelas*, and *S. atrovirens*), ranging in total length from 14 to 32 cm. *In situ* measurements of fish length ranged from < 1 cm to > 2 cm actual length (average difference was +0.13 cm  $\pm$  0.98 SD).

Total length (TL) measurements were converted to standard lengths (SL) using SL/TL ratios provided for each species by Lea et al. (1999). A mass-standard length regression, based on >500 individuals of *S. atrovirens* and *S. chrysomelas* collected near Monterey, California between 1984 and 1986 (Larson et al., unpublished data), was then used to convert standard length to biomass. We did not extrapolate these biomass estimates to other species as small differences in length/weight relationships can translate into large differences in biomass for equal numbers of fish. Fish biomass per unit area for each site was calculated by summing the product of mass for each size class of fish, the proportion of fish in the corresponding size class, and fish density.

Length-specific fecundity (LSF) was calculated for *S. atrovirens* and *S. chrysomelas* using relationships with standard length (SL) provided by Romero (1988) and Zaitlin (1986) respectively (*S. atrovirens*-- $LSF=2.1 \times 10^{-5} SL^{4.134}$ ,  $r^2=0.788$ ; *S. chrysomelas*-- $LSF=1.36 \times 10^{-8} SL^{5.59}$ ,  $r^2=0.92$ ). We use the term "reproductive potential" to describe the number of eggs produced per area of habitat by a population. Reproductive potential was estimated for each species and site by summing the product of the fecundity for each size class, the proportion of fish measured in the corresponding size class, and fish density (Table 2). For this computation, fish density was determined by dividing the fish counts by 2, assuming that rockfish have a 50:50 sex ratio (as there is no evidence to the contrary for inshore rockfishes).

Table 2. Total length at which rockfish populations reach 50% sexual maturity (from Wyllie Echeverria 1987) and proportion of fish estimated to be sexually mature from this study (From Paddock and Estes 2000).

| Fish Species          | Size (& age)<br>at 50%<br>maturity | Area      | Reserve                   |             | Non-Reserve      |             |
|-----------------------|------------------------------------|-----------|---------------------------|-------------|------------------|-------------|
|                       |                                    |           | n, size range<br>(TL, cm) | %<br>mature | n, size<br>range | %<br>mature |
| <i>S. carnatus</i>    | 17 (4 yr)                          | Hopkins   | 28, 15-35                 | 96          | 15, 10-30        | 87          |
|                       |                                    | Pt. Lobos | 18, 16-37                 | 94          | 55, 10-35        | 82          |
|                       |                                    | Big Creek | 123, 9-35                 | 93          | 85, 9-32         | 94          |
| <i>S. chrysomelas</i> | 15-16 (3 yr)                       | Hopkins   | 44, 18-36                 | 100         | 29, 11-31        | 88          |
|                       |                                    | Pt. Lobos | 11, 26-35                 | 100         | 25, 10-30        | 92          |
|                       |                                    | Big Creek | 11, 15-34                 | 100         | 6, 13-26         | 83          |
| <i>S. atrovirens</i>  | no data<br>(assume 16 yr)          | Hopkins   | 165, 10-42                | 96          | 100, 10-33       | 89          |
|                       |                                    | Pt. Lobos | 28, 16-39                 | 100         | 42, 17-34        | 100         |
|                       |                                    | Big Creek | 41, 20-45                 | 100         | 39, 16-37        | 100         |
| <i>S. caurinus</i>    | 32-34 (4-6 yr.)                    | Hopkins   | 6, 23-30                  | 0           | 5, 25-30         | 0           |
|                       |                                    | Pt. Lobos | 5, 37-44                  | 100         | 3, 19-26         | 0           |
|                       |                                    | Big Creek | 6, 15-43                  | 50          | 8, 28-43         | 38          |
| <i>S. nebulosus</i>   | 27 (4 yr.)                         | Hopkins   | none                      |             | none             |             |
|                       |                                    | Pt. Lobos | none                      |             | 1, 24            | 0           |
|                       |                                    | Big Creek | 1, 26                     | 0           | none             |             |
| <i>S. miniatus</i>    | 37-38 (5 yr.)                      | Hopkins   | none                      |             | none             |             |
|                       |                                    | Pt. Lobos | 1, 21                     | 0           | 2, 30-35         | 0           |
|                       |                                    | Big Creek | 5, 33-40                  | 60          | 4, 40-48         | 100         |

## Results

### *Habitat analyses*

Frequency distributions of substrate types did not differ within areas between reserve and non-reserve sites (Kolmogorov Smirnov 2 sample tests: Hopkins  $P=0.15$ , Pt. Lobos  $P=0.26$ , Big Creek  $P=0.36$ ). Frequency distributions of percent cover for articulated coralline, encrusting coralline, brown, and foliose red algae varied considerably between each reserve/non-reserve pair (Fig. 3). G-tests showed that 8 of 12 possible pairwise comparisons (3 areas x 4 algal categories) between reserve and non-reserve sites differed significantly. However, there were no consistent trends for any algal class between reserve versus non-reserve areas (Fig. 3). Neither giant kelp (ANOVA,  $F_{1,2}=1.24$ ,  $P=0.38$ ) nor epibenthic stipitate kelp (ANOVA,  $F_{1,2}=3.99$ ,  $P=0.18$ ) densities differed significantly between reserve and non-reserve sites (Table 3). A multiple regression showed significant correlations between articulated coralline algae ( $p=0.01$ ), brown algal ground cover ( $p=0.03$ ), and stipitate kelps ( $p=0.009$ ) and rockfish density. Collectively however, these factors described only 4% of the variation in fish density ( $r^2=0.041$ ).

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Table 3. Mean counts (No. 10 m<sup>2</sup> ±SD) of canopy (*Macrocystis pyrifera*) and understory (*Laminaria spp.*, *Pterygophora californica*, *Eisenia arborea*) kelps. (From Paddock and Estes 2000).

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| Area      | Canopy<br>$F_{1,2}=1.24, P=0.38$ |             | Understory<br>$F_{1,2}=3.99, P=0.18$ |             |
|-----------|----------------------------------|-------------|--------------------------------------|-------------|
|           | Reserve                          | Non-Reserve | Reserve                              | Non-Reserve |
| Hopkins   | 1.6 ± 1.27                       | 1.6 ± 0.71  | 1.2 ± 1.91                           | 0.4 ± 0.62  |
| Pt. Lobos | 1.7 ± 1.77                       | 1.3 ± 0.78  | 5.3 ± 1.39                           | 2.4 ± 2.11  |
| Big Creek | 3.6 ± 1.82                       | 0.7 ± 0.60  | 4.9 ± 3.87                           | 4.4 ± 3.45  |

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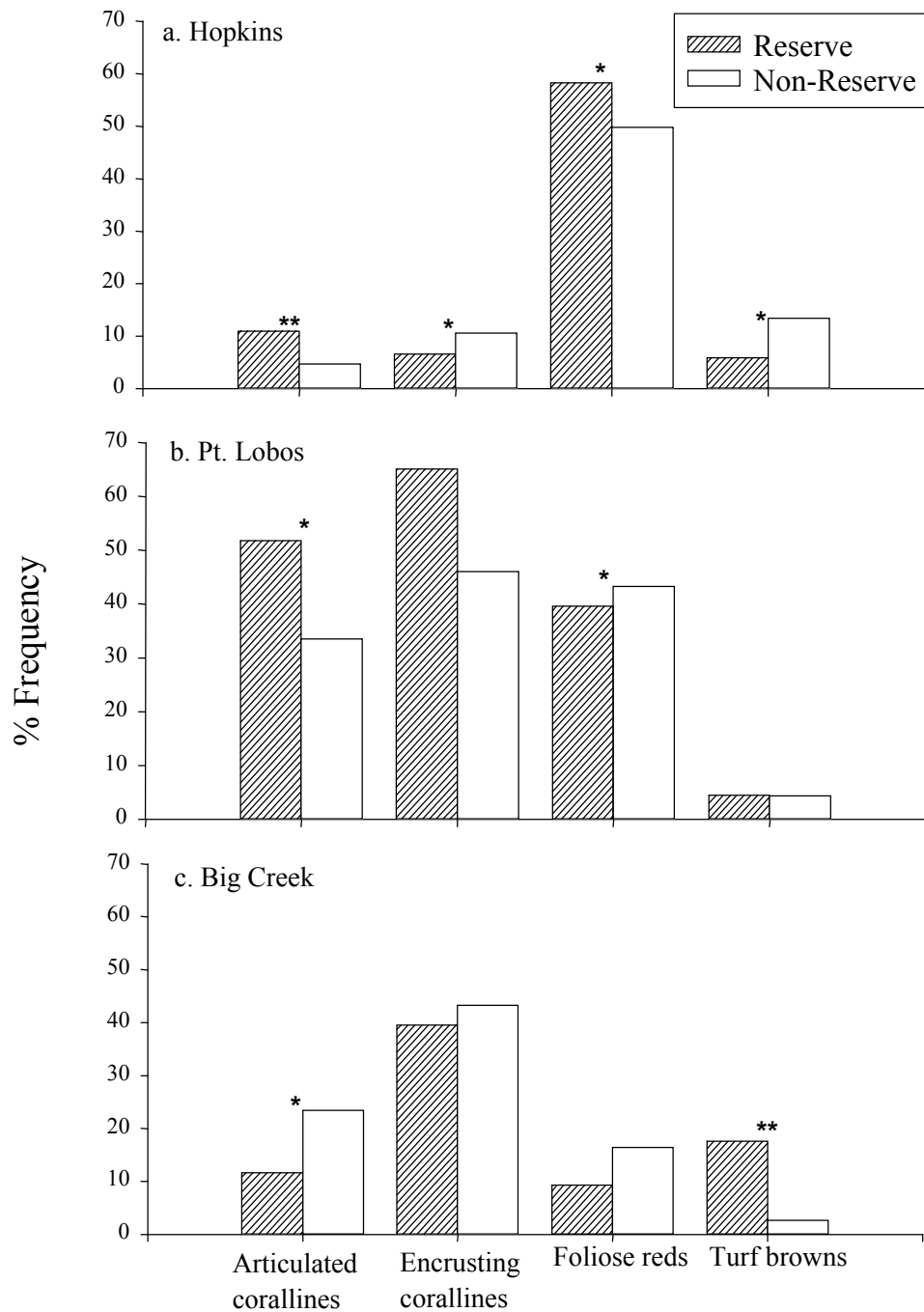
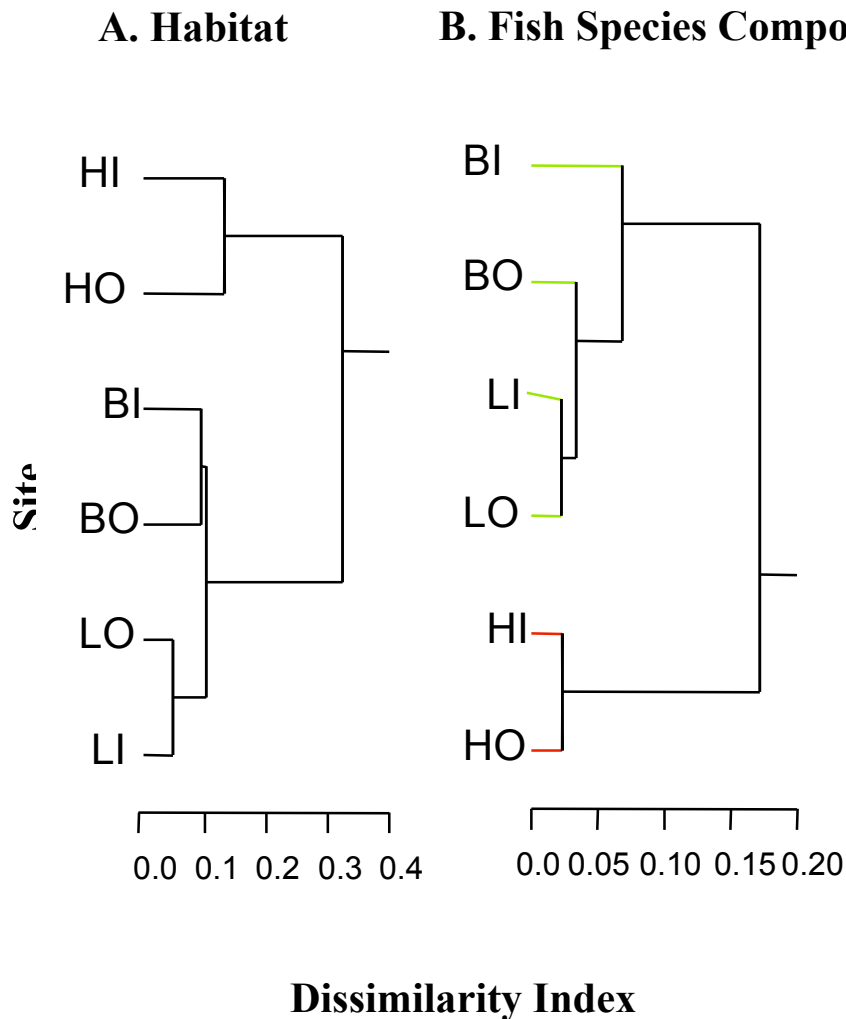


Figure 3. Average percent cover of turf algae in reserve and non-reserve areas for each of four categories (\*\* -  $P < 0.01$ , \* -  $P < 0.05$ ).

### Fish Species Composition

Fish species composition differed somewhat among the three study areas, due to the absence or very low density of certain species at some sites, but this did not appear to be due to reserve status (Fig. 5). Cluster analysis indicated that species composition was most similar between each reserve/non-reserve pair (Fig. 4b). At Pt. Lobos and Big Creek, *S. carnatus* was the most common species whereas *S. atrovirens* was the most common species at Hopkins. *S. nebulosus* was rare, found only in the Big Creek sites and the non-reserve site at Pt. Lobos. *Semicossyphus pulcher* (a labrid fish near the northern end of its geographic range in Central California--Miller and Lea 1972) occurred at Big Creek, was rare at Pt. Lobos, and was absent from our samples at Hopkins.



**Figure 4.** Cluster diagrams based on a) habitat variables and b) species composition. Dissimilarity measure is 1 minus the Pearson product-moment correlation coefficient. (H= Hopkins, L= Pt. Lobos, B= Big Creek, I= Inside reserve, O= Outside reserve).

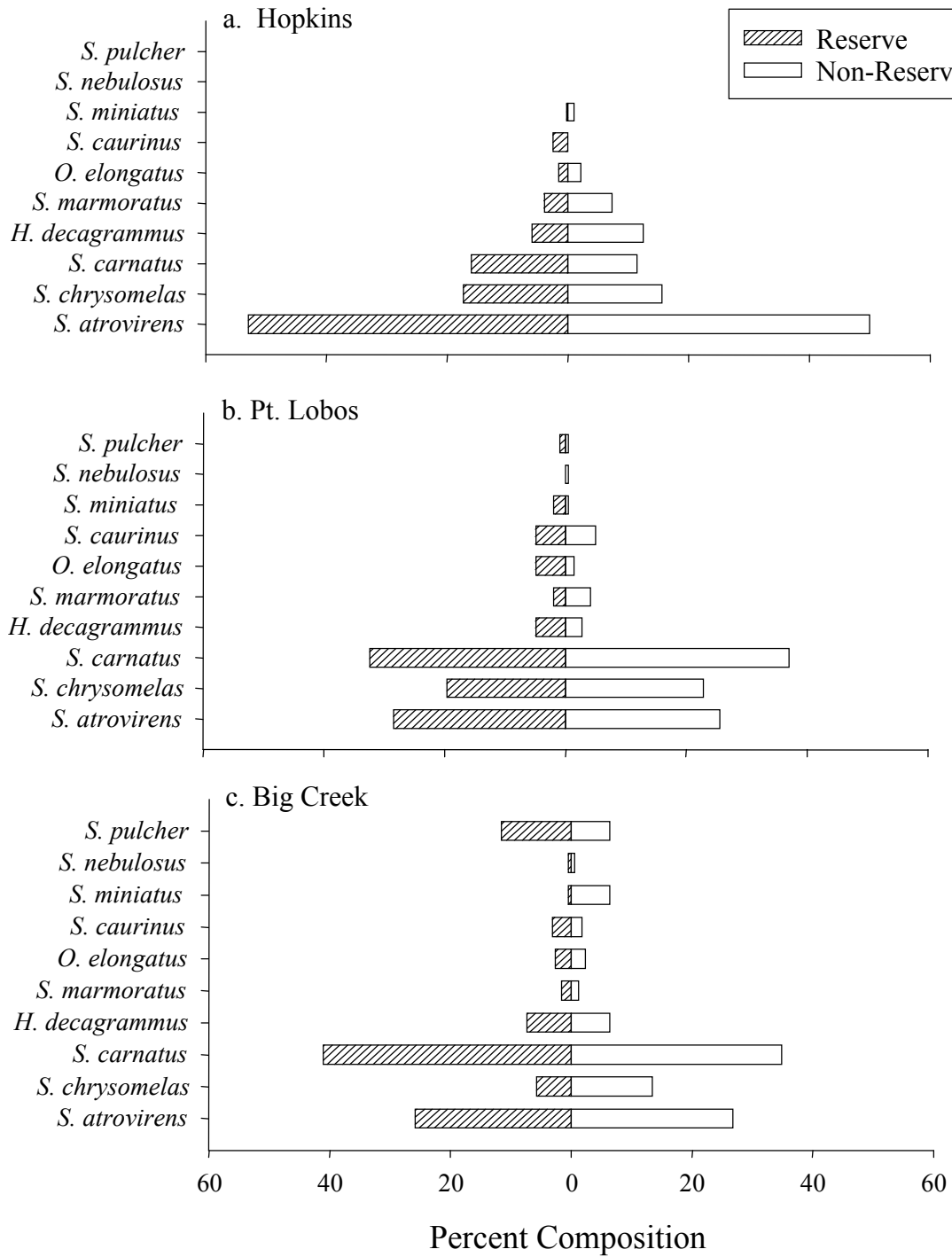


Figure 5. Species composition of surveyed fish species at each reserve/non-reserve site.

## Fish Density

Fish density was 12-35% greater in the reserves than the adjacent non reserves (Fig. 6, Table 4), but these results were not statistically significant (two-way blocked ANOVA,  $F_{1,2}=12.751$ ,  $P=0.07$ ). It should be noted, however, that the power of this test is very low ( $1-\beta < 0.20$  for effect size of 0.12 and  $1-\beta = 0.41$  for an effect size of 0.25).

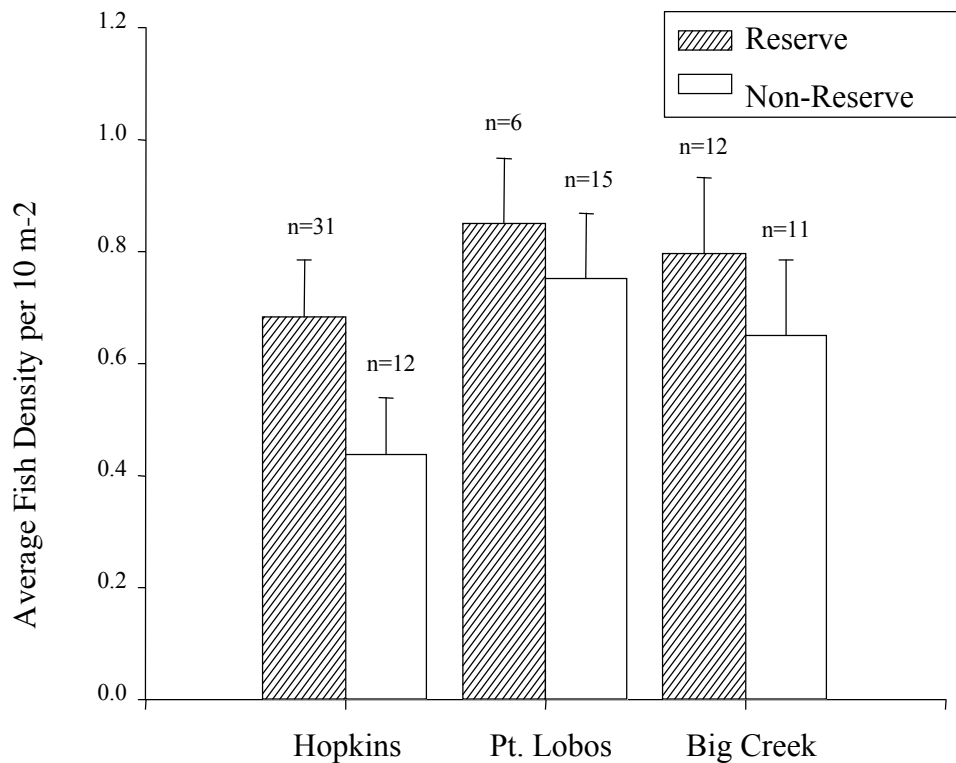


Figure 6. Fish densities (mean  $\pm$  SEM) in reserve and non-reserve areas for all species combined (n=number of transects per area).

Table 4. Density of each fish species (No. 10 m<sup>-2</sup> ± SD). The F statistics below each fish species test for overall reserve effects. (From Paddock and Estes 2000)

| Fish species   | Hopkins Reserve | Non-Reserve | Pt. Lobos Reserve | Non-Reserve | BigCreek Reserve | Non-Reserve |
|--|-----------------|-------------|-------------------|-------------|------------------|-------------|
| <i>Sebastes carnatus</i><br>F <sub>1,2</sub> =3.40, P=0.21         | 0.12±0.17       | 0.04±0.06   | 0.28±0.26         | 0.28±0.22   | 0.33±0.27        | 0.22±0.23   |
| <i>S. chrysomelas</i><br>F <sub>1,2</sub> =0.01, P=0.93            | 0.13±0.13       | 0.08±0.12   | 0.17±0.18         | 0.17±0.13   | 0.05±0.10        | 0.10±0.23   |
| <i>S. caurinus</i><br>F <sub>1,2</sub> =2.87, P=0.23               | 0.02±0.04       | 0           | 0.04±0.07         | 0.04±0.09   | 0.03±0.05        | 0.01±0.03   |
| <i>S. atrovirens</i><br>F <sub>1,2</sub> =8.99, P=0.10             | 0.33±0.41       | 0.23±0.30   | 0.24±0.15         | 0.18±0.30   | 0.20±0.30        | 0.17±0.15   |
| <i>S. miniatus</i><br>F <sub>1,2</sub> =0.34, P=0.62               | 0               | 0           | 0.02±0.03         | 0           | 0                | 0.04±0.04   |
| <i>S. nebulosus</i><br>F <sub>1,2</sub> =0.13, P=0.75              | 0               | 0           | 0                 | 0.003±0.01  | 0.004±0.01       | 0.002± 0.01 |
| <i>Scorpaenichtys marmoratus</i><br>F <sub>1,2</sub> =0.20, P=0.70 | 0.03±0.04       | 0.03±0.04   | 0.02±0.03         | 0.03±0.04   | 0.02±0.04        | 0.01±0.03   |
| <i>Semicossyphus pulcher</i><br>F <sub>1,2</sub> =1.25, P=0.38     | 0               | 0           | 0.01±0.02         | 0           | 0.09±0.19        | 0.04±0.07   |
| <i>Ophiodon elongatus</i><br>F <sub>1,2</sub> =2.60, P=0.25        | 0.01±0.02       | 0.01±0.02   | 0.04±0.06         | 0.01±0.02   | 0.02±0.05        | 0.02±0.02   |
| <i>Hexagrammos decagrammus</i><br>F <sub>1,2</sub> =2.18, P=0.28   | 0.04±0.06       | 0.05±0.06   | 0.04±0.06         | 0.02±0.03   | 0.07±0.07        | 0.04±0.06   |

### *Population Structure*

Size-frequencies were combined within sites for those rockfish species having similar maximum sizes (i.e., *S. carnatus*, *S. chrysomelas*, *S. atrovirens*, *S. caurinus*; Fig. 7). For the two areas protected the longest (Hopkins, 12 yr; Pt. Lobos, 23 yr), average lengths were significantly greater in reserves than non-reserves (Hopkins-- $t=9.29$ ,  $df=390$ ,  $P<0.001$ ; Pt. Lobos-- $t=7.10$ ,  $df=191$ ,  $P<0.001$ ). Mean lengths at Big Creek (protected for 1 yr. at the time of our study) did not differ significantly ( $t=0.51$ ,  $df=328$ ,  $P=0.304$ ). Kolmogorov-Smirnov two sample tests for each reserve/non-reserve comparison also showed significantly different length-frequency distributions (i.e., population structures) at Hopkins ( $P<0.001$ ) and Pt. Lobos ( $P<0.001$ ), but not Big Creek ( $P=0.99$ ). Pooled Kolmogorov-Smirnov tests resulted in a significant difference between reserves and non-reserves overall ( $P=0.0002$ ). The populations of non-reserve fish at Hopkins and Pt. Lobos were dominated by small size classes, whereas at Big Creek the size class distributions were similar between reserve and non-reserve sites (Fig. 7). This pattern held for each of the common rockfish species.

### *Biomass*

Biomass density (g fish  $10\text{ m}^{-2}$ ) estimates for *S. atrovirens* and *S. chrysomelas* at both Hopkins and Pt. Lobos were  $>2$  times higher in the reserve than non-reserve sites while at Big Creek there was no discernible difference between the reserve and non-reserve sites (Fig. 8).

### *Reproductive Potential*

While the length-fecundity equations are power functions typical of mass-length allometries (Zaitlin 1986, Romero 1988), they diverge considerably between *S. atrovirens* and *S. chrysomelas* for fish  $>20$  cm standard length. Based on these length-fecundity relationships and size frequency distributions, estimated reproductive output for both species was greater in the reserve than the non-reserve sites at Hopkins and Pt. Lobos, but similar at Big Creek (Fig. 9). The comparatively low estimates for Big Creek result from the general lack of larger individuals in both reserve and non-reserve sites.

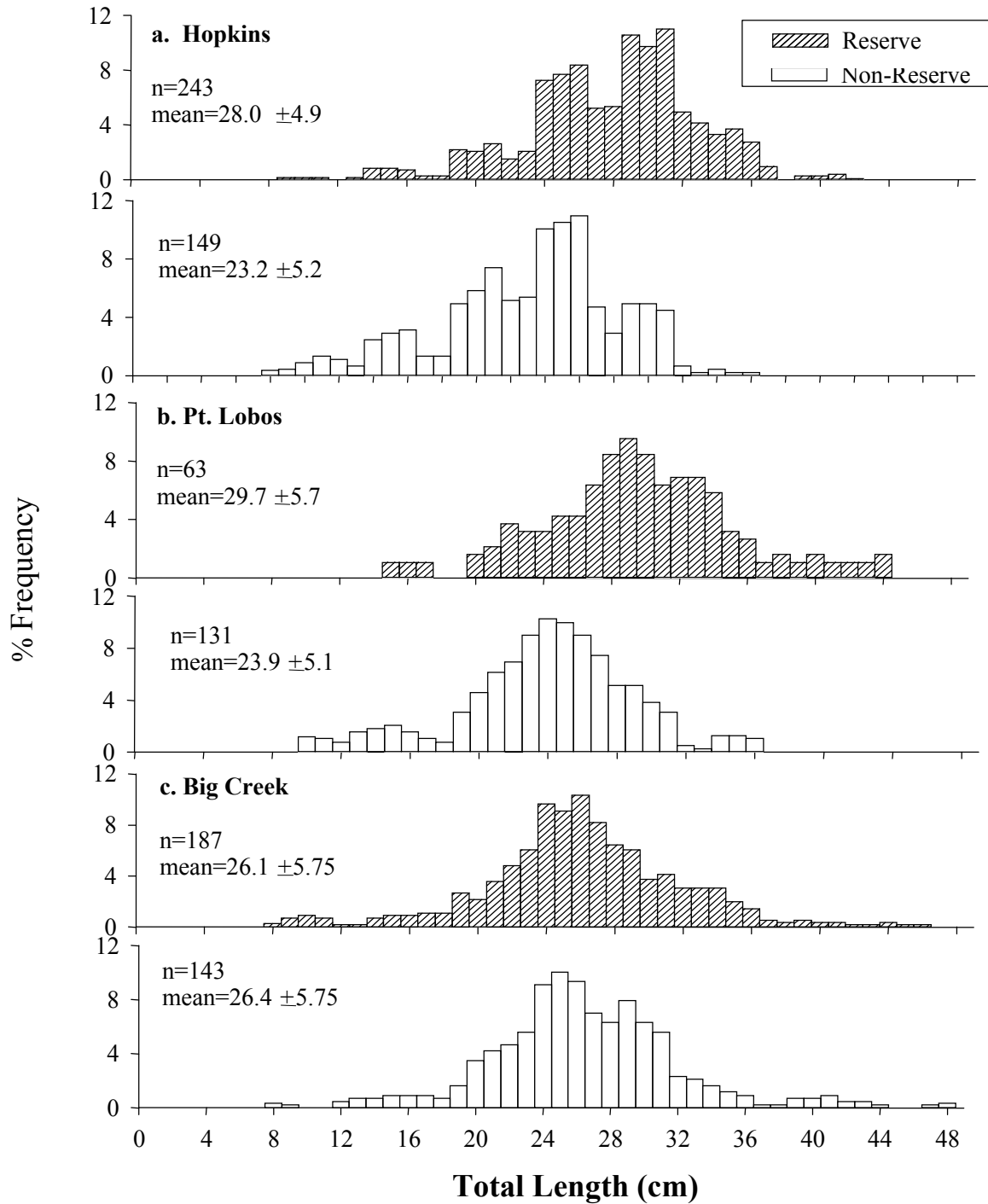


Figure 7. Length frequency distributions of rockfish species combined (*S. atrovirens*, *S. chrysomelas*, *S. carnatus*, *S. caurinus*) in the reserve and non-reserve areas at Hopkins, Pt. Lobos, and Big Creek.

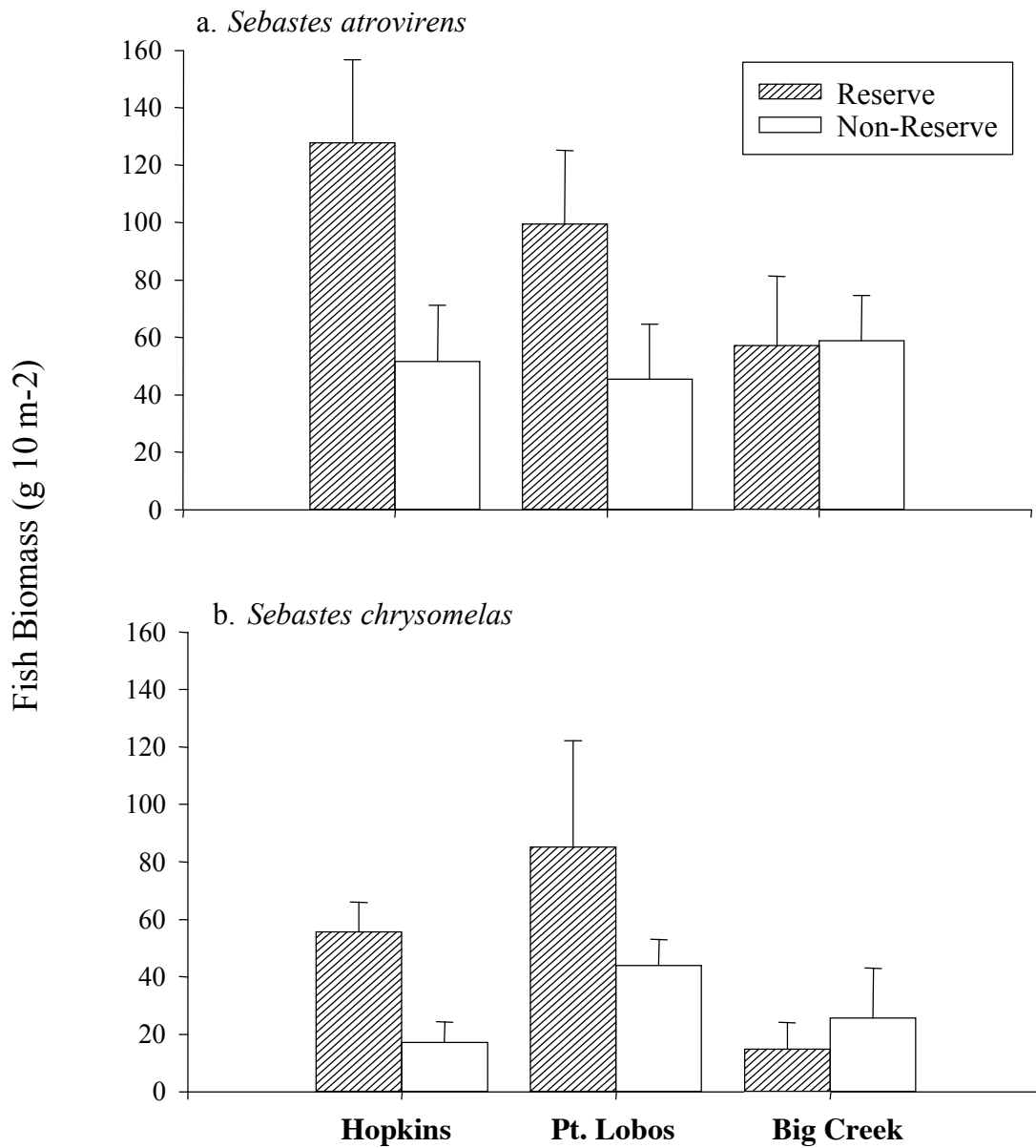


Figure 8. Average biomass ( $\pm$ SEM) per unit area (10 m<sup>2</sup>) for *S. atrovirens* and *S. chrysomelas* in reserve/non-reserve areas.



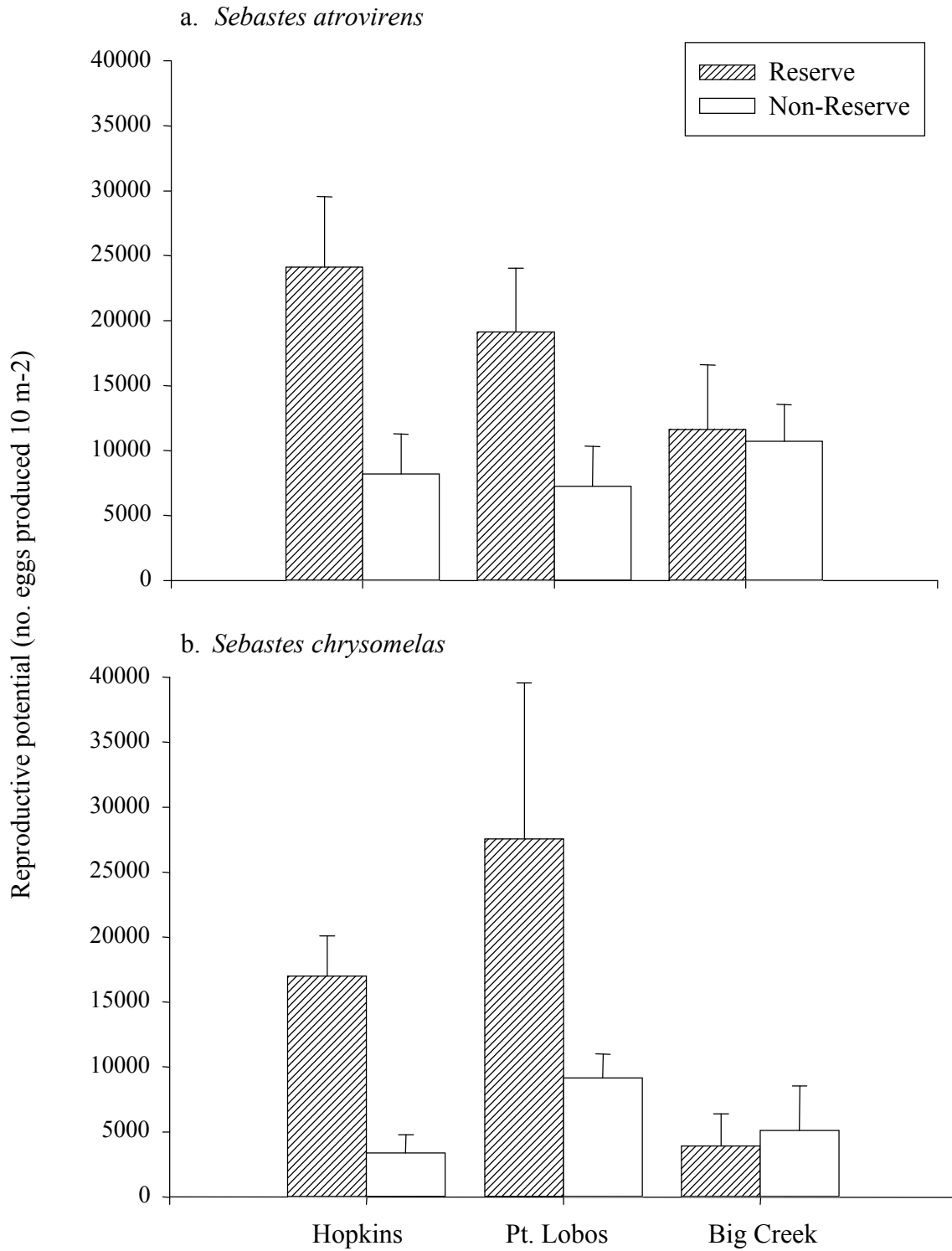


Figure 9. Estimated reproductive potential (mean  $\pm$  SEM) for *S. atrovirens* and *S. chrysomelas* in reserve and non-reserve areas.

## Discussion

Our findings, like those of so many others who have studied the influences of marine reserves (National Research Council 2001, Halpern in press), provide a clear but surprisingly subtle sign of human impact. Are these data an accurate metric of human impact or are they only showing us the tip of the proverbial iceberg? They might represent the “tip of an iceberg” if 1) reserves are too small to allow buildup of biomass or retain viable populations of rare species, 2) high spillover occurs from reserves to non-reserves, 3) recovery in the reserves from overexploitation is still ongoing, 4) poaching is a significant factor, or 5) earlier impacts of human exploitation resulted in large-scale changes in ecosystem function that influence reserve and non-reserve areas similarly.

The possibility that the reserves we studied are too small to capture the full influence of human take is especially worthy of consideration. Conservation biologists have been quibbling over optimum reserve size for decades (e.g., the SLOSS debate; Diamond 1975, Simberloff and Abele 1976). A renewed interest in this topic has led some terrestrial ecologists and conservation biologists to conclude that small reserves are inadequate to preserve biodiversity, owing largely to a growing realization that top-down forcing processes are important in many natural systems (Pace et al. 1999, Estes et al. 2001) and large areas of habitat are necessary to maintain viable populations of large, apex predators (Soulé and Terborgh 1999). For instance, when home range size is large relative to reserve size, a species’ probability of extinction within a reserve or protected area may be high (Woodroffe and Ginsberg 1998). Furthermore, if their normal movements frequently take individuals beyond a reserve boundary and the risk of being caught in that area is high, stock increases within the reserve are unlikely to realize their full potential. Small or isolated reserves also are more likely to depend upon external sources of larvae, making them vulnerable to recruitment over-fishing in heavily fished areas (DeMartini 1993, Jennings et al. 1996, Roberts 1997, Carr and Raimondi 1998).

The sizes of most coastal marine reserves worldwide are miniscule compared with their terrestrial analogues, and the reserves we studied are no exception (Carr et al., in press). Might larger reserves show more dramatic or even qualitatively different effects of human exploitation than their small counterparts? Halpern (in press) surveyed the literature on marine reserves and found no evidence for an effect of reserve size on species diversity, density of exploited populations, or trophic disfunction. While this study was based on an impressively long list of case studies, an effect of reserve size could be masked by profound and large-scale effects of historical overfishing (Jackson et al 2001) and the extremely small size of most existing marine reserves. Thus, even the largest marine reserves may be too small to capture a reserve size effect. This possibility seems especially likely to us, given that historical overfishing targeted the large apex predators (Pauley et al. 1998), the spatial scale of key forcing processes is greater in the sea than it is on land, and nearly all marine reserves are very small compared with their terrestrial counterparts.

Several studies have shown increased fish abundance in exploited areas nearby marine reserves, a phenomenon attributed to adult spillover from the reserves (Polunin and Roberts 1993, Attwood and Bennett 1994, Russ and Alcala 1996, Wantiez et al. 1997). This effect also may decrease rockfish density differences between exploited and protected areas by enhancing

populations in exploited areas and reducing populations in protected areas. While there is some evidence that rockfish move from areas of high to low population density (Matthews 1985), they also have been shown to have small home ranges (Miller and Geibel 1973, Larson 1980). Rockfish density and biomass also may be slow to recover from fishing pressure (Holland and Brazee 1996, Russ and Alcala 1996, Gunderson 1997, McClanahan 1997a) as these species are typically long-lived, slow growing, and late to mature (Love et al. 1990, Leaman 1991). Poaching, which undoubtedly occurs in Central California, may also help to explain the modest differences in fish populations between reserves and adjacent protected areas.

The length-frequency data from our study provide additional and statistically stronger support for rockfish population differences between protected and unprotected kelp forest sites. Increased fish sizes within reserves also have been demonstrated in many other studies (Roberts and Polunin 1993, Dufor et al. 1995). But here again, do these patterns properly reflect the full effect of human impact on fish size or might truly unexploited populations contain much larger fish? Examples of fish from aboriginal midden sites that are far larger than those known from modern or more recently historic populations seems to support this latter possibility. Because reproductive output often scales to fish size and biomass, this same caveat applies to our calculations of reserve effects on the supply of young fish.

The perception of understanding human impacts on the kelp forest ecosystems of Central California by contrasting reserve/non-reserve areas must be further tempered by other sources of uncertainty. One of these is the lack of any real historical information on the structure and function of these systems before the time of human contact. This is especially important for the New World, which was not peopled until about 14,000 ybp. The influence of human exploitation on coastal marine systems is more extreme and has a deeper history than many people realize or even care to admit (Jackson et al. 2001). Worldwide, spectacularly abundant populations of large animals inhabited coastal waters and in many cases exerted important functional influences on these systems. Many of these large animals have been so depleted that their ecological roles are extinct, even if the species themselves still survive. Central California kelp forests lack Steller's sea cows (*Hydrodamalis gigas*), a large herbivorous mammal that was common in the area until late in the Pleistocene and may have exerted an importance grazing influence on kelp forests (Estes et al. 1989). Sea otters (*Enhydra lutris*) are another functionally important species in Central California kelp forests although they have arguably recovered to historic levels in Monterey Bay. The pervasive influence of sea otters as predators on benthic invertebrates, and the cascading effects of this interaction on kelp populations (Estes and Palmisano 1974) may explain the absence of any evident top-down influence of the marine reserves in our study as the otters forage across the reserves and their adjacent non-protected sites. Nonetheless, other important consumers, especially the larger coastal fishes, are probably reduced or absent compared with earlier times. The functional ghosts left by these species may have preconditioned their associated ecosystems for further change, which we now consider natural (Dayton et al. 1998, Jackson 2001). An even greater potential for change through time relates to the fact that many natural systems undergo catastrophic phase-shifts due to non-linearities in the forcing functions (Scheffer et al. 2001). Phase shifts of this nature are known to occur in kelp forest ecosystems (Konar and Estes in press) and probably take place in other coastal marine ecosystems as well (Petraitis and Dudgeon 1999). A highly significant consequence of this

dynamic is that small disturbances can cause large change while large counter-disturbances to the altered system may result in no effect at all (May 1977, Scheffer et al. 2001).

In sum, while our research provides unequivocal evidence for both a detrimental effect of human take on marine resources of the Monterey Bay National Marine Sanctuary and thus a beneficial influence of marine reserves on these same resources, these findings may mean more than is apparent. The extremely small size of these reserves, their relatively young age, and uncertainties relating to both an appropriate historical baseline and the chaotic features of ecosystem behavior might very well render as grossly incorrect the naïve and simplistic view that pristine populations and ecosystems have somehow been re-created by removing the influence of human exploitation from our coastal marine reserves.

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## CHAPTER 2

### **Baseline *In Situ* Surveys And Landing Creels Of Nearshore Fishes Within The Vicinity Of Big Creek Ecological Reserve**

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#### **Abstract**

The decline in the success and quality of Central California's nearshore rockfish fisheries over the past decades and the rapid expansion of the live/premium fish fishery, which targets shallow water kelp forest fishes, have underscored the need for a revised management strategy for this resource. Marine protected areas have been suggested as an alternative or additional management tool to create sustainable fisheries. The overarching goal of the study is to provide a benchmark or "line in the sand" of data collected in a newly established marine protected area. Because both fishery independent and dependent data are crucial in the evaluation of the reserve as a fishery management tool, we collected data from *in situ* surveys of fish populations within and adjacent to the Big Creek Ecological Reserve (BCER) and monitored the landings of commercial and recreational fisheries targeting fishes in the vicinity of BCER.

We examined numbers of all fish observed during random transects conducted within and adjacent to BCER. Counts north of BCER in 1995 and 1998 were significantly different (41% declined); however, counts within and south of BCER were not statistically different between 1995 and 1998. A high percentage of 1998-99 Big Sur commercial skiff fishery landings were composed of cabezon, and black-and-yellow and gopher rockfishes. This study documented that 4.5 times more Commercial Passenger Fishing Vessels (CPFV) trips were made to the Cape San Martin region as were made to the Point Sur region. Most species taken in the Point Sur region were larger than those taken in the Cape San Martin region; however, there was not a significant difference in the CPUE between the two areas.

## Introduction

Marine reserves, established as a fisheries management tool, are intended to manage exploited stocks by protecting populations of sexually mature species vulnerable to overfishing and thus insuring a continual source of new recruits. Reserves have been reported to enhance fisheries in other parts of the nation and world; however, limited information is available to evaluate their effectiveness on California's sport and commercial rockfish fisheries. Rockfish are an important and heavily exploited component of sport and commercial fisheries in California. The estimated value of the sport fishery to California's economy is about one billion dollars annually (Lenarz 1987), while the annual ex-vessel value of commercial fisheries for rockfish exceeds ten million dollars (CDFG unpubl. data). In 1995, California Commercial Passenger Fishing Vessels (CPFV) reported landing 1,174,991 rockfish; 62% were landed in Central California (CDFG unpubl. data).

The success and quality of Central California's nearshore rockfish sport fisheries have declined over the past decades, particularly in areas close to ports (Miller and Gotshall 1965, VenTresca and Lea 1984, Reilly et al. 1993, Karpov et al. 1995, Mason 1995). In the northern and Central California sport fishery, between the late 1950s and early 1980s, the average weight per rockfish declined by 13 percent, or 0.43 pounds per fish; average weight decreased for 12 of 16 major rockfish species (Karpov et al. 1995). In Monterey Bay, from 1987 to 1991, the average length of several rockfish species sampled in the CPFV fishery was below the average size at 50% sexual maturity (Reilly et al. 1993 and Reilly et al. 1998). Presently, fishery managers are concerned with rapid increase in fishing effort and landings of the emerging commercial live/premium fish fishery that has developed coast-wide for rockfish, cabezon, lingcod, and kelp greenling during recent years. The rapid expansion of this fishery is of greatest concern in the nearshore, shallow, rocky habitats where many of these species are concentrated. As stocks proximal to port become depleted, fishers have expanded their range further from port and into deeper waters. Many of these nearshore areas, especially along the Big Sur coast, had received little fishing pressure and until recently had functioned as *de facto* reserves.

Limited movement, relatively long life span, and late maturity of rockfishes, coupled with heavy sport and commercial fishing pressure, has resulted in removal of many mature fish from reefs proximal to fishing ports (Lea et al. 1999). Consequently, commercial and sport fishers are traveling farther from port to maintain a quality catch. Once large reproductive adults are removed, continued fishing pressure prevents remaining fish from reaching the size at which sexual maturity occurs. Reproductive potential is also affected by natural phenomena, such as an El Niño event. During these events growth rates and survival are often reduced (Bailey and Incze 1985, VenTresca et al. 1995).

Recognizing that a change in the current resource management approach is warranted, alternative regulatory and enhancement strategies are of paramount importance. Rockfish appear to be an excellent candidate for enhancement using a reserve management approach because in addition to being residential and long-lived they are extremely fecund and have a lengthy larval stage. Reserves, utilized as a rockfish management tool, would protect critical spawning stock biomass and ensure a continual recruitment supply to fished areas via larval dispersal and emigration. However, to determine the effectiveness of a reserve to enhance rockfish fisheries,

baseline information is first needed to demonstrate that densities and sizes of fish populations within the reserve are greater than adjacent fished areas and that absolute numbers of species of concern within the reserve are sufficient to provide a reasonable/significant source of larvae.

Marine fisheries reserves offer potential benefits to coastal fisheries and marine resource management, including: the enhancement and restoration of fishery yields through larval/egg transport; "spillover" from reserves to surrounding areas; protection of reproductive potential of targeted species; maintenance of biological diversity; and increased social and economic benefits to local communities (Russ 1985, Davis 1989, Bohnsack 1990, Roberts and Polunin 1991, Rowley 1994, Roberts et al. 1995). Additionally, reserves may increase the resiliency of heavily exploited fisheries during episodic environmental anomalies by serving as a buffer against drastic declines and hastening the recovery of these fisheries by supplying harvested stocks with new recruits (Carr and Reed 1993). Although reserves provide an excellent opportunity for *in situ* testing of concepts related to the protection and enhancement of fish populations, studies demonstrating a significant increase in fish population densities and average length within a reserve are few (Rowley 1992, Dugan and Davis 1993, Palsson and Pacunski 1995, Roberts et al. 1995, Russ and Alcala 1989, and Paddock and Estes 2000). Studies showing enhancement or "spillover" are also limited but are probably most critical in convincing public user groups of the value of reserves (Rougharden and Iwasa 1986, Battershill 1993, Rowley 1992).

Although marine reserves have been reported to enhance fisheries in other parts of the nation and world, limited information is available to evaluate their effectiveness on California's sport and commercial rockfish fisheries. Because improperly designed reserves may endanger a fishery by providing a false sense of protection, determining the effectiveness of a reserve is of utmost importance (Carr and Reed 1993, Yoklavich 1998). The establishment of the Big Creek Ecological Reserve (BCER) in Monterey County in January 1994 provided an excellent opportunity to evaluate the effects of a reserve on the nearshore rockfish resource; however, without first obtaining baseline information on species composition, densities, and length frequencies of rockfish populations within and adjacent to BCER, determining change in population parameters or future benefits to adjacent and distant fisheries will be difficult at best.

Rockfishes are an excellent candidate species group for this study. They are long-lived, residential, fecund, and have an open ended reproductive system that distributes offspring spatially. Resident populations of large sexually mature rockfishes within the reserve would be a continual source of larvae that in turn would be carried via nearshore currents to other sections of the coastline (Chelton et al. 1982, Wyllie-Echeverria 1987, VenTresca et al. 1996). A future management strategy might include a system of coastal reserves that would reinforce and enhance the reproductive strategies of rockfish.

Baseline information on species composition, densities, and size frequencies of rockfish populations within and adjacent to BCER are crucial to evaluate the effectiveness of this reserve. In addition, resource managers will be unable to evaluate marine reserves as an alternative management tool for rockfish. Furthermore, assessing the status of nearshore fish populations/stocks in the vicinity of BCER is timely and relevant due to the recent dramatic increase in commercial hook-and-line landings of nearshore species throughout California. The extensive kelp forests along the Big Sur coast, until recently were one of the last remaining

unexploited marine habitats in California. Now this area is being intensely fished to provide for the increasing demand in the lucrative live fish market. As fishing effort increases on these formerly pristine stocks, it becomes imperative to evaluate the effectiveness of a Central California coast reserve to protect and enhance rockfish stocks.

Most studies of marine protected areas have focused on assessing changes in adjacent fisheries or changes in the population structure of fishes within the reserves and/or adjacent areas. Both of these sources of information are crucial in the evaluation of the reserve as a fishery management tool. We monitored both the landings of a commercial skiff fishery and the CPFV fisheries that target fishes near BCER. We also used random transects and permanent stations sampling of *in situ* fish population of fishes within and adjacent to BCER. The overarching goal of this study is to provide a benchmark or “line in the sand” of data collected on a newly established marine protected area. Our hope is that we have done an adequate job so that future studies can statistically document temporal changes in nearshore fish populations within and adjacent to BCER.

## Methods

Population parameters of selected fish species in nearshore habitats within and adjacent to BCER were assessed and analyzed utilizing two approaches: 1) *in situ* subtidal-reef fish population data were collected using standardized scuba techniques in summer/fall 1995-98 and 2) species composition, catch-per-unit-effort (CPUE) and length frequencies of fishes taken in commercial skiff and CPFV fisheries in the areas adjacent to BCER were collected from fall 1997 through winter 1999.

### *In Situ Fish Density Estimates*

The BCER subtidal study site, located 50 miles south of Monterey, Monterey County, was divided into three areas: 1) BCER, which encompasses 2.5 na. mi. of coastline from Oyster Catcher Point in the south, northward to Rat Creek; 2) north of BCER from the northern BCER boundary, northward to Slate Rock, a distance of 4 na. mi.; and 3) south of BCER from the southern BCER boundary, southward to Lopez Point, a distance of 4 na. mi. (Figure 1). Total number of fish counted, i.e., the sum of the counts of both scuba divers surveying the transect were use for the analysis. Total area surveyed by each transect was 320 m<sup>2</sup>.

Optimal conditions for obtaining *in situ* visual density estimates of subadult and adult fish populations using scuba along the Central California coast occur after the spring plankton blooms during the summer, and before winter storms in the fall (personal observation). Due to the remote location of the Big Creek study area and in an attempt to collect a high number of observations during similar conditions of underwater visibility and surge, the Department’s *R/V MAKO* was utilized for 2-3 week survey cruises. The *R/V MAKO* provided a platform to accommodate 7-8 divers and an air compressor to fill scuba tanks. Divers made 3-4 dives per day, commencing after 0900 hours to optimize the underwater visibility.



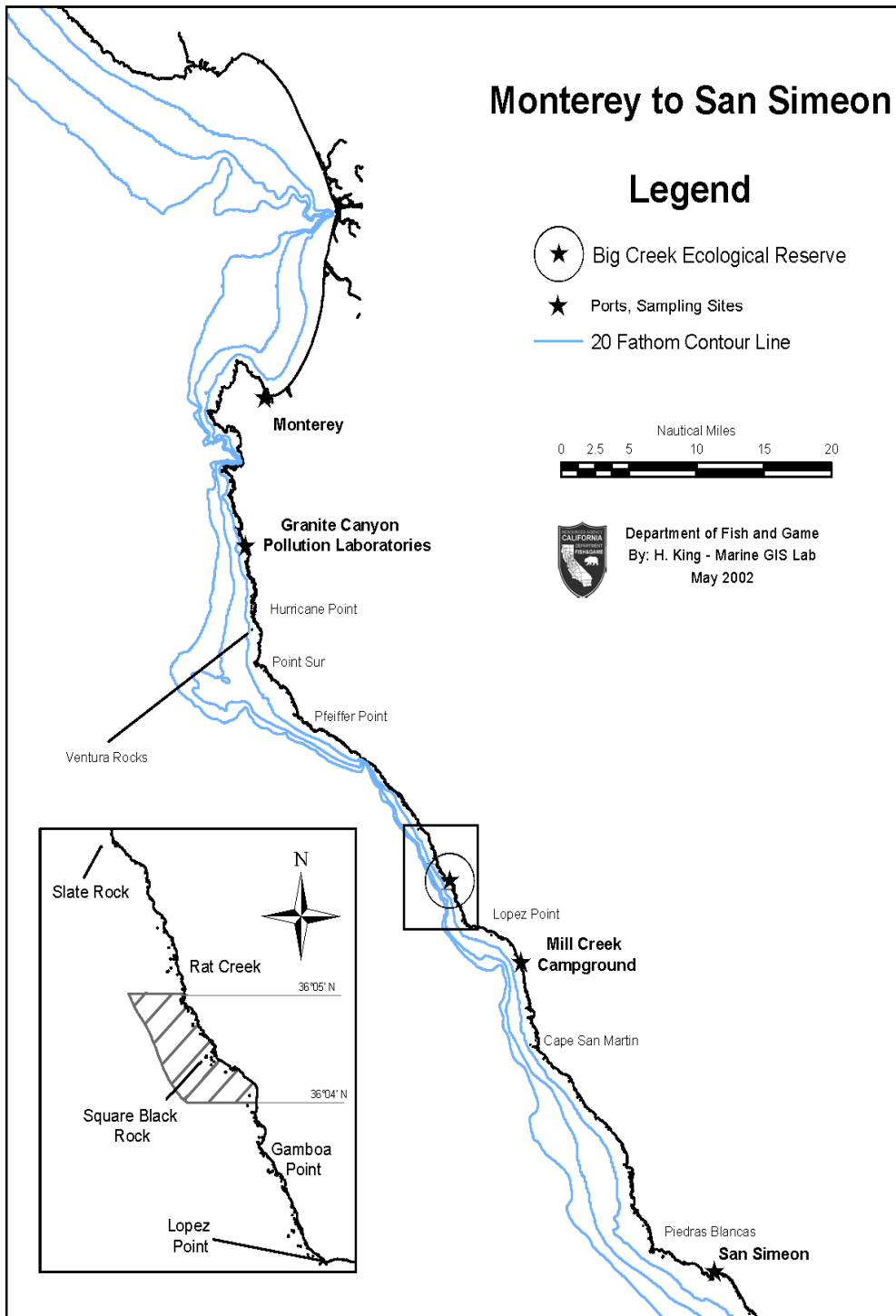


Figure 1. Commercial Passenger Fishing Vessel and Commercial Skiff Fishery fishing sites and scuba survey area (in vicinity of Big Creek Ecological Reserve), Monterey County.

A survey team consisted of two divers, each equipped with standard personal scuba gear, a plastic slate that securely held data recording sheets and a 10-m retractable transect (Ugortez et al. 1997). Divers were trained in underwater species identification and assessment of size of sexual maturation. Scuba transects were deployed in 10-20 m depths in kelp forest habitats. Target fishes for this study were nearshore subadult and adult fish species that are harvested commercially and/or recreationally in Central California nearshore rocky bottom areas. These species include: *Sebastes atrovirens* (kelp rockfish), *Sebastes carnatus* (gopher rockfish), *Sebastes chrysomelas* (black-and-yellow rockfish), *Sebastes caurinus* (copper rockfish), *Sebastes nebulosus* (China rockfish), *Sebastes melanops* (black rockfish), *Sebastes miniatus* (vermillion rockfish), *Sebastes serranoides* (olive rockfish), *Hexagrammos decagrammus* (kelp greenling), *Ophiodon elongatus* (lingcod), and *Scorpaenichthys marmoratus* (cabezon). All of these species are readily quantifiable using benthic scuba transects. Several infrequently observed species such as: California sheephead (*Semicossyphus pulcher*), kelp bass (*Paralabrax clathratus*), treefish (*Sebastes serriceps*), and wolf-eel (*Anarrhichthys ocellatus*) were also recorded when observed.

### *Landing of Commercial Skiff Fishery*

Commercial skiff fishery landings along the Big Sur coast were monitored and sampled during 1997-1999. Sampling procedures were, in part, based on criteria developed by the California Cooperative Commercial Groundfish Survey (Thomas et al. 1995). We interviewed commercial skiff fishermen as they returned from fishing, on the beach at Mill Creek State Park or BCER, Monterey County (Figure 1). For each returning fishermen, the sampler recorded date, port of landing, boat number, gear, estimated sampled weight and total weight of catch, CDFG Block number, site-specific fishing location, time, and depth fished. Because fishermen wanted to transport the live fish to market as soon as possible they did not permit the sampler to weigh the catch with a scale. Fishers used plastic laundry baskets and 5-gallon buckets to transport live and dead fish, respectively, from their skiffs to cars. The average weight of fish in a full basket was 65-75 pounds and 30-35 pounds in a full 5-gallon bucket. Fishers and the sampler estimated total weight and sampled weight of catch independently. Agreement between estimated weights was good. Fish in the sample subset were individually identified to species, their total length was measured on a measuring board, and it was noted whether they were dead or alive.

### *Catch of CPFV Fishery*

Sport angler catch aboard CPFVs departing from San Simeon Cove, San Luis Obispo County, was sampled to obtain species composition and size data from remote areas along the Big Sur coast. Commercial passenger fishing vessels from San Simeon occasionally conducted 1- and 2-day long-range sport fishing trips that fished from Piedras Blancas north to Point Sur and encompassed the vicinity of BCER (Figure 1). In 1997-98, a sampler was placed on these trips, space and weather permitting, to observe and record fish caught, using sampling methodology established in 1987 by the Department of Fish and Game's (CDFG) Central California Marine Sport Fish Survey (Reilly et al. 1998). This effort provided a cost-efficient supplementary database of fish sizes and fishing effort of nearshore sport fishes in the vicinity of BCER, an area that historically has received less sport and commercial fishing pressure than areas proximal to Monterey and Morro Bay.

Samplers were trained in marine fish species identification and were equipped with foul weather gear, gloves, clipboard, waterproof data sheets, fish-length measuring board, lead pencils, and field guides to California marine fishes. The sampler contacted the landing to secure passage on the vessel prior to the trip. CDFG vessel number, port code, departure time, type of fishing trip (offshore, nearshore, surface, bottom, mix), number of paid and free anglers, and type of fishing tackle used were recorded on a standard sampling form.

When the vessel arrived at the first fishing location, the sampler chose a subset of anglers to observe throughout the trip and recorded this number (usually less than 15). The sampler recorded bottom depth, the time when fishing lines were lowered, the number of observed and total anglers, and either latitude and longitude, LORAN coordinates, or compass bearings and coastline features. When the last observed fishing line was raised, signifying the end of a "drift", time and depth were recorded and the process was repeated throughout the day.

Samplers observed anglers in the stern half of the vessel, where a larger sample size could be obtained. An assumption in this sampling methodology, proven statistically in 1993 (Wilson-Vandenberg et al. 1996), was that catch, effort, and CPUE data from observed anglers in the stern of the vessel were representative of all anglers on the vessel. Samplers identified to species and counted each fish caught by all observed anglers. The ultimate fate of each observed fish was recorded as either kept, released, used as bait, or unknown. If a fish was released, the sampler attempted to determine if it survived or died. The combined catch by species for all observed anglers was recorded on one data sheet; individual catch per angler was not recorded.

When fishing had ceased for the day, the sampler measured total length in mm of as many observed kept fishes as possible by marking the length of each fish on a plastic measuring board, keeping all species separated. Not all observed kept fishes were measured due to refusal of an angler to have his/her catch examined, early filleting by the deck hand, or hazardous working conditions caused by inclement weather. If time permitted, fishes kept by unobserved anglers also were measured and their lengths were recorded separately from observed fishes' lengths.

## **Results**

### ***In Situ Density Estimates Derived From Random Transects***

From 1995 to 1998, 27, 22, 15, and 76 random transects, respectively, were conducted in the Big Creek study area for a total of 140 randomly selected transects. Areas were sampled unequally within and among years. Data for counts of fish per transect north of BCER passed tests of normality and equal variances; however, counts per transect data within and south of BCER did not pass these tests; therefore, analysis was confined to examination of annual mean counts per transect within each area separately for the four sampling years (1995-98). The non-normal distribution of count data within and south of BCER is most likely due to anomalous high counts in 1997 (Figure 2).

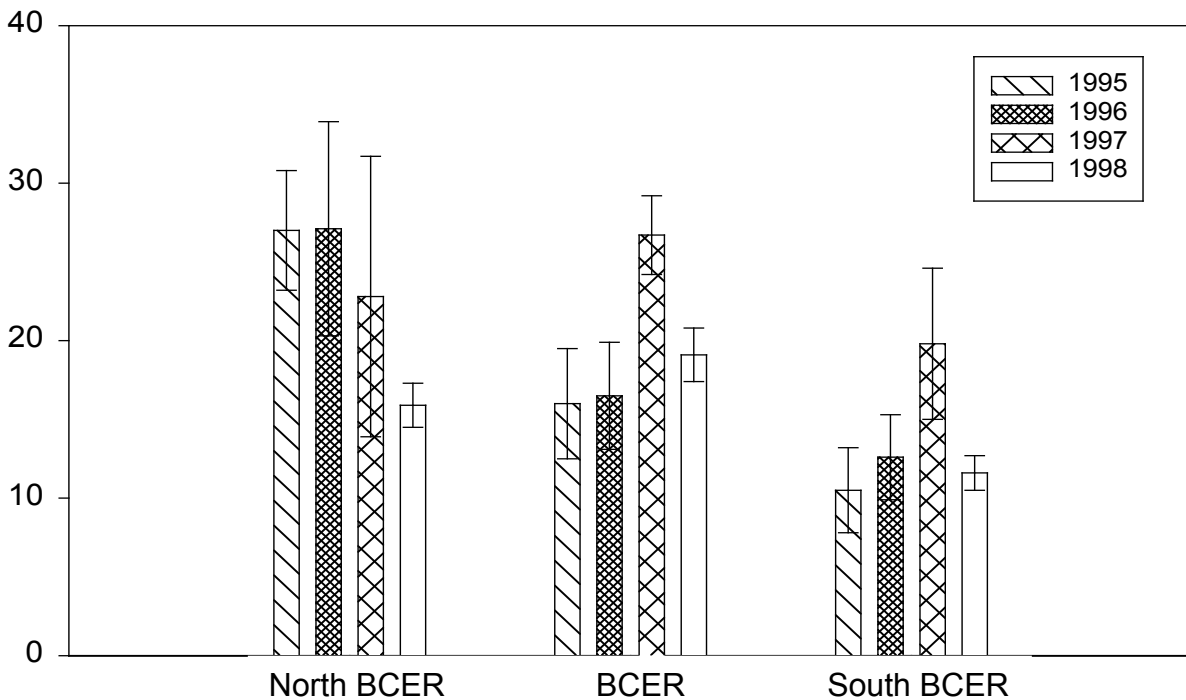


Figure 2. Counts of all species observed during random transects within, north, and south of the Big Creek Ecological Reserve (BCER) by area, Monterey County, 1995-98.

Annual means of numbers of fish observed per transect north of BCER decreased 41% (from 27.0 to 15.9 fish per transect) from 1995 to 1998 (Figure 2). Within BCER, except for the high 1997 value of 26.7 fish per transect, annual mean counts were within 3.0 fish per transect for the four-year period. From 1995 to 1998, mean numbers of fish per transect increased 19 percent from 16.0 to 19.1, respectively. South of BCER, annual mean counts of fish per transect were within 2.1 fish, except for the high 1997 value. The percent difference over the 4-year period was similar to BCER, with a 20 percent increase observed.

A two-sample t test was conducted separately for each area to determine significance of differences between annual mean counts of fish per transect in 1995 compared to 1998. Counts north of BCER in 1995 and 1998 were significantly different ( $p=0.002$ ); however, counts within and south of BCER were not statistically different ( $p=0.453$  and  $p=0.669$ , respectively) between 1995 and 1998.

### *Commercial Skiff Landings*

From August 1997 to December 1999, landings of 149 commercial skiff fishing trips were sampled on the beach at Mill Creek State Park and BCER, Monterey County (Table 1). When commercial fishers launched from and returned to Mill Creek, their fishing sites ranged from Fuller's (36° 12.70'N), which is 9.2 na. mi. north of the northern boundary of BCER to Gorda Point (35°53.75'N), which is 13.6 na. mi. south of the southern boundary of BCER (Figure 1). When they launched from and returned to BCER their fishing sites ranged from Fuller's to Limekiln (36°00.75'N), which is 2.5 na. mi. south of the southern boundary of BCER. Reported depth of fishing ranged from 5-600 feet (1.5-181.8 m); however, the majority of fishing occurred between 20-40 feet (6.0-12.1 m). Although rod-and-reel was occasionally used, most fishermen used "stick gear" and carried 10-15 units in their skiffs. Each stick gear unit was comprised of a five-foot long, weighted one inch PVC pipe to which five to eight circle hooks were attached. Each unit had a line and buoy attached. Fishermen deployed each unit separately and retrieved them within 1-3 hours.

The total catch of the 149 landings weighed 15,015 pounds. Fifty-three percent (8,017 pounds) of the total catch, representing 5,151 fish of 24 species, were sampled for species composition and length. Common and scientific names of fishes are listed in Table 2. From 1997 through 1998, we sampled 14%, 54%, and 65% of the landings, respectively. For the three-year sampling period the majority of the landings occurred from July through October. During these months, sea conditions are more conducive to launching and landing a small skiff along the Big Sur Coast.

From 1997 to 1998, blue rockfish declined from comprising 32% of the sampled landings to 2% and in 1999 they comprised less than 2% (Figure 3). Cabezon comprised 7% of the sampled landings in 1997, dramatically increased to 42% in 1998 (6-fold increase), and then declined slightly to 38% in 1999. Black-and-yellow rockfish showed a steady increase in percent composition from 1997 (15%), to 1998 (20%), to 1999 (32%). Gopher rockfish comprised 16% of the sampled landings in both 1997 and 1998 and declined to 9% in 1999. Grass rockfish increased in percent composition of sampled landing from 1997 (5%) to 1998 (13%) and then declined in 1999 (11%).

Table 1. Monthly Big Sur commercial nearshore skiff sampling effort and landings, 1997-99.

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| Year   | Month        | # Boats<br>Sampled | # Fish<br>Measured | Total<br>Landed (Lbs) | Total<br>Sampled (Lbs) |       |
|--------|--------------|--------------------|--------------------|-----------------------|------------------------|-------|
| 1997   | 8            | 6                  | 150                | 0                     | .                      |       |
|        | 9            | 8                  | 194                | 325                   | .                      |       |
|        | 10           | <u>15</u>          | <u>304</u>         | <u>1635</u>           | <u>285</u>             |       |
|        | Totals       | 29                 | 648                | 1960                  | 285                    |       |
| 1998   | 6            | 14                 | 474                | 1380                  | 165                    |       |
|        | 7            | 13                 | 639                | 1850                  | 900                    |       |
|        | 8            | 15                 | 748                | 1975                  | 1520                   |       |
|        | 9            | 10                 | 470                | 1565                  | 1045                   |       |
|        | 10           | <u>3</u>           | <u>188</u>         | <u>360</u>            | <u>250</u>             |       |
| Totals | 55           | 2519               | 7130               | 3880                  |                        |       |
| 1999   | 4            | 11                 | 395                | 905                   | 810                    |       |
|        | 5            | 6                  | 168                | 455                   | 420                    |       |
|        | 7            | 18                 | 565                | 1565                  | 1170                   |       |
|        | 8            | 8                  | 309                | 815                   | 575                    |       |
|        | 9            | 10                 | 172                | 1235                  | 395                    |       |
|        | 10           | 5                  | 69                 | 365                   | 175                    |       |
|        | 11           | 6                  | 132                | 375                   | 217                    |       |
|        | 12           | <u>1</u>           | <u>35</u>          | <u>210</u>            | <u>90</u>              |       |
|        | Totals       | 65                 | 1845               | 5925                  | 3852                   |       |
|        | Grand Totals |                    | 149                | 5,012                 | 15,015                 | 8,017 |

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Table 2. Common and scientific names of species observed in landings of Big Sur commercial nearshore skiff or commercial passenger fishing vessel fisheries, 1997-99.

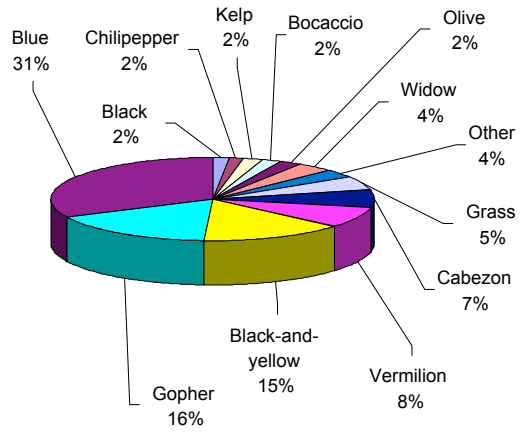
| <i>Common Name</i>        | <i>Scientific Name</i>            |
|---------------------------|-----------------------------------|
| Bank rockfish             | <i>Sebastes rufus</i>             |
| Black rockfish            | <i>Sebastes melanops</i>          |
| Black-and-yellow rockfish | <i>Sebastes chrysomelas</i>       |
| Blue rockfish             | <i>Sebastes mystinus</i>          |
| Bocaccio                  | <i>Sebastes paucispinis</i>       |
| Brown rockfish            | <i>Sebastes auriculatus</i>       |
| Canary rockfish           | <i>Sebastes pinniger</i>          |
| Chilipepper               | <i>Sebastes goodei</i>            |
| China rockfish            | <i>Sebastes nebulosus</i>         |
| Copper rockfish           | <i>Sebastes caurinus</i>          |
| Cowcod                    | <i>Sebastes levis</i>             |
| Flag rockfish             | <i>Sebastes rubrivinctus</i>      |
| Gopher rockfish           | <i>Sebastes carnatus</i>          |
| Grass rockfish            | <i>Sebastes rastrelliger</i>      |
| Greenspotted rockfish     | <i>Sebastes chlorostictus</i>     |
| Greenstriped rockfish     | <i>Sebastes elongatus</i>         |
| Kelp rockfish             | <i>Sebastes atrovirens</i>        |
| Olive rockfish            | <i>Sebastes serranoides</i>       |
| Quillback rockfish        | <i>Sebastes maliger</i>           |
| Rosy rockfish             | <i>Sebastes rosaceus</i>          |
| Speckled rockfish         | <i>Sebastes ovalis</i>            |
| Squarespot rockfish       | <i>Sebastes hopkinsi</i>          |
| Starry rockfish           | <i>Sebastes constellatus</i>      |
| Treefish                  | <i>Sebastes serriceps</i>         |
| Vermilion rockfish        | <i>Sebastes miniatus</i>          |
| Widow rockfish            | <i>Sebastes entomelas</i>         |
| Yelloweye rockfish        | <i>Sebastes ruberrimus</i>        |
| Yellowtail rockfish       | <i>Sebastes flavidus</i>          |
| Cabazon                   | <i>Scorpaenichthys marmoratus</i> |
| California halibut        | <i>Paralichthys californicus</i>  |
| California sheephead      | <i>Semicossyphus pulcher</i>      |
| Kelp greenling            | <i>Hexagrammos decagrammus</i>    |
| King salmon               | <i>Oncorhynchus tshawytscha</i>   |
| Lingcod                   | <i>Ophiodon elongatus</i>         |
| Ocean whitefish           | <i>Caulolatilus princeps</i>      |
| Pacific bonito            | <i>Sarda chiliensis</i>           |
| Pacific mackerel          | <i>Scomber japonicus</i>          |
| Pacific sanddab           | <i>Citharichthys sordidus</i>     |
| Rock greenling            | <i>Hexagrammos superciliosus</i>  |
| Rock sole                 | <i>Lepidopsetta bilineata</i>     |
| Sand sole                 | <i>Psettichthys melanostictus</i> |

### 1997

| Species          | # Observed | Percent |
|------------------|------------|---------|
| Black            | 10         | 2       |
| Black-and-yellow | 95         | 15      |
| Blue             | 209        | 31      |
| Bocaccio         | 13         | 2       |
| Cabezon          | 45         | 7       |
| Chilipepper      | 10         | 2       |
| Gopher           | 109        | 16      |
| Grass            | 33         | 5       |
| Kelp             | 12         | 2       |
| Olive            | 14         | 2       |
| * Other          | 23         | 4       |
| Vermilion        | 52         | 8       |
| Widow            | 23         | 4       |

Total Observed = 648

\* Other = species less than 2% of total catch.

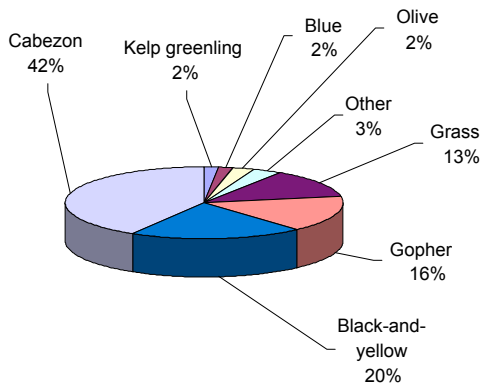


### 1998

| Species          | # Observed | Percent |
|------------------|------------|---------|
| Black-and-yellow | 541        | 20      |
| Blue             | 45         | 2       |
| Cabezon          | 1099       | 42      |
| Gopher           | 430        | 16      |
| Grass            | 343        | 13      |
| Kelp greenling   | 44         | 2       |
| Olive            | 65         | 2       |
| * Other          | 91         | 3       |

Total Observed = 2658

\* Other = species less than 2% of total catch.



### 1999

| Species          | # Observed | Percent |
|------------------|------------|---------|
| Black-and-yellow | 589        | 32      |
| Cabezon          | 688        | 38      |
| Gopher           | 173        | 9       |
| Grass            | 196        | 11      |
| Lingcod          | 46         | 2       |
| * Other          | 95         | 5       |
| Vermilion        | 58         | 3       |

Total Observed = 1845

\* Other = species less than 2% of total catch

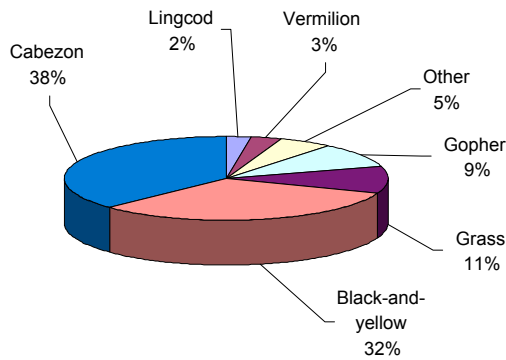


Figure 3. Species composition of landings of Big Sur commercial skiff fishery, 1997-99.



### *Commercial Passenger Fishing Vessel Catch*

From July 1997 to July 1998, twelve 1-day and three 2-day CPFV trips (total of 18 days) were observed. All trips departed from San Simeon, San Luis Obispo County and headed north to as far as Hurricane Point (Figure 1). During four days, fishing occurred in the Point Sur area, which extended from Ventura Rocks (2 na. mi. north of Point Sur, Latitude-36° 20.24'N) to Pfeiffer Point (5 na. mi. south of Point Sur, latitude-36° 12.31'N). During the remaining 12 days, fishing took place in the Cape San Martin area, which extended from Gamboa Point (3 na. mi. north of Lopez Point, latitude-36° 02.60'N) to Piedras Blancas (latitude-35° 41.14'N). Fishing occurred in water depths ranging from 6 fathoms (10.9 m) to 90 fathoms (163.6 m). From collected data we could not discern a significant difference between depths fished in the two areas. The average number of drifts that occurred for one day of fishing was 12 and the range was from 7 to 26 (Table 3). Total fishing time per day averaged approximately 4 hours and the average number of fish taken per day was 223. The sampler observed on average 10 anglers per day and each angler caught an average of 5.7 fish per angler-hour. We calculated daily CPUE for the 18 days of observed fishing. Fourteen days were spent in the Cape San Martin area and four days in the Point Sur area. Due to the insufficient sample size we were unable to analyze this data with a student t-test; therefore, we use a non-parametric equivalent Kruskal-Wallis, using alpha=0.05 and found no significant difference ( $p=0.089$ ) in CPUE between the two areas.

A total of 4020 fish were observed taken during the 18 days of sampling. Three thousand and forty-eight fish, representing 34 species, were observed taken during 12 days of fishing in the Cape San Martin region and 972 fish, representing 21 species, were observed taken during 4 days of fishing in the Point Sur region. Blue rockfish ranked number one in species composition and comprised 26% of the total catch of fishes taken during fishing in the Cape San Martin region; whereas, blue rockfish ranked third and comprised 11% of the catch in the Point Sur region (Figure 4). Yellowtail rockfish ranked second and comprised 12% of the catch at Cape San Martin, but were in the first rank and comprised 20% of the catch from Point Sur. Olive rockfish ranked third and comprised 11% of the Cape San Martin catch; however, at Point Sur they shared the sixth rank with widow rockfish and only comprised 6% of the catch. Bocaccio ranked sixth and comprised 7% of the catch in the Cape San Martin region; whereas, they were the second most abundant species taken at Point Sur and comprised 16% of the catch.

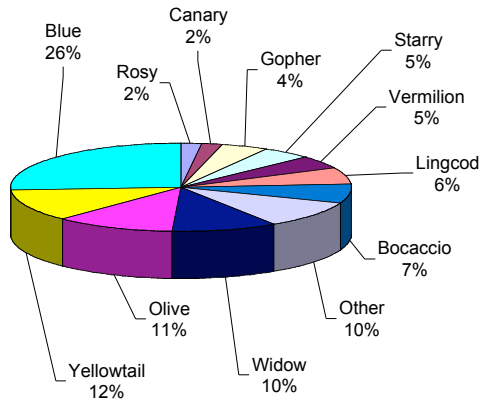
Fishery length data are usually assumed not to be normally distributed. We tested the total length of fish for each species taken in the Point Sur and Cape San Martin areas for assumptions of equal variances and normal distribution and, as expected, these test failed for the majority of species. We then performed a non-parametric equivalent (Kruskal-Wallis) for each species using alpha=0.05. There was a highly significant difference between Point Sur and Cape San Martin in the total lengths for 8 of the 11 species tested (Figure 5 and Table 4). Mean total lengths of blue, canary, copper, olive, vermilion, widow, and yellowtail rockfishes and lingcod were larger in the Point Sur region compared to the Cape San Martin region.

Table 3. Monthly fishing summary of catches from commercial passenger fishing vessels fishing in the Cape San Martin and Point Sur regions of the Central California coast, 1997-98.

| Year                  | Date            | Fishing Region  | Trip | Total  | # Fish   | Total        | Avg Number | Catch per Angler Hour |          |
|-----------------------|-----------------|-----------------|------|--------|----------|--------------|------------|-----------------------|----------|
|                       |                 |                 |      | Drifts | Observed | Minutes      | Anglers    | Mean                  | Variance |
| 1997                  | July 25         | Cape San Martin | 1    | 7      | 102      | 171          | 5          | 7.54                  | 3.56     |
|                       | August 1        | Cape San Martin | 2    | 11     | 153      | 186          | 6          | 8.52                  | 16.30    |
|                       | August 8        | Cape San Martin | 3    | 18     | 160      | 201          | 8          | 6.57                  | 12.55    |
|                       | August 15       | Cape San Martin | 4    | 10     | 229      | 204          | 12         | 5.96                  | 23.76    |
|                       | August 22       | Cape San Martin | 5    | 9      | 207      | 175          | 12         | 5.53                  | 6.66     |
|                       | August 29       | Cape San Martin | 6    | 9      | 129      | 266          | 6          | 4.64                  | 3.50     |
|                       | September 12    | Cape San Martin | 7    | 12     | 618      | 281          | 12         | 10.84                 | 60.01    |
|                       | September 19    | Cape San Martin | 8    | 8      | 175      | 203          | 7          | 7.92                  | 7.16     |
|                       | October 24      | Cape San Martin | 9    | 11     | 267      | 250          | 12         | 6.19                  | 8.92     |
|                       | December 27     | Point Sur       | 10 A | 19     | 266      | 315          | 12         | 4.09                  | 11.60    |
| December 28           | Cape San Martin | 10 B            | 14   | 250    | 199      | 12           | 6.13       | 33.57                 |          |
| 1998                  | January 24      | Point Sur       | 11 A | 26     | 323      | 381          | 11         | 4.48                  | 19.26    |
|                       | January 25      | Cape San Martin | 11 B | 8      | 148      | 153          | 12         | 3.42                  | 7.73     |
|                       | May 9           | Point Sur       | 12 A | 19     | 241      | 324          | 8          | 5.05                  | 7.78     |
|                       | May 10          | Point Sur       | 12 B | 10     | 142      | 217          | 8          | 4.95                  | 27.04    |
|                       | May 31          | Cape San Martin | 13   | 14     | 172      | 289          | 12         | 2.71                  | 3.87     |
|                       | June 28         | Cape San Martin | 14   | 8      | 147      | 235          | 12         | 3.07                  | 2.25     |
|                       | July 26         | Cape San Martin | 15   | 11     | 291      | 305          | 10         | 4.86                  | 12.10    |
| Sum Total             |                 |                 | 18   | 224    | 4020     | 72 hr 35 min | 176        | -                     | -        |
| Total Average per Day |                 |                 | -    | 12.4   | 223.3    | 4 hr 2min    | 10         | 5.69                  | -        |

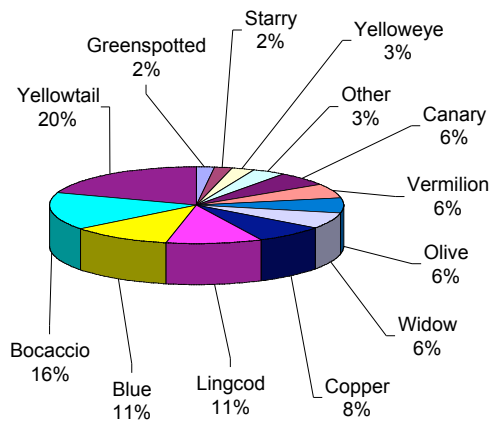
### Cape San Martin

| Species            | # Observed | Percent |
|--------------------|------------|---------|
| Blue               | 759        | 26      |
| Bocaccio           | 201        | 7       |
| Canary             | 58         | 2       |
| Gopher             | 127        | 4       |
| Lingcod            | 181        | 6       |
| Olive              | 332        | 11      |
| * Other            | 288        | 10      |
| Rosy               | 57         | 2       |
| Starry             | 136        | 5       |
| Vermilion          | 141        | 5       |
| Widow              | 296        | 10      |
| Yellowtail         | 338        | 12      |
| Grand Total = 2914 |            |         |



\* Other = Species less than 2% of the total catch.

### Point Sur



| Species            | # Observed | Percent |
|--------------------|------------|---------|
| Blue               | 113        | 11      |
| Bocaccio           | 164        | 16      |
| Canary             | 58         | 6       |
| Copper             | 79         | 8       |
| Greenspotted       | 19         | 2       |
| Lingcod            | 108        | 11      |
| Olive              | 66         | 6       |
| * Other            | 37         | 3       |
| Starry             | 21         | 2       |
| Vermilion          | 63         | 6       |
| Widow              | 66         | 6       |
| Yelloweye          | 26         | 3       |
| Yellowtail         | 200        | 20      |
| Grand Total = 1020 |            |         |

\* Other = Species less than 2% of the total catch.

Figure 4. Species composition of fishes sampled aboard commercial passenger fishing vessels fishing in the Cape San Martin and Point Sur regions of the central California coast, 1997-98.

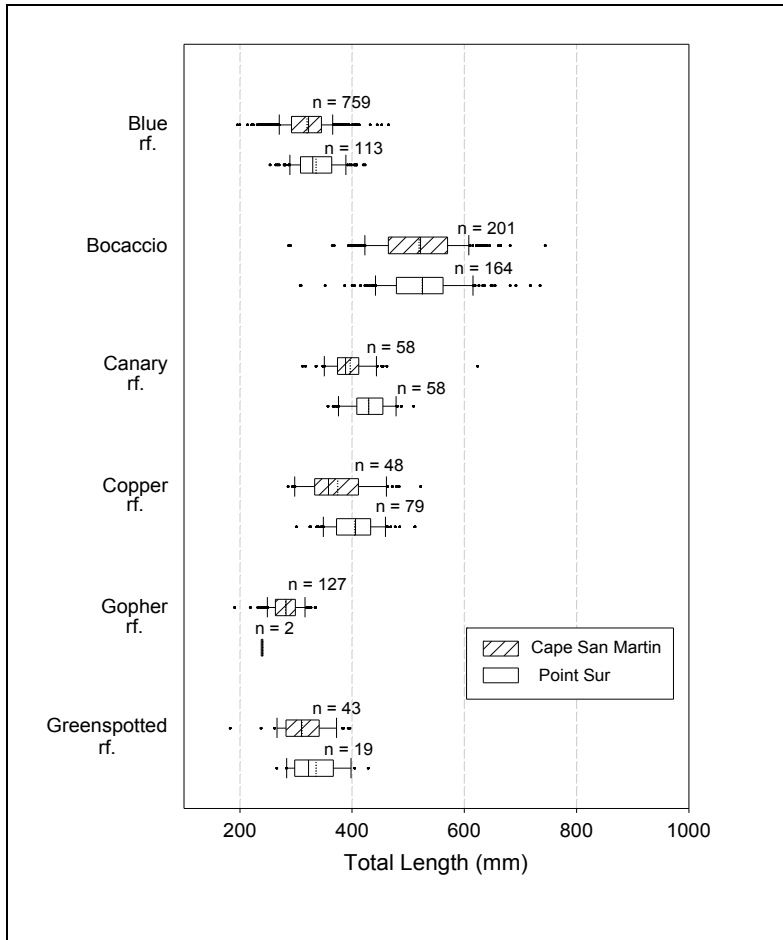


Figure 5. Total length of 12 species sampled aboard commercial passenger fishing vessels fishing in the Cape San Martin and Point Sur regions of the Central California coast, 1997-98. Dotted line represents the mean, solid line represents the median, boxed area is the interquartile range (50% of values), and the upper and lower “whiskers” represent 90<sup>th</sup> and 10<sup>th</sup> percentile points, respectively.

Table 4. Results of Kruskal-Wallis non-parametric test of species' total lengths between sampled catch aboard commercial passenger fishing vessels fishing in the Cape San Martin and Point Sur regions of the Central California coast, 1997-98.

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| Species               | Kruskal-Wallis<br>p Value | Significant Difference<br>Between Areas |
|-----------------------|---------------------------|---|
| Blue rockfish         | <0.001                    | Yes                                     |
| Bocaccio              | 0.561                     | No                                      |
| Canary rockfish       | <0.001                    | Yes                                     |
| Copper rockfish       | 0.001                     | Yes                                     |
| Greenspotted rockfish | 0.082                     | No                                      |
| Olive rockfish        | 0.033                     | Yes                                     |
| Vermilion rockfish    | <0.001                    | Yes                                     |
| Widow rockfish        | <0.001                    | Yes                                     |
| Yelloweye rockfish    | 0.975                     | No                                      |
| Yellowtail rockfish   | <0.001                    | Yes                                     |
| Lingcod               | <0.001                    | Yes                                     |

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## Discussion

The decline in the success and quality of Central California's nearshore rockfish fisheries over the past decades and the rapid expansion of the live/premium fish fishery, which targets shallow water kelp forest fishes, have underscored the need for revised management strategy for this resource. Marine protected areas have been suggested as an alternative or additional management tool to create sustainable fisheries. There is little doubt that eliminating fishing in a designated area will help maintain the area's natural biological diversity and provide a "heritage area" for aesthetic and scientific purposes. However, before managers can advocate marine reserves as an effective management tool, they must first be able to demonstrate that the biomass and reproductive potential of target species within reserves is significantly greater than in fished areas.

The overarching goal of the study is to provide a benchmark or "line in the sand" of data collected in a newly established marine protected area. Our hope is that we have done an adequate job so that future studies can statistically document temporal changes in nearshore fish populations within and adjacent to BCER. Because both fishery independent and dependent data are crucial in the evaluation of the reserve as a fishery management tool, we collected data from *in situ* surveys of fish populations within BCER and monitored the landings of commercial and recreational fisheries targeting fishes in the vicinity of BCER.

We also examined numbers of all fish observed during random transects conducted within and adjacent to BCER. Randomly collected data is more representative of the surveyed areas than data collected at permanent stations; however, it is more variable due to differences in habitat. Significant differences among years and areas were noted for randomly collected data. We suspect high fish density estimates in 1993 and 1997 were related to the El Niño conditions during these years. The last quarter of 1993 was the later portion of the 1992-93 major El Niño events along the Central California coast (Hayward et al. 1994 and Lynn et al. 1995) and 1997 was a minor warm water event (Norton et al. 1999). El Niño conditions have been documented to reduce primary productivity, affect species distribution and abundance, promote recruitment failure for some species of rockfish, and cause poor growth and condition of adult rockfish (Lenarz et al. 1995 and VenTresca et al. 1995). During these warm water periods we observed "more fish out in the open looking for food" compared to non El Niño years. This variability among years underscores the necessity incorporating environmental parameters with *in situ* estimates of fish densities for spatial and temporal comparisons. Excluding 1997 from the 1995-98 within area annual comparisons shows little change within and south of BCER and a significant decline north of BCER from 1995-1998. We suspect that the decline of numbers of fish per transect observed north of BCER from 1995 to 1998 is due to increased fishing pressure from the commercial live/premium fish fishery in this area. The area south of BCER has been fished for many years by the commercial small skiff fishery that used launch sites located south of BCER.

Landings of the Big Sur commercial skiff fishery reveal a concentration on species commonly observed in kelp forest. A high percentage of 1998-99 landings were composed of cabezon and black-and-yellow and gopher rockfishes. It will be extremely interesting to see if these species maintain a high percentage in future landings and if the densities of these species

changes in future *in situ* surveys. The majority of the CPFV fishery occurred in areas not adjacent to BCER. This study documented that 4.5 times more trips, which departed from San Simeon, San Luis Obispo County, were made to the Cape San Martin region as were made to the Point Sur region. Personnel communications with the CPFV industry confirmed that the Cape San Martin region receives more fishing pressure than does the Point Sur region. Most species taken in the Point Sur region were larger than those taken in the Cape San Martin region; however, there was not a significant difference in the CPUE between the two areas.

The experimental design of this survey, like many previous studies of California's central coast nearshore reef fishes, stratified random sampling of fish densities using the canopy of the giant kelp, *Macrocystis pyrifera*, as a proxy for reefs/rocky bottom strata. The increased availability of precise high-resolution maps of the seafloor and habitat, generated from acoustic remote-sensing methods and enhanced navigation tools (dGPS) makes it possible to accurately locate and randomly sample specific habitats. In the future, *in situ* surveys samples to estimate fish densities can be allocated in specific strata to minimize variability. We recognize that an *a priori* stratification of sampling effort may be a more efficient and statistically powerful approach (Cuff and Coleman 1979, McCormick and Choat 1987, Jolly and Hampton 1990, Smith and Gavaris 1993, and VenTresca et al. 2002).

## CHAPTER 3

### Deepwater Habitat And Fish Resources Associated With The Big Creek Ecological Reserve

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#### Abstract

Big Creek Ecological Research Reserve (BCER), located off Central California, has been closed to fishing since January 1994. We used side scan sonar and an occupied submersible to collect baseline information on species-habitat relationships, density, and species and size composition of fishes inside and outside BCER. Forty-three dives were made in Fall of 1997 and 1998, in depths from 20-250 m. From 142 video transects, we identified over 70,000 fishes of 82 taxa, including 36 species of rockfishes. About 93% of the 25,159 fishes inside BCER were rockfishes comprising at least 20 species. Young-of-the-year rockfishes dominated rock outcrops in 20-90 m depth inside and outside BCER. Four distinct fish assemblages were associated with: 1) fine smooth sediment in deep water; 2) bedrock with uneven surface in deep water; 3) sand waves and shell hash in shallow water; and 4) boulders and organic habitats on rock in shallow water. There were no significant differences in fish density among locations (inside and outside BCER), depths, or between years. Density was significantly higher in high relief rock habitat than in low relief soft and mixed sediments, regardless of location. There were no consistent patterns of larger fishes inside compared to outside the protected area. We recommend development of a monitoring program to continue these surveys after increased time of protection and with increased assessment effort in the appropriate habitats of economically valuable species. In addition, extending the boundaries of BCER seaward would protect habitats and fishes in water depths >100 m.

Detailed information about this project can be found in the following publications:

Yoklavich, M., G. Cailliet, R.N. Lea, H.G. Greene, R. Starr, J. De Marignac, and J. Field. 2002. Deepwater habitat and fish resources associated with the Big Creek Ecological Reserve. California Cooperative Fisheries Investigation Report. CalCOFI MS2002-03.

Yoklavich, M., G. Cailliet, R.N. Lea, H.G. Greene, R. Starr, J. De Marignac, and J. Field. 2001. Deepwater habitat and fish resources associated with a Marine Reserve: Implications for Fisheries Management. Part 1. California Sea Grant College Program. Marine Ecological Reserves Research Program Research Results R/BC 1.



## **CHAPTER 4**

### **Anacapa Island Ecological Reserve: A Research Review**

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#### **Abstract**

The Natural Area of Anacapa Island Ecological Reserve was established by the Fish and Game Commission in 1978 and consists of 12 hectares along the north shore of East Anacapa Island. This area is closed to all fishing. Several biological monitoring programs operating in the Northern Channel islands (Channel Islands National Park, Partnership for Interdisciplinary Studies of Coastal Oceans (PISCO) and Channel Islands Research Program (CIRP)) include the Anacapa reserve. In this chapter I review existing studies for the Anacapa reserve which may shed light on the following questions: 1) Are target species larger or more abundant in the Anacapa reserve?, 2) Does the reserve serve as a control to evaluate changes in nearby fished areas?, 3) Does the Anacapa reserve promote ecosystem function?, and 4) Is the reserve large enough to protect target species?

## *Anacapa Island Ecological Reserve*

### *Background to the Natural Area of the Anacapa Island Ecological Reserve.*

Anacapa Island is in the Northern Channel islands and consists of three islands separated by small passages. The Natural Area of Anacapa Island Ecological Reserve was established by the Fish and Game Commission and consists of 12 hectares along the north shore of East Anacapa Island. This area was closed to all fishing in 1978 and is one of the smallest no-take areas in California (Mc Ardlé 1997). In addition, a small area on the north side of West Anacapa island is designated as a brown pelican fledging area. No entry is allowed in this area from January 1 through October 31. The brown pelican fledging area covers approximately 4000 ft of shoreline and extends out to a water depth of approximately 120 ft.

### *Existing monitoring programs that include Natural Area of Anacapa Island Ecological Reserve*

The Channel Islands National Park has maintained a monitoring program for kelp forests throughout the Northern Channel islands (Santa Barbara, Anacapa, Santa Cruz, Santa Rosa and San Miguel Islands) since 1982 (Davis et al. 1997). The kelp forest monitoring program (KFM) samples a variety of organisms annually at 16 sites spread throughout these islands. Two of these KFM sites are located in the reserve at Anacapa. The protocol includes several techniques for monitoring a variety of organisms including macro algae (kelps, understory algae), macro invertebrates (urchins, lobsters, starfish, etc.), recruitment of selected invertebrates (urchins, abalone, etc.) and fishes (Davis et al. 1997).

PISCO (Partnership for Interdisciplinary Studies of Coastal Oceans) established rocky reef monitoring sites at Anacapa and Santa Cruz islands in 1999. PISCO also focuses on kelp forest habitats and uses complementary but not identical protocols to the KFM. Several sites are monitored by both KFM and PISCO including 2 sites in the Anacapa reserve. Important differences in the protocols include the following: PISCO monitors on randomly placed transects within each site, KFM uses a fixed transect, PISCO species lists are more detailed than KFM and PISCO monitors physical and chemical oceanography with fixed moorings at the sites as well as monthly settlement dynamics for invertebrates and fishes.

The Channel Islands Research program (CIRP) has been conducting marine biological studies around the eight islands in the Southern California Bight each year since 1980, with preliminary cruises dating back to 1978. Cruises include: water temperature studies, especially during El Niño - La Niña, mantis shrimp studies, echinoderm and fish surveys at Santa Catalina and San Clemente Islands; kelp/urchin community studies at Anacapa Island; eelgrass ecosystem surveys, echinoderm aggregations and disease around the northern islands; intertidal monitoring projects in cooperation with various agencies seeking information on marine issues of concern in California. CIRP does not specifically target the marine reserve at Anacapa for long-term monitoring.

## *Review of research at the Anacapa Ecological reserve*

In general, very few studies have been conducted or published that documented research on Anacapa Island. There are even fewer studies that focused on the Anacapa reserve to test specific goals or functions of marine reserves. This is likely due to its extremely small size and/or the lack of other reserve areas nearby for comparison. However, there are a few studies (both published and unpublished) that have given some insight into several aspects of the functioning of the Anacapa reserve. For the rest of this document, the term Anacapa reserve refers to the Natural Area of the Anacapa Island Ecological Reserve unless other specified.

### *Are target species larger or more abundant in the Anacapa reserve?*

Increases in the size and the abundance of organisms inside reserves relative to outside of reserves have been well documented in the literature (Halpern in press). More and larger individuals will have an important effect on potential production in a reserve. The effects of high density are relatively straightforward. Barring any density-dependent effects, more fish produce more eggs. However, this effect is greatly magnified when the fish are larger. For fish, the relationship between body length and fecundity (number of eggs) is not linear. A small increase in body size results in a disproportionately large increase in egg production. If a reserve is large enough to contain the movements of adults within the boundaries yet allow larval dispersal out, then the reserve is likely to export young to outside areas and increase fisheries yields outside the reserve. The presence of large individuals also has simple conservation benefits. People enjoy observing large fish and other marine organisms. The presence of large individuals, especially in areas where they no longer exist (e.g., Southern California) could also benefit tourism and provide additional recreational opportunities.

Tests of this effect are less common in Southern California reserves than other parts of the world. In one study, the abundance and size structure of fish populations were compared inside and outside of 5 no-take marine reserves in Southern California (Tretault, in prep). Two of these five sites were the Anacapa reserve and the Pelican closure at Anacapa where fishing is prohibited during the pelican breeding season (January through October). For two harvested fish species, kelp bass (*Paralabrax clathratus*) and California Sheephead (*Semicossyphus pulcher*), both the density of all sizes combined and of only harvestable sized fish were higher inside of both protected areas relative to outside. The average density of harvestable sized kelp bass was 17 times higher inside the protected areas than outside. Harvestable California sheephead were on average 11 times more abundant inside than outside. She also measured the density of several common but non-harvested species (Garibaldi (*Hypsypops rubicundus*), Seniorita (*Oxyjulis californica*) and Rock wrasse (*Halichoeres semicinctus*)). In contrast to the results for the harvested species, non-harvested species did not show significant increases in density or size according to protection status. In fact, seniorita were significantly more abundant outside of the pelican closure than inside. Importantly, habitat characteristics of the reserve and control areas were not significantly different.

Data on red sea urchin abundance collected by the National Park Service showed that there has been a decline in the abundance of large individuals, and suggests that fishing may be the cause. Since 1985, the abundance of harvestable sized red urchins (>82.5mm test diameter) has

declined more than 5% per year at fished sites on Santa Rosa and San Miguel islands (the sites contributing most to the overall catch) relative to non-fished reserve sites on Anacapa (S. Schroeter and D. Reed, unpublished analysis of KFM data). Similar declines were not observed in the abundance of red sea urchin recruits (<25mm test size).

*Does the reserve serve as a control to evaluate changes in nearby fished areas?*

Although the use of marine reserves for fisheries enhancement is widely discussed (and debated), their use for testing stock assessment methods or traditional management strategies has received less attention (NRC 2001). No-take areas may be critical for providing baseline data with which to test and improve fisheries management. By strictly controlling the effects of human impacts, marine protected areas can be used to focus more intense local management designed to increase yield and allow research to help define sustainability and protect against uncertainty (Dayton et al. 2000). One study has used the reserve at Anacapa to directly compare between two methods of stock assessment for an emerging fishery.

Schroeter et al. (2001) used fishery independent data (long-term abundance from the NPS KFM program) to evaluate the status of the newly emerging dive fishery on warty sea cucumber (*Parastichopus parvimensis*) in Southern California. They employed a BACI design, using abundance data from inside a marine reserve (Anacapa) and outside (the other KFM sites) both before and after the onset of fishing. They then compared the results of this stock assessment using fishery-independent data with a stock assessment based on CPUE (catch per unit effort-fishery dependent data).

Monitoring data showed that the abundance of warty sea cucumber decreased throughout the islands at the fished sites following the onset of the fishery. At two unfished sites in the reserve at Anacapa, the abundance of sea cucumbers showed no significant change but tended to increase. All seven fished sites used in the BACI analysis (those sites that met the assumptions of BACI) showed significant declines relative to the unfished sites following the onset of fishing. These declines ranged from 33% to 83%. Stock assessment based on CPUE data differed dramatically from that based on the monitoring data. CPUE did not decline at any of the 6 islands during the 3-6 year period after the onset of fishing despite the general decline in abundance at the sites.

The combination of a no-take marine reserve with substantial and long-term monitoring data inside and out made it possible for these authors to a) test the coherence of stock assessments based on fishery dependent data versus fishery independent data and b) disentangle the potentially confounding effects of natural changes in populations from those due to fishing mortality.

The authors point out that no-take areas can provide beneficial information even for well-established fisheries. By comparing the trends in populations before and after establishment of a marine reserve we can begin to understand the resiliency of stocks to recover in the absence of fishing mortality and the extent to which a fishery has caused impacts to stock. This knowledge is impossible to get by reducing fishing pressure throughout the spatial range of a fishery, but will require spatial set asides large enough for populations to recover.

*Does the Anacapa reserve promote ecosystem function?*

One goal of a marine reserve is to promote healthy ecosystems by protecting biodiversity and allowing all members of the community to be present and function ecologically. Marine systems have been changed drastically by fishing and other anthropogenic influences. In many systems the higher-order predators and other important species have been virtually eliminated. In many others, the benthic habitat itself has been disrupted. Understanding the changes in marine ecosystems is especially difficult due to logistic constraints, the lack of long-term data and the potentially lasting consequences of rare or episodic events (Dayton et al. 2000). Yet, without natural systems or benchmarks to compare the observed changes, the difficulties are magnified. Without pristine systems important questions cannot be studied--for example, how the ecosystem roles of various species can be assessed, how they can be managed in a sustainable manner, and how we can evaluate resilience or relative rates of recovery. (Dayton et al. 2000).

Here I review two studies that have investigated the community or ecosystem dynamics in and around the Anacapa reserve. The Anacapa reserve contains, among other habitats, temperate kelp forests. Although kelp bed dynamics are complex, the interactions between macroalgae, grazers such as urchins, and predators on those grazers are widely recognized as controllers in these systems (Tegner and Dayton 2000). Both studies reviewed here investigate those dynamics.

A community analysis using NPS KFM data from the marine reserve at Anacapa concluded that the ecosystem within the reserve has a more “natural balance” than ecosystems exposed to fishing pressure (Sladek Nowlis in press). Two of the major urchin predators in Southern California kelp beds are the spiny lobster and the California sheephead (Tegner and Levin 1983). Both of these predators were more abundant in the marine reserve at Anacapa than other KFM sites outside the reserve. Spiny lobster densities inside were 10 times what they were outside, while California sheephead were 1 1/2 times more abundant inside than outside. There are three main urchin species in the Channel islands; red sea urchins that are harvested by humans and white and purple urchins, which are not harvested commercially. All three are algal grazers. This analysis showed that the non-targeted purple and white urchins have grown dramatically in abundance in fished areas, probably as a result of lack of competitors or predators. The density of the non-harvested, white sea urchins has increased approximately 4 times in the reserve compared to 15 times at sites outside the reserve since 1983. As a consequence, the large canopy forming giant kelp, *Macrocystis*, which provides the majority of the structure in a healthy kelp forest, has all but disappeared in sites outside of the reserve, while increasing in the reserve sites. Sladek Nowlis (in press) concludes that by protecting even small areas from the effects of fishing, ecosystems are able to achieve a level of health and natural balance not possible in areas that continue to be fished.

In another analysis of the KFM data, Lafferty and Kushner (2000) investigated the roles of predation and disease in the population regulation of purple sea urchins throughout the Channel Islands, including the Anacapa reserve. They found that urchin abundance was not explained by recruitment patterns. There was an inverse relationship between urchin density and invertebrate

urchin predator abundance (Spiny lobsters and sunflower stars) but not for fish predators. They also investigated the prevalence of disease in the urchin populations and found that outbreaks of bacterial disease tended to occur in high-density populations but disease was not related to water temperature. In the western sites where lobsters are rare and the major invertebrate predator is the sunflower star, urchin densities tend to be low and disease was rare. In the eastern sites where the major predator is the spiny lobster, which are fished heavily, urchin abundance is high and urchin disease is prevalent. The exception to the pattern in the eastern sites was the marine reserve at Anacapa. Presumably due to protection, spiny lobsters are more abundant in the reserve, urchins were less abundant, and disease was rare.

*Is the reserve large enough to protect target species?*

Among many potential fisheries benefits of protected areas, two relate to flux of individuals between fished and unfished areas. One possible but largely unproved benefit is that reserves protect a critical minimum spawning stock and maintain or enhance fishery yields at a large scale by larval export out of reserves (“recruitment” effect). Although there is mounting evidence that both population sizes and individual sizes can be larger inside of reserves (Halpern in press), evidence for successful recruitment of larvae produced by these protected fish is lacking. A second potential benefit is the maintenance or enhancement of fishery yields in areas adjacent to a reserve by movement of adults or juveniles across the boundaries after a period of initial growth in the reserve (the “spillover” effect). Transfer rates across reserve boundaries depend not only on the permeability of those boundaries but also on the size of the reserve relative to the movements and home range sizes of the target species.

Surprisingly little is known about normal fish movements and home range sizes, and evidence for a spillover effect is minimal. Since acceptance of reserves (and hence effectiveness in terms of enforcement) often depends on strong community support, locally increased catches (spillover) may be more convincing to anglers than regional recruitment benefits. In either case, the ability of a reserve to provide protection depends on consistent use of the protected area by individual fish. Thus, knowledge of the movement and activity patterns of target species is critical.

The author (Caselle) is currently conducting a study of fish movements at the Anacapa reserve. Using commercial live-fish trap gear, we capture and tag (with external colored tags) fish throughout the reserve. We divided the reserve into 3 movement blocks (blocks 1-3). Each has a linear distance along the shoreline of roughly 750 m. Within each of the 3 blocks in the reserve, we fish as similarly as possible to how commercial fishermen would fish normally. That is, each trapset location is not predetermined, nor for the most part is the effort we put into each block. We use a different color fish tag for each block. We assess movement by recapturing the fish (which gives exact distance between captures) and by visually resighting using SCUBA surveys (which gives estimates of movement from block to block). The majority of the trapped, retrapped, and resighted individuals are California sheephead.

The average distance between trap recaptures for California sheephead was 115 m. The minimum distance between recaptures was 18m and the maximum was 860 m. There was no relationship between distance traveled and either body size or time at large. We resighted 417

California sheephead to date. Most fish (80-100%) were resighted in the block in which they were tagged. The small percentage that moved from one block to another, usually moved to an adjacent block. This shows that movements on the scales of 500-750 m alongshore are not common.

These movement estimates are in line with the home range size of California sheephead estimated from ultrasonic telemetry on Catalina island (Christopher Lowe, unpublished data). In that study, home range areas (not linear distances) ranged from 1000's of square meters to approximately 70,000 m<sup>2</sup>.

## CHAPTER 5

### **Marine Reserves in the Monterey Bay National Marine**

*Sanctuary: What We Do and Don't Know*

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### **Abstract**

Rapidly growing global consideration for the development of coastal networks of marine reserves has focused attention on their potential application in the coastal waters of California. This focus stems in part from recent State and federal legislation necessitating development and implementation of some form of marine reserve network throughout California's coastal waters, and from the potential for past and present research in three existing marine reserves (Big Creek Ecological Reserve, Point Lobos Ecological Reserve, and Hopkins Marine Life Refuge) to inform managers on the potential value of marine reserves as tools for conservation and fisheries management. Simultaneous with this increased interest has been a rapid increase in theoretical and analytical studies on the design and evaluation of marine reserves, as well as empirical accounts of reserve effects throughout the world. Few studies of California's marine reserves exist. All of these studies have focused solely on the response of selected fish species to protection within reserves. From the few studies conducted in the three reserves encompassed within the Monterey Bay National Marine Sanctuary (MBNMS), few general conclusions can be drawn. To date, there is some evidence that fishes inside two of three reserves within the MBNMS appear to have greater potential larval production than nearby populations outside of reserves inhabiting comparable areas of reef habitat. Studies in the third reserve, Big Creek Ecological Reserve, were carried out only within the first four years subsequent to establishment. Given the episodic nature of fish recruitment, it is not surprising that marked differences were not observed. The magnitude of the reserve effect on increased potential of larval production appears to be positively related to the age of the reserve. To date, little information exists on community- or ecosystem-wide responses to protection by reserves and only one study has examined the state of fish populations for more than two years. Moreover, the lack of sampling over several years necessary to test for trajectories in populations inside and outside reserves makes attributing causal inferences to reserve effects equivocal. To develop a better understanding for the potential of reserves as tools for conservation and fisheries management, it is clear that ecosystem-wide responses have not been examined and remain unknown, the temporal persistence of differences between reserve and non-reserve populations is unknown, and unequivocal ascription of differences between reserve and non-reserve populations to causal reserve effects is tenuous. Better understanding of the consequences of reserve establishment requires both creation of new reserves in concert with well-designed monitoring studies over many years.



## *I. Why the current interest in marine reserves within the MBNMS?*

Marine reserves are designated areas within which human activities that can result in the removal or alteration of biotic and abiotic components of an ecosystem are prohibited or greatly restricted (NRC 2001). This definition corresponds with what is typically referred to in the literature as “no-take” marine protected areas (MPAs) or marine reserves (MRs). Activities specifically curtailed within a marine reserve are extraction of organisms (e.g., commercial and recreational fishing, kelp harvesting, commercial collecting), mariculture, and those activities that can alter oceanographic or geologic attributes of the habitat (e.g., mining, shore-based industrial-related intake and discharges of seawater and effluent, respectively). Local examples of marine reserves and such restrictions include Hopkins Marine Life Refuge, Point Lobos State Ecological Reserve and Big Creek Ecological Reserve (McArdle 1997, Brown 2000).

Current and urgent interest in the proposed development of a network of marine reserves within the MBNMS stems from three contemporaneous events. At the global scale, national governments and conservation oriented non-governmental organizations (NGO's) are promoting networks of marine reserves as an approach for conservation akin to more familiar terrestrial reserves. This impetus is motivated both by recognition of the global paucity (< 1%) of reserves in the coastal marine environment (Kelleher 1999) relative to terrestrial environments, and growing evidence for overfishing and related detrimental impacts to marine ecosystems (NRC 1995, Vitousek et al. 1997, Botsford et al. 1997, Jackson et al. 2001). Focus on marine reserves as a tool for ecosystem and fisheries conservation has been stimulated by a growing literature on their potential positive consequences in the scientific literature (Plan Development Team 1990, Agardy 1997, Allison et al. 1998, Bohnsack 1998, Lauck et al. 1998, Murray et al. 1999, NRC 2001, Roberts et al. 2001). These arguments have been made on conceptual inferences, theory based on fishery models, and patterns or results derived from observational studies of marine reserves (i.e., empirically based inferences).

At the national scale, Executive Order 13158 signed by retiring U.S. President Clinton directed the federal government to work with public and private partners to significantly strengthen and expand the national system of MPAs. This goal includes analysis of the existing MPAs in U.S. waters and consideration for the development of a nation-wide network of MPAs. This Order also established the national Marine Protected Area Center (<http://mpa.gov/welcome.html>) to implement these objectives. Because of parallel activities in the state of California and growing interest in other west coast states, the Center is following closely processes currently underway here. Simultaneously, National Marine Sanctuaries along the west coast are examining the potential role of marine reserves and other MPAs as they relate to their mandated objective of protecting habitat within Sanctuaries.

At the regional level, the Pacific Fisheries Management Council has also been examining the potential applicability of marine reserves (see “marine reserves” link at <http://www.pcouncil.org/>). Following the recommendations of an ad-hoc committee (see Parrish et al. 2000), the Council adopted six fishery management objectives that might be addressed by marine reserves as a supplemental tool for management of groundfish fisheries. The objectives that marine reserves might help address, ranked in order of descending priority, include; (1) Stock Rebuilding: assist in rebuilding overfished stocks and maintaining them at productive

levels (2) Biological Productivity: enhance long-term biological productivity, (3) Economic Productivity: assist in achieving long-term economic production, while minimizing short-term negative economic impact on all users, (4) Insurance: provide protection for the resource, as a hedge against the realities of management uncertainty and the effects of natural environmental variability, (5) Habitat Protection: conserve and protect EFH, and (6) Research and Education: provide unfished areas for research that will serve as controls for assessment of the effects of long-term environmental variations and the potential habitat alterations due to fishing, and also increase our understanding of the role marine reserves may play in fishery management.

At the state level, two recent legislative acts prompted consideration of marine reserves for both conservation and fisheries management. The Marine Life Management Act was enacted in 1999. The Act (MLMA) stipulates several new fisheries management and conservation objectives for California's marine living resources (Weber and Heneman 2000). Rather than focusing on single fisheries management, the MLMA calls for an ecosystem-wide approach to management. This includes better understanding of how fishery stocks respond to natural changes in ecosystems (e.g., decadal oscillations, El Nino-La Nina events) as well as the impact of reducing fished stocks and populations on ecosystem structure, function and services. As such, the MLMA requires management approaches that consider not only species taken commercially or recreationally, but to all marine wildlife and their habitats. The Act also requires development of Nearshore Fishery Management Plans (NFMPs) initially targeting 13 coastal finfishes. These NFMPs are to also consider more precautionary approaches to fisheries management than has been demonstrated by past management approaches. For these NFMP objectives and the broader ecosystem-based approach to management, the California Department of Fish and Game is to consider any existing or new management approaches, including marine reserves. More information on the objectives, process and state of implementing the Act is available at <http://www.dfg.ca.gov/mrd/mlma/index.html>.

Concurrently, the Marine Life Protection Act (MLPA) requires consideration of marine protected areas (MPAs) as one of several complimentary management approaches for conserving nearshore marine ecosystems. More information on the objectives, process and state of implementing the Act is available at <http://www.dfg.ca.gov/mrd/mlpa/>. Six goals of MPAs established in Fish and Game Code Section 2853(b) include the use of MPA's to:

- (1) protect the natural diversity and abundance of marine life, and the structure, function, and integrity of marine ecosystems,
- (2) help sustain, conserve, and protect marine life populations, including those of economic value, and rebuild those that are depleted,
- (3) improve recreational, educational, and study opportunities provided by marine ecosystems that are subject to minimal human disturbance, and to manage these uses in a manner consistent with protecting biodiversity,
- (4) protect marine natural heritage, including protection of representative and unique marine life habitats in California waters for their intrinsic value,
- (5) ensure that California's MPAs have clearly defined objectives, effective management measures, and adequate enforcement, and are based on sound scientific guidelines, and
- (6) ensure that the state's MPAs are designed and managed, to the extent possible, as a network.

## II. Why evaluate the effectiveness of marine reserves?

Management agencies are motivated to evaluate how well a marine reserve achieves the objectives it was created to accomplish for several reasons. Now, more than ever in the history of resource management, particularly in the state of California, management agencies are being held accountable for policy decisions and are compelled to provide both clear statements of reserve objectives and the means by which effectiveness will be measured. Indeed, like many other recently created marine reserve programs, the MLPA identifies potential “sunset clauses” that require assessment of reserve effectiveness every three years for the continued implementation of a reserve.

The accountability mentioned above stems from two concerns. First is the limited financial and human resources available to any resource agency to develop and implement management strategies. When limited, resources allocated to one management plan preclude their allocation to another. This becomes problematic when resources are allocated to the less effective of alternative approaches. Thus any management approach comes with a potential cost if either it does not succeed to achieve its objective or does so less efficiently than an alternative approach. Second is the perceived, potential, or realized impacts that regulations have on stakeholders, who for this reason demand accountability for regulatory policies.

Another concern for determining reserve effectiveness is to avoid a false sense of security or achievement when a reserve is assumed to be achieving its objects, but in fact is not. This is exacerbated if other regulations are relaxed because of the presumed precautionary role reserves are intended to achieve. Failure of any management approach, due either to poor design or evaluation, can potentially endanger the resources it was designed to protect. Moreover, poor evaluation can jeopardize the future of a management approach if it fails to identify the real value of a management approach or leads to an incorrect interpretation that the approach has failed. The sooner the benefit (or cost) of a management approach is evaluated and recognized, the more quickly that approach can be targeted for (or steered clear of) allocation of resources. Reserve design is unlikely to evolve, through adaptive management, to become more effective if the relative effectiveness of different designs are not determined and compared. As such, the design of realistic and achievable conservation targets, and the measurement of their effectiveness, will be crucial to the successful establishment of new MPA sites and to their long-term success through adaptive management (Carr and Raimondi 1999, Murray et al. 1999).

### *III. Approaches to evaluating reserve effectiveness*

Understanding the various approaches to evaluating reserve effectiveness and their relative costs and benefits is critical both to interpreting the value (and shortcomings) of past and existing studies and the design of future reserve evaluation programs. Evaluating the effectiveness of a marine reserve requires clearly identified goals and objectives, some knowledge of the many sources of uncertainty—process, model, causal, and measurement—both in reserves achieving objectives and in our ability to accurately evaluate them, and a well-designed evaluation program (Syms and Carr 2001). The design and scope of an evaluation program requires objective-based effectiveness parameters (i.e. response variables: population abundance or size distribution, species composition or diversity, habitat condition), targets (e.g., specified levels or directions of

each of these parameters or response variables), limits (acceptable deviations from specific targets), as well as a spatial (over what area a target is to be realized) and temporal (how soon and for how long a target is to be met) context (Syms and Carr 2001).

Reserve goals and objectives come from conceptual and theoretical inferences as well as empirically based inferences drawn from results detected in existing reserves (see *Section II* below). The effectiveness targets defined by these objectives may be of three forms. They may be absolute values to be attained over some defined spatial area and temporal period. For example, there may be some reason to target a particular density or abundance of a species within a reserve. Alternatively, a target may be a relative value, such as some percent increase in abundance or density within a reserve relative to non-reserve populations. Or, a target may be a rate of change in the difference between reserve and non-reserve effectiveness parameters over time (i.e. a pre-defined trajectory of the difference between reserve and non-reserve populations). There are strong arguments for any of these three forms of effectiveness parameters depending on the reserve objective and the particular effectiveness parameter.

Two critical components of an effectiveness parameter are the spatial and temporal scale of the parameter. For example, some parameters may be restricted to within the boundary of reserve (e.g., increased larval production) and others may be manifested over a far greater spatial expanse (e.g., larval dispersal to and replenishment of fished populations outside a reserve). Similarly, some parameters may be expected to respond rather rapidly after reserve establishment (e.g., change in population size structure of a fast growing species within a reserve) while others may take many years to fruition (e.g., the increased recruitment of a slow growing species into a catchable stock outside a reserve). For realistic spatial and temporal expectations and effectiveness targets, as well as appropriately designed sampling programs, some estimate of the spatial and temporal scales of an effectiveness parameter must be made.

The design and scope of an evaluation program will also depend on the timing and duration of the sampling or monitoring program relative to the establishment of a reserve. Three approaches exist, of which the worst-case scenario represents the vast majority of evaluation studies. This scenario occurs when only one reserve has been established long before an evaluation program is initiated. In this case, differences between the reserve and non-reserve treatment levels are confounded by all other site differences and cannot be unequivocally attributed to protection afforded by the reserve. It will never be clear whether observed differences (reserve vs. non-reserve) were caused by the reserve or if these differences already existed before the reserve was established.

If instead, one or more reserves is to be evaluated and sampling can be initiated at the proposed reserve site(s) and non-reserve “control” site(s) prior to reserve establishment, then inferences about reserve effects become much stronger. Two approaches are commonly used when ‘Before’ and ‘After’ data are available. The Impact vs Reference Site (IVRS) approach treats reserves and controls as formal randomized experimental replicates, and hence makes inferences about ‘reserve’ effects in general. IVRS requires that sites are truly independent and sites are assigned randomly to either reserves or control treatments (Stewart-Oaten and Bence 2001). In practice, often these conditions do not hold and so the alternative Before-After-Control-Impact (BACI) sampling design is used. BACI requires that reference sites be as similar

to reserves as possible, and is based on the model that temporal differences in sites are attributable to reserve effects. Consequently, BACI approaches make site-specific statements of reserve effectiveness.

BACI designs have been used more frequently in the literature, in particular to test for single coastal environmental impacts, and a rich literature on this design and analysis exists (Stewart-Oaten and Murdoch 1986, Stewart-Oaten and Bence 2001, and references therein). Non-reserve control sites at varying distances from the reserve (spatial gradient approach) may be incorporated into this design to examine the spatial extent of reserve effects. Alternatively, the effectiveness of a reserve at protecting a species targeted for exploitation can be determined by employing a BACI approach before and after exploitation begins as long as monitoring of that species in and out of the reserve has been conducted over that period. An excellent example of this approach is provided by Schroeter et. al. (2001) in which the status of an exploited invertebrate was monitored in reserves and fishing grounds before and after initiation of the fishery. This example demonstrates not only how a reserve can protect a population of a targeted species, despite strong declines on fishing grounds, but also the role of reserves for assessment of stocks and the impact of fishing. If instead evaluation sampling cannot be initiated prior to, but near, the time of reserve establishment, trends in the difference between reserve and non-reserve sites can be compared to determine if the sites are changing in predicted ways (i.e. increasing differences over time in density and mean size of individuals within reserves relative to non-reserves).

The design, scope and inferences drawn from an evaluation program will also be strongly influenced by the design of the MPAs to be evaluated (e.g., the number, size, distribution and environmental conditions). If only one reserve is to be evaluated, any inferences regarding the effectiveness of that reserve are largely constrained to only that reserve and cannot be generalized to reserves (more importantly, potential reserves) in general given the great environmental and biotic heterogeneity of the coastal marine environment. Any environmental characteristics (species composition, geologic or oceanographic conditions) unique to that reserve preclude generalizing how reserves in other areas would respond to protection. This is particularly true with respect to effectiveness targets that are relative differences between reserve and non-reserve sites because the relative differences (or trajectories) will depend on the magnitude of human impacts (e.g., fishing catch) outside the reserve. If multiple reserves and non-reserves can be sampled simultaneously, broader inferences regarding reserve effectiveness can be made (general reserve effects rather than the effect of a specific reserve). Moreover, environmental, design and management differences among reserves can be evaluated relative to one another. Such an approach is critical to the adaptive management of reserves. Thus, the design, implementation, analysis, inferences, and success of studies conducted to evaluate effectiveness of a marine reserve will be influenced greatly by, and therefore must consider, all of the criteria identified above.

## *II. Consequences of reserve establishment: theoretical and empirical inferences*

### *II.a. Conceptual inferences*

Many postulated consequences (hence, objectives) of reserves are simply intuitive and based on our limited understanding of the population, community and ecosystem ecology of marine organisms. Summaries of the hypothesized benefits of marine reserves include the Plan Development Team's NOAA-NMFS Technical Memorandum (1990), Bohnsack (1998), Murray et al. (1999), and the NRC (2001). Because these hypothesized or realized benefits translate into reserve objectives, they in turn identify parameters by which to measure the effectiveness of a reserve. These parameters are summarized in Table 1 and follow from the following "logic/argument".

*Conservation* -- For conservation purposes, protection of an intact ecosystem (i.e. biological communities and their geologic and oceanographic environment) contributes to the persistence of that ecosystem's ecological integrity (e.g., species interactions and physical-biological interactions), structure (e.g., species composition and relative abundance) and function (e.g., productivity, nutrient and mineral storage and cycling, habitat structure and integrity). Intact ecosystems in turn contribute to the persistence of the communities and populations that constitute it by providing them resources and maintaining species interactions that determine or regulate populations either locally or regionally. By protecting entire ecosystems it is presumed that, biodiversity, including rare and endangered species, is better protected. Species densities or abundances achieved in reserves in the absence of fishing mortality or habitat-altering activities are presumed to have stronger ecological interactions and effects more characteristic of their ecological roles in more "natural" or "pristine" ecosystems (e.g., Jackson et al. 2001). The greater integrity of such ecosystems is believed to lend them more stable (i.e. less variable over time), resistant (i.e. requiring stronger natural or anthropogenic perturbations, including invasive species, to cause them to change) and resilient (i.e. more likely to and quicker to return to a pre-perturbation state). Thus, several characteristics of ecosystems (e.g., productivity, nutrient cycling, habitat quality) are targets for management and parameters of reserve effectiveness (Table 1).

The same benefits are presumably conveyed to communities and populations that constitute protected ecosystems. Thus, attributes of natural communities including species diversity, community structure, as well as natural densities of species that are either ecologically important in maintaining community structure or whose persistence reflects intact community/ecosystem structure are targets for assessing whether a reserve is influencing the state or trajectory of a community (Table 1). Species interactions of particular importance are those that have community-wide consequences including the effects of keystone predators in contributing to the maintenance of diversity, cascading trophic interactions, and the persistence and natural densities of habitat forming species. In coastal temperate reef systems, examples of predators that have strong cascading trophic effects are common (see recent reviews by Babcock et al. 1999, Pinnegar et al. 2000, and Carr et al. in press).

At the population level, reduced fishing mortality and habitat protection within a reserve should lead to increases in both density (and abundance) and the average size of fishes in a

population. Combined, these two attributes should lead to increased larval production of a protected population relative to populations that are growth or recruitment overfished. These higher population densities can be more resistant and resilient to the combined effects of natural perturbations (e.g., storms, El Ninos) and human impacts. As for ecosystem and community objectives, restrictions on activities that destroy or diminish habitat quality also enhance persistence of populations within reserves. Because the genetic diversity of any population (hence it's potential ability to adapt to changes in environmental conditions) increases with the number of individuals in that population, protecting populations within reserves increases greater genetic diversity and the ability of that population to persist in the face of environmental perturbations and change (e.g., diseases). Thus, both population and environmental attributes constitute critical effectiveness parameters (Table 1). Importantly, because of the great dispersal potential of offspring (spores, eggs, larvae) produced by many marine species, populations protected within a reserve can contribute to the replenishment (hence, conservation) of populations of these species well beyond the border of reserves (Carr et. al. in press, Shanks et al. in press, Kinlan and Gaines in review).

*Fisheries conservation/management -- One role of marine reserves posited for fisheries conservation overlaps greatly with the goals of conservation reserves. That is the precautionary role, or security of spawning stocks and ecosystems protected from fishing effects in light of the difficulty of regulating fishing effort through most traditional management approaches (Larkin 1977, Ludwig, et al. 1993, Botsford, et al. 1997, Guenette, et al. 1998, Lauck et al. 1998). Thus, the protection and sustainability of populations of fished species within reserves is a fundamental objective of reserves created for the conservation of fisheries (Table 2).*

*In addition, because of the great dispersal potential of reproductive propagules (spores, eggs, larvae) produced by many marine organisms (referred to as "larval export"), and the potential movement of benthic juvenile and adult fishes from within to outside reserves (referred to as "spillover"), some benefits accrued by populations protected within reserves can be exported to populations outside reserves. This has been a leading consideration for the potential application of marine reserves for the conservation or management of fisheries. Propagules transported to populations outside reserves can replenish exploited populations to counter "recruitment overfishing" (when the reproductive potential of a population has been reduced to the extent that larval production and recruitment are diminished). Because most marine fishes exhibit a positive non-linear increase in fecundity with body size (mass), fish allowed to survive and grow to older ages and larger sizes have a disproportionately higher per-capita production of larvae (Bagenal 1978, Thresher 1984). Indeed, size-fecundity relationships for rockfishes (Genus Sebastes) exhibit this non-linear relationship (Love et al. 1990, Love and Johnson 1999). (Note that the linear relationship described by Gunderson et al. (1980) reflects the narrow size range of fishes sampled.*

Table 1. Effectiveness parameters for individual and networked conservation reserves.

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I. Species population parameters

Abundance  
Density  
Size structure  
Age structure  
Size specific fecundity  
Larval production (product of density and size specific fecundity)  
Spawning biomass  
Population stability  
Population resilience  
Population resistance  
Genetic diversity (within and between populations throughout network)  
Demographic rates (reproduction, mortality, immigration and emigration)  
Mean individual growth rates  
Local population viability estimates  
Larval dispersal (to assess extent to which reserve populations are self-replenishing)  
Connectivity of larval dispersal with other reserves  
Species-specific habitat quality and abundance

II. Community parameters

- A. Focal species (e.g., rare, endangered, keystone, indicator, umbrella and flagship species)  
- All or subset of species population parameters identified above  
- Emphasis on interaction strengths and effects of keystone and exploited predator species
- B. Community-wide  
Species composition  
Species richness  
Relative densities of species  
Species diversity  
Trophic richness  
Trophic diversity  
Trophic structure  
Guild structure and dynamics  
Species redundancy  
Species interactions and strengths (e.g., competition, predation, parasitism, mutualism)  
Community stability and dynamics (e.g., resistance, resilience, constancy and persistence)  
Spatial relationships of populations  
Community function (e.g., primary and secondary productivity)  
Breadth of resource use (e.g., dietary breadth of predators)



Complementarity  
Genetic diversity and structure  
Threshold effects—potential alternative stable states

### III. Ecosystem parameters

Habitat structure (size, shape, spatial arrangement of habitats)

Habitat richness

Habitat diversity

Habitat representativeness

Physical (structural) complexity (of abiotic and biotic substrata)

Interactions between biogenic physical structures and species that alter them.

Productivity (C gm fixed / area / time; total and by trophic level)

Nutrient and matter cycling and fluxes (e.g., rates of change, rates of cycling, fluxes,  
nutrient ratios, nitrogen fixation)

Detrital production and export.

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Table 2. Effectiveness parameters for individual and networked fishery reserves

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I. Population parameters

A. Local (within reserve)

Abundance

Density

Size structure

Age structure

Size specific fecundity

Larval production (product of density and size specific fecundity)

Spawning biomass

Mean individual growth rates

Demographic rates (reproduction, mortality, immigration and emigration)

Population stability and dynamics (e.g., resistance, resilience, constancy and persistence)

Genetic diversity (within and between populations throughout network)

Local population viability estimates

Larval dispersal (to assess extent to which reserve populations are self-replenishing)

Density, dynamics, and stability of by-catch species

B. Regional (outside reserve)

Larval production and export rate (from inside to outside reserve)

Larval dispersal and recruitment patterns (outside reserves)

Emigration (i.e. “spillover”) and immigration of benthic stages inside and outside of reserves

Stock stability and dynamics (e.g., resistance, resilience, constancy and persistence)

Fishery yield

II. Community parameters

A. Local (within reserve)

By-catch assemblage composition, structure, dynamics, and stability

Density, dynamics, and stability of resource requirements for exploited species

B. Regional (outside reserve)

Community stability, to extent that reserves contribute to regional stock abundance and stability, and exploited species influence community structure

III. Ecosystem parameters

A. Local (within reserve)

Abundance and quality of spawning, recruitment and other habitat requirements

Abundance and quality of other ecosystem-based resource requirements

B. Regional (outside reserve)

Ecosystem stability, to extent that reserves contribute to regional stock abundance and stability, and exploited species influence ecosystem structure

Ecosystem stability, to extent that reserves contribute to production and export of ecosystem components (e.g., larval export and replenishment of biogenic habitat)

Thus, individuals protected in reserves can contribute disproportionately (relative to their numbers) to larval production and recruitment to fished populations. Moreover, there is growing evidence in the fishery literature that older females produce young of greater physiological condition (Zastrow et al. 1987, Chambers and Legett 1989, Buckley et al. 1991) that translates into higher growth and survival of those larvae (Blaxter and Hempel 1963, Houde 1987, Hislop 1988, Monteleone and Houde 1990). Indeed, this “maternal effect” of female age on larval condition and survival has been demonstrated in laboratory studies of the black rockfish, *Sebastes melanops* (Steve Berkeley, personal communication). This may increase further the proportionate contribution of larval recruitment to fished populations by older females protected in reserves. These contributions to larval recruitment become more important when considering how quickly populations rebound from disturbances. If populations of larger, older females in reserves are less vulnerable to disturbances (because the greater stability and resiliency of larger populations and intact ecosystems), and fished populations outside reserves are recruitment overfished, populations in reserves can produce and export large numbers of recruits to fished populations after disturbances or during episodic environmental opportunities for recruitment. Thus, reserves may act to hasten the rate that fished populations rebound from perturbations or take advantage of episodic environmental conditions advantageous for recruitment. For all these reasons, export or emigration to, and many predicted responses of, fished populations outside of reserves are potential parameters of effectiveness of fisheries reserves (Table 2).

If individuals in a reserve grow to sizes they otherwise would typically not achieve in a fished population, movement of these individuals from reserves to fished populations (i.e. “spillover”) acts to increase the average size of fishes caught in the fishery adjacent to a reserve. This replenishment of larger individuals in a fished population counters “growth overfishing” (reductions in the average size of fish in a fishery caused by removal of larger individuals). The spatial extent of this benefit depends on the range of individual movement of a species. The movement of individuals (larvae, juveniles or adults) from reserves to fished populations increases the genetic diversity of fished populations and provides the benefits of greater genetic diversity to fished populations described in the *Conservation* section above.

Another important potential application of reserves for both fisheries conservation and management is their role as “reference” or “control” sites to assess the influence of fishing activities on marine populations and ecosystems. It is extremely difficult to tease apart the effects of fishing and natural variation in the environment from one another without populations that are not subjected to one or the other. Comparison of populations (and ecosystems) subjected to and protected from fishing may allow the perceived or hypothesized effects of fishing to actually be tested (and therefore supported or refuted). This is critical to understanding how different levels of fishing mortality and related activities influence stocks and their ecosystems.

## *II. b. Theoretical (model-based) inferences*

Conservation -- A large number of models developed for conservation reserves have focused on issues of design theory rather than exploring reserve effects. There is a surprising dearth of mathematical models developed to predict or understand how communities or ecosystems would respond to protection by marine reserves. Perhaps to some extent this reflects the variety and complexity of coastal ecosystems, which might require more specific mechanisms, knowledge of

the relative openness of the various populations, and therefore more specific (not necessarily more detailed) models to be useful. For example, some models have identified the importance of the strength and stage at which density dependence occurs (traits that vary widely among species in a community) in determining how a population will respond to relaxed fishing mortality (Hastings and Botsford 1999, Walters et al. 1999, St. Mary et al. 2000).

Fisheries conservation/management -- In contrast to the paucity of models developed to explore the effects of conservation reserves, a large number of models have been constructed to examine the potential of reserves for fisheries purposes. One recent summary of these models is provided in the NRC (2001) review of marine protected areas (Table 6.3, Appendix G.). Reserve models can be categorized by four primary objectives; (1) their role as a precautionary means to ameliorate accidental overfishing, (2) their potential contribution to ameliorate growth overfishing by exporting older life stages to a fishery (i.e. spillover), (3) their effect on fishery yield and how their effect relates to and compares with more traditional approaches to controlling fishing effort, and (4) predictions of socio-economic consequences to a fishery. This last category (socio-economic consequences) is dealt elsewhere in this report.

Precautionary management -- Several recent essays have recognized the difficulties of traditional management approaches to controlling fishing effort and catch. These difficulties stem in large part on problems of stock assessment, the great natural variability of marine fish populations, and the uncertain political structure of management decisions (Larkin 1977, Lauck et al. 1998, Hilborn and Walters 1992, Mangel 2000a). Thus, many models have examined the potential role of reserves in contributing to more precautionary approaches to fisheries management and the long-term sustainability of fisheries (Goodyear 1993, Mace and Sissenwine 1993, Mace 1994, Mann et al. 1995, Lauck et al. 1998, Roughgarden 1998, Soh et al. 1998, Foran and Fujita 1999, Guenette and Pitcher 1999, Guenette et al. 2000, Mangel 1998, 2000a, 2000b, 2000c, NRC 1999, 2001). These models identify that the proportion of fished stocks protected within reserves depends on rates of fishing mortality and that the larger the portion protected within a reserve, the more precaution afforded the fishery. These models predict reasonable to very high proportions (20-70%) of a stock, depending on the species, are typically necessary to be set-aside within reserves in order to contribute to sustainability when fishing mortality varies from moderate to very high (i.e. the “scorched earth” scenario) rates, respectively. Because these models are developed to examine a reserve’s precautionary role, these worst-case scenarios are often applicable.

*“Spillover”* -- A small subset of fisheries reserve models have focused on potential effects of fish emigration from a reserve (i.e. “spillover”) to a fishery (Polacheck 1990, Russ et al. 1992, DeMartini 1993). These models suggest that emigration of adults into fished areas can lead to increases in yield per recruit, but these increases occur only when emigration rates are moderately high and fishing mortality outside the reserve is high, but regulated. Thus, these models imply that knowledge of rates of bi-directional movement of exploited species into and out of reserves, and of the various factors that contribute to movement rate, direction and distance (e.g., fish size or age, density, habitat attributes or quality) can allow inferences of the contribution of “spillover” to yields (and increased fish sizes) in adjacent fisheries. These models also indicate that the contribution to increased yield is constrained to areas close to the reserve. Therefore, sampling designs to detect the degree and spatial extent of the spillover

effect should incorporate an appropriate sampling gradient (e.g., for a linear coastline, sampling concentrated toward the reserve boundary by distributing samples by the square root of distance from the boundary). These studies have two additional important implications for assessing the effectiveness of existing reserves in the MBNMS. First, is that the effect of a reserve in exporting larger individuals (i.e. “spillover”) to a fishery depends on (1) the size of an individual reserve relative to movement patterns and ranges of a targeted species, (2) the degree of growth overfishing outside a reserve, and (3) the intensity of fishing within the range of spillover from a reserve. To date, no studies have directly examined spillover from a reserve within MBNMS, nor has information on the degree of growth overfishing or location and intensity of fishing outside and adjacent to reserves been collected or examined. However, information on fishing adjacent to Big Creek Ecological Reserve is being collected (John Smiley, pers. comm.). But also critical to this evaluation is information on movement ranges of targeted species and how that relates to the size of existing reserves. Without such information it is very difficult to estimate rates of spillover and contributions of existing reserves to their potential in countering growth overfishing and the size distribution of catches in a fishery.

*Fishery yield* -- Most fishery reserve models have examined the effects of reserves in contributing to total fishery yield as well as its sustainability and reducing its temporal variability. Most have compared the effects of varying total reserve size with levels of effort control by traditional management approaches. Overall, these models indicate that the effect of adding reserves on total yield is essentially the same as decreasing fishing mortality (reviewed by Botsford et. al., in review). Mangel (1998, 2000b) demonstrated that yield depended on the product of fishing mortality and area not in reserves, not on the specific values of each. Predictions of how reserves contribute to increasing yield vary widely among these models. Some models suggest that comparable yields can be achieved through traditional effort control and reserves (Mangel 1998, 1999, 2000a, Hastings and Botsford 1999). Several studies indicate that reserves can increase yields only when stocks have been heavily overfished (Quinn et al. 1993, Holland and Brazee 1996, Sladek Nowlis and Roberts 1997, 1999, Guenette and Pitcher 1999). These models suggest that the proportion of a stock necessary to be set aside within reserves to enhance yields ranges from 20 to 50% (NCR 2001, Table 6-3). Thus, predictions of the magnitude to which a reserve will influence yield from stocks outside reserves will depend not only on the proportion of a stock protected in a reserve, and larval production and export, but also on the state of the fishery and continued level of exploitation. Thus, any study measuring the relative effectiveness of a reserve in protecting or enhancing populations either inside or outside a reserve will benefit greatly from (if not absolutely require) good estimates of spatial and temporal patterns of fishing mortality. Optimally, unequivocal conclusions regarding the effect of a reserve on fishery yield are drawn from comparisons of yield between independent fished stocks before and after, as well as with and without, presence of a marine reserve (i.e. a BACI design with or without replicate stocks). To date, no such comparisons exist. The spatial scale of the comparison (i.e., independent fish stocks) and the temporal scale that would provide sufficient statistical power to distinguish these effects will require more information and much longer continuous studies than currently exist, especially along the west coast of the United States.

*Fishery variability* -- Some models have suggested that reserves can contribute to the resiliency of a stock (Foran and Fujita 1999), or dampen the magnitude of temporal variability (Sladek

Nowlis and Yoklavich 1998, Sladek Nowlis and Roberts 1999). Determining to what extent a reserve actually contributes to resiliency (how fast population size returns to a pre-disturbed level following a disturbance) or dampen temporal variability of a fished population is very difficult. As in the case of assessing reserve effects on yield, unequivocal conclusions are drawn from comparison of these variables (resiliency and variability) between independent fished stocks before and after, as well as with and without, presence of a marine reserve (i.e. a BACI design). To date, no such comparisons exist. The spatial scale of the comparison (i.e. independent fish stocks) and the temporal scale that would provide sufficient statistical power to distinguish these effects will require more information than currently exists, especially along the west coast of the U. S.

One very important conclusion and implication from all of the above models considering reserves for fishery conservation or management is that measurable (detectable) effects of reserves on fisheries yield and variability require total reserve areas (across the entire network) that are far larger than the existing reserves throughout the MBNMS, California, and the West Coast. This is very important when interpreting assessments of reserve effects from the existing reserves within the MBNMS. Likewise, a large body of models has been developed to examine and optimize the design of fisheries reserves. These models examine the effects of reserve design (specifically, the overall size of a reserve network, individual reserve area, and the number and spacing of reserves in a network).

### *II. c. Empirically-based inferences*

*Conservation* -- There is widespread empirical evidence of the predicted increases in population density, biomass and size frequency of exploited species to protection within reserves. Recent syntheses of examples include Jones et al. (1992), Rowley (1994), Roberts and Hawkins (2000), NRC (2001), and Halpern (in press). Halpern's review is the most comprehensive, having surveyed 89 separate studies, 41% of which were in temperate ecosystems. Halpern examined three population level variables (density, biomass, and average size of organisms) inside and outside of reserves or before and after reserve establishment. Relative to non-reserve references (either before establishment or areas outside reserves), 63% of reserves had higher density, 90% of reserves had higher biomass, 80% of reserves had larger organisms, and 59% of reserves had higher diversity (all Chi-square analyses significant at  $p < 0.001$ ). This pattern was consistent across the four functional groups examined (carnivorous fishes, herbivorous fishes, planktivorous fishes/invertebrate eaters, and invertebrates), with the exception of invertebrate biomass and size. A small number of reserves had lower values for these three population measures (7%, 0%, and 2% of reserves had lower density, biomass, and organism size, respectively). Although there was great variance in the magnitude of these effects among reserves, density doubled, biomass nearly tripled, and organism size increased by 20 to 30% relative to the values for unprotected areas. Thus, these three desired and predicted responses of species to protection within a reserve are reasonable parameters by which to examine and evaluate reserve effectiveness.

In addition to the 3 studies conducted within marine reserves within the MBNMS--described in detail in this report--specific examples from temperate reserves along the West Coast include Palsson's (1998) surveys of rockfishes at 7 sites in Puget Sound, two of which were no-take

marine reserves. Results varied among species, largely corresponding to fishing intensity. Heavily and moderately fished species like copper and black rockfish, respectively, were at significantly greater densities within reserves. Quillback rockfish exhibited a similar result for larger individuals. In contrast, densities of lightly fished species (brown and Puget Sound rockfish) did not differ significantly between reserve and fished sites. Length frequency distributions of copper and quillback rockfish were also shifted to larger sizes within reserves. In a separate study, lingcod density was 3 times greater in a Puget Sound reserve compared to fished areas (Palsson and Pacunski 1995). Another example of increased density of an exploited temperate reef fish within reserves is provided by Martell et al. (2000). The density of spawning lingcod, *Ophiodon elongatus*, was significantly greater in two small reserves than in adjacent fished areas within the Straits of Georgia.

Similar patterns of increased density and individual size has been demonstrated for exploited species on temperate New Zealand rocky reefs (Jones et al. 1992). For example, McCormick and Choat (1987) recorded that abundance of the red moki, *Cheilodactylus spectabilis*, was six times higher in a New Zealand marine reserve than adjacent sites. Similarly, increased density and sizes of snapper (*Pagurus auratus*) and spiny lobster (*Jasus edwardsii*) occurred in New Zealand reserves (Babcock et al. 1999). Snapper were 6 to 9 times more dense and 50% longer within reserves. Lobster were 1.6 to 3.7 times more dense and had an average of 1.6 cm longer carapace length. In the first before-after comparison of a temperate reserve in New Zealand, Cole et al. (1990) also found similar increases in abundance of all three of these previously mentioned species as well as the commercially important blue cod (*Parapercis colias*), and decreases in 3 others. Commercially taken snails (*Concholepas*) in the temperate intertidal of Chile increased in density 5 to 14 times and doubled in body size in reserves relative to exploited areas (Castilla and Duran 1985). Rock lobster (*Jasus*) and a reef fish abundance increased by 1 and 2 orders of magnitude on temperate rocky reefs within reserves in Tasmania (Edgar and Barrett 1999). The more persistent (and increased) size of populations of the warty sea cucumber within reserves in spite of simultaneous declines on fishing grounds in the Channel Islands of California provide another example of protection afforded species within reserves on temperate rocky reefs (Schroeter et al. 2001). These documented responses from other temperate ecosystems suggest that increases in densities of lobster, finfish and other species seem reasonable to anticipate and test for in California, particularly in heavily fished regions.

Community and ecosystem responses have focused on changes in species diversity and cascading effects of ecologically influential species protected from fishing. In Halpern's (in press) comprehensive review mentioned in the preceding section, he also examined differences in species diversity, inside and outside of reserves or before and after reserve establishment. Fifty-nine percent of reserves had higher species diversity (Chi-square analysis:  $p < 0.001$ ) than their corresponding non-protected references. This pattern was also consistent across all four functional groups examined (carnivorous fishes, herbivorous fishes, planktivorous fishes/invertebrate eaters, and invertebrates). Diversity decreased within reserves in 10% of the studies reviewed. Although variable among studies, on average, diversity increased by 20 to 30% relative to the values for unprotected areas.

Actual examples of the cascading effects created either by reduced human predation or by increases in predator density in temperate reserves are few (reviewed by Steneck 1998, Castilla

1999, Pinnegar et. al. 2000) at least in part because very few studies have tried to assess them. Those that are most conspicuous involve the role of predators that influence major grazers (especially sea urchins) that in turn influence the cover and density of macroalgae (Pinnegar et al. 2000). One particular example provided by Babcock et al. (1999), was the cascading consequences of increased densities of snapper and lobster in reserves. With increased densities of these two sea urchin predators within reserves, urchin densities declined and macroalgal cover increased significantly, relative to fished areas outside reserves. Similar consequences of the cascading effects of increased densities of urchin predators within reserves has been described in coral reef systems (Sala et. al. 1998, McClanahan 2000). These general responses have two important implications for measuring potential responses of communities and ecosystems inside reserves established in areas where sea urchin predators or sea urchins themselves are heavily fished. Community responses to protection within the MBNMS and adjacent temperate rocky reef ecosystems might exhibit broader cascading responses to protection, but the response is likely to vary markedly with regional variation in community structure. In southern areas (e.g., the Channel Islands and Southern California Bight) where urchin predators including sheephead and lobster are fished, effects of protection from fishing is likely to cause cascading effects on urchins and macroalgal assemblages. Predicted increases in macroalgal stands in turn is likely to influence the recruitment of many reef fishes (Carr 1989, 1991, 1994, Carr and Reed 1992). In contrast, throughout Central California and the MBNMS, sea otters appear to limit sea urchin density and protection of finfishes or invertebrates within reserves is less likely to influence urchin and kelp densities. Further north, in areas of Northern California where human take of urchins is the primary determinant of local sea urchin density, increased urchin densities within reserves is likely to reduce macroalgal stands, an effect opposite of that predicted for Southern California. Here, another important indirect effect of protection of sea urchins within reserves is their apparent influence on recruitment of abalone. Rogers-Bennett and Pearse (2001) describe significant increases in abalone recruitment in the presence of sea urchins within reserves. This influence of sea urchins on abalone recruitment appears to occur over a broad geographic range, including Southern California (Tegner and Dayton 1977). Hence, conceptual and empirically-based inferences suggest that such indirect effects are critical measures of reserve effectiveness with respect to ecosystem and biodiversity objectives in the MBNMS.

Fisheries conservation/management -- Global impacts of reserve implementation on adjacent fisheries have been recently reviewed elsewhere (Rowley 1994, Roberts and Hawkins 2000, NRC 2001). We focus this discussion on examples from coastal temperate examples from the northern hemisphere that bear more directly on potential effects of reserves within the MBNMS.

Mentioned previously, an example of temperate reserves both protecting populations of a stock declining on fishing grounds and providing a more accurate assessment of fishery trends and impacts on the stock was provided by Schroeter et al. (2001). These investigators compared long-term monitoring trends of the warty sea cucumber (*Parastichopus parvimensis*) within and outside reserves prior to and during exploitation of that species at the Channel Islands off Santa Barbara, California. The assessment demonstrated that populations within reserves remained constant (or increased!) as populations on fishing grounds declined. Any differences in cucumber densities between these sites prior to the fishery were significantly less than differences in density over the 7-year period following initiation of the fishery.



Although fishery closures on Georges Bank were established to enhance recovery of finfishes, populations of the scallop, *Placopecten magellanicus*, increased 14-fold over a 4-year period within reserves relative to adjacent trawled areas (Murawski et al. 2000). In late 1994, three large areas on Georges Bank and in Southern New England, totaling 17,000 km, were closed year-round to any gears capable of retaining groundfish (trawls, scallop dredges, gill nets, hook fishing). Four years later, by the end of the study, total and harvestable scallop biomasses were 9 and 14 times denser, respectively, in closed than in adjacent open areas. Moreover, spatial patterns of subsequent recruitment of young scallops outside the reserve strongly suggested that spawning stocks protected within the reserves were largely responsible for larval export and replenishment of adjacent fished populations.

*“Spillover”* -- More empirical studies on the movement of adult fishes relative to reserve boundaries have been conducted on coral reef fishes in the tropics than on temperate rocky reef fishes. Recent and notable examples include McClanahan and Mangi (2000) and Roberts et al. (2001). One rather obvious but important pattern that emerges from these tropical studies are that individuals whose home ranges straddle reserve boundaries move frequently in and out of reserves, while individuals whose home ranges are either entirely within or outside the reserve, may rarely cross reserve boundaries (Zeller and Russ 1998). Also critical to predicting and interpreting rates of spillover is knowledge of how emigration responds to the local density of conspecifics (i.e. density-dependent movement). Predicted increases in survival and density within reserves, relative to continued depletion of conspecifics outside reserves creates a potential density gradient that could induce individuals to emigrate to adjacent fishing grounds (St. Mary et al. 2000, Sanchez Lizaso et. al. 2000). Of temperate zone studies, Attwood and Bennett (1994) used a tag-recapture approach to estimate emigration of galjoen (*Coracinus capensis*) from a surf-zone reserve in South Africa. Over 5.5 years, roughly 9% of the tagged fish were recaptured, of which roughly 18% had emigrated from the reserve to the adjacent fished areas. However, movement in the opposite direction (immigration into the reserve) was not estimated. Thus, knowledge of the distribution, relative movement rates, and factors that contribute to movement (e.g., density and habitat quality) are critical information for predicting and interpreting spillover from a reserve.

*Fishery yield* – We are unaware of any empirical studies in temperate oceans that have explicitly examined effects of marine reserves on fisheries yield; none have done so along the west coast of the United States. Theoretical studies indicate that total fisheries yield will only increase when large reserves are present in a highly fished population (Hastings and Botsford 1999), and modeling studies suggest that local increases in yield will be dependent on reserve size and movement rates (Polacheck 1990, DeMartini 1993, Attwood and Bennett 1995). Empirical studies have generally interpreted changes in abundance with distance from reserve boundaries as potential increases in yield (eg. Cole 1994). However, yield increases have been recorded adjacent to a tropical marine reserve in the Philippines Islands (Alcala and Russ 1990), and reported from St. Lucia (tropical Caribbean) and sub-tropical Florida (Roberts et al. 2001, but see an alternative interpretation by Tupper et al. 2002). These empirical studies demonstrate that increases in yield are generally localized, and should be compared to overall fisheries yield. It is unlikely that existing marine reserves in California contribute substantially to fisheries yield. The small size of existing reserves would probably make any contribution to total yield insignificant.

The small boundary interface of most reserves with fished areas, in combination with the small size of any abundance differences between reserve and adjacent non-reserve areas, would in all likelihood preclude any local perception of yield increases.

*III. What have past and present studies in existing California reserves addressed and concluded?*

Only three major studies have explicitly addressed the effectiveness of subtidal marine reserves within the MBNMS. All three studies have focused solely on the response of selected fish species to protection within reserves, and are more thoroughly summarized elsewhere in this report. In general, these studies have by and large reinforced the most common and conspicuous pattern of responses of exploited species to protection within a reserve. Estes and Paddock (see also Paddock and Estes 2000) compared fish density and sizes inside and out of three reserves within the MBNMS (Big Creek Ecological Reserve, Point Lobos Ecological Reserve, and Hopkins Marine Life Refuge) and found a statistically non-significant trend of higher densities of fishes in reserves compared to fished areas outside reserves. However, the average length of rockfish (genus *Sebastes*) was significantly greater in 2 of the 3 reserve sites, as was the proportion of larger fish. In combination, the greater population density and sizes combined to produce substantially greater biomass and therefore potential larval production per unit of area within the reserves. Importantly, the magnitude of these effects seemed to be correlated with the age of the reserve.

The second and third studies examined fish assemblages inside and outside of Big Creek Ecological Reserve. VenTresca et al. conducted diver surveys the first 4 years (1995-1998) following reserve establishment. They detected no temporally consistent pattern of difference in density or size of fishes inside and outside of the reserve. The third study (Yoklavich et al.) examined fish assemblages inside and outside of the reserve 3-4 years (1997-1998) after reserve establishment. Again, no significant difference in density of any species inside and outside of reserves was detected. Moreover, they did not detect any significant difference in size of seven economically valuable species. These comparisons were of combined densities across species. Individual species comparisons were not made.

None of the three studies conducted in MBNMS reserves to date found significant differences in fish densities between reserve and non-reserve areas. However, Paddock and Estes found weak evidence that fish were larger in two of the three reserves. The other two studies were conducted within 4 years of reserve establishment, and it is unlikely that any effects of protection would have had time to be manifested. It is likely that even small increases in the number of large fish will exert a disproportionate effect on the potential larval production of the protected population (refer to section *II.a. Conceptual inferences: Fisheries conservation/management*). Given the extended larval durations (1-4 mo) and dispersal potential for most of the fishes that demonstrate increased potential larval production, export to and replenishment of populations outside reserves is very likely. However, given the small size and limited number (i.e. low probability of connectivity) of reserves in this region, it is unclear and perhaps unlikely that these reserves function as a network that could buffer the effects of any overfishing from surrounding areas (Carr and Reed 1993, Botsford et al. 2001).

#### *IV. Filling in the gaps: critical directions for future research*

To develop a better understanding for the potential of reserves as tools for conservation and fisheries management, it is clear that far more monitoring effort will be required. In addition to more information on the responses of targeted species, community-level and ecosystem-wide responses need to be examined. Of the many effectiveness parameters identified from the literature (Tables 1-3), few have been explicitly assessed in these studies. Without continuous monitoring, temporal persistence of differences between reserve and non-reserve populations will remain unknown. With the present system of reserves and monitoring effort, unequivocal ascription of differences between reserve and non-reserve populations to causal reserve effects is tenuous. Better understanding of the consequences of reserve establishment requires both creation of new reserves in concert with well-designed monitoring studies over many years. Evidence from a great variety of theoretical and empirical studies across a wide diversity of coastal ecosystems indicates that reserves have the potential to achieve many conservation and some fisheries management objectives. If reserves are to be developed further and successfully used as a management tool, it is necessary to develop a structured and well supported monitoring program, which clearly identifies a set of effectiveness parameters that measure how well a reserve achieves a stated objective.

#### *Evaluation design and analysis*

All three of the published evaluation studies on MBNMS MPA's have been "snapshots" of population and community states after implementation of reserves. Consequently, inferences about reserve 'effects' are weak. However, even if a suitable formal approach is employed (BACI or impact vs reserve analysis) it is still not sufficient to just measure the response variables when evaluating MPA effectiveness. A range of other features of the habitat, reserve configuration, and exploitation pattern alter both empirical and theoretical evaluation (Table 3).

Table 3. Covariates required to assess MPA effectiveness

| <u>Habitat variables</u>                                   | <u>Reserve characteristics</u>   | <u>Exploitation variables</u>                                     |
|--|----------------------------------|---|
| 1. Physiographic habitat structure                         | 1. Size                          | 1. Total fishing effort   |
| 2. Oceanographic environment                               | 2. Shape                         | 2. Spatial location of fishing effort                             |
| 3. Biotic habitat structure                                | 3. Age                           | 3. Targeted species   |
| 4. Connectivity with other biotic and abiotic habitats     | 4. Enforcement effectiveness     | 4. Effect of other effort control regulations in managing fishery |
| 5. Climatic variation                                      | 5. Reserve network configuration | 5. Gear type and effects on habitat                               |
| 6. Proximity of essential habitats (e.g., nursery grounds) | 6. Edge permeability             | 6. Bycatch composition and abundance                              |
| 7. Potential for other threats (e.g., pollution)           |                                  | 7. Temporal distribution of fishing effort                        |
| 8. Natural disturbance regime                              |                                  | 8. Amount of area fishable by different gear types                |

These effects will strongly influence the ability of monitoring designs to make statements of effect. BACI analyses, for example, require that the underlying dynamics of reserve and non-reserve, with the exception of the reserve ‘effect’, be correlated. Conversely, an IVRS approach requires that the dynamics of reserve and non-reserve be independent.

Marine populations and communities are subject to a range of forcing processes, such as oceanographic and climate regimes, that can alter both population and community dynamics and might alter the synchrony of reserve and non-reserve areas. Other variables might alter the strength of interactions between populations in different reserves. For example, habitat quality and structure might alter the replenishment and survivorship patterns of ‘response’ organisms. This extra variability, in combination with events such as natural or anthropogenic disturbances, might obscure or confound effectiveness measures.

The history and ability of the reserve to actually protect the organisms is also important to incorporate into any analysis. Temporal lags in species population dynamics might define a temporal limit to reasonably expect a protection effect. The size and shape of the reserve and the permeability of its boundaries to emigration, in combination with the degree of enforcement are important to establishing the size of the stock that is actually protected from fishing mortality.

Exploitation patterns are one of the least appreciated sets of variables that should be included in MPA evaluation. Many models assume complete removal of stock from fished areas, but this is rarely the case. It is important to measure the actual extraction rate, the location and temporal structure of extraction, and other issues such as gear type, bycatch, and gear

disturbance issues. If fishing mortality is comparatively low, then it is unrealistic to expect that an MPA will have significantly higher abundances of an exploited species. This of course does not argue against MPA's as a management tool, but suggests that their effectiveness lies not in augmenting a fishery, but more as an insurance policy.

One of the most important classes of covariables are those that correspond to the linkages between the MPA and surrounding habitats. Modeling approaches are sensitive to boundary conditions, i.e., assumptions about limitations to values that go beyond the model domain. In nature, boundaries are defined by oceanography and physical habitat structure. In addition, many models make assumptions about connectivity (e.g., a common larval pool, or an assumed larval dispersal distance) that are unlikely to be realistic or precisely estimated. Connectivity in marine systems operates on at least two levels. Oceanographic connectivity is of central importance to understanding linkages between stocks in different locations. Features such as eddies and longshore currents will have very different effects on larval connectivity. Landscape connectivity is important for identifying critical bottlenecks in ontogeny (e.g., proximity of reserve to nursery or spawning habitats) and recognizing variability due to permeability of habitat types between reserve and non-reserve areas.

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## CHAPTER 6

### **Effectiveness of Marine Reserves: Socio-Economic Considerations**

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#### **Abstract**

Marine reserves require careful consideration of a range of socio-economic, or more accurately, human dimensions factors, including how people and communities perceive, value, and use the marine environment, and the informal and formal rules and structures that influence these values and uses. The rationale for attending to these is both legal and practical, and follows from the understanding that 1) marine reserves generate social and economic impacts which both individuals and society must bear; 2) human responses and adaptations to marine reserves have ecological as well as socio-economic implications; and 3) attention to the human dimensions can afford resources that are essential to marine reserves processes and outcomes. Public participation and social science research are complementary means for addressing the human dimensions of marine reserves. Public participation is useful for identifying and addressing social, economic, and ethical concerns and issues associated with marine reserves, and generating trust, support, and other social resources to assist in their design, management and evaluation. It cannot, however, substitute for systematic collection and analysis of socio-economic data to inform the process. To date, however, limited social science research has been done, with the majority focused on identifying the necessary and sufficient conditions for the establishment of marine reserves. Although the theory of marine reserves posits that social and economic benefits will outweigh their costs, little research to test this hypothesis, or to inform and evaluate the human dimensions of marine reserve design, implementation, management and outcomes more generally, has been done. Among the four marine reserves discussed in this report, only the Big Creek Ecological Reserve has been the focus of directed social science research. Basic questions about changes in use patterns, perceptions, attitudes and beliefs, economic values, costs and benefits, as well as social and economic relationships and institutions, can be used to guide social science research on marine reserves effectiveness. Reliable and valid answers to these questions, however, require baseline information on the human dimensions of marine reserves, monitoring of these attributes throughout the marine reserve process, and insuring their integration into management as it changes over time.

## Introduction

As interest in marine reserves as a fishery and broader ocean management tool has grown, so has the recognition of the need to consider their human, as well as their ecological, dimensions (Fiske 1992, Alder et al. 1994, Cocklin et al. 1998). The rationale for attending to these is practical as well as legal. The human dimensions include not only the potential social, cultural and economic impacts of marine reserves, but also the social, cultural, economic and political context in which they are designed, implemented and managed. Because marine reserves are established to serve social and economic, as well as ecological, goals, attention to the human dimensions throughout the process (i.e., design, implementation, management and evaluation) is necessary in order to evaluate their effectiveness in achieving these goals. Moreover, given that people interact with the marine environment within and outside marine reserves, and that marine reserve boundaries are permeable, understanding the human dimensions is also central to evaluating the ecological effectiveness of marine reserves.

To date, there has been limited social scientific study of the human dimensions - the so-called “socio-economic considerations” - of marine resource management in general and marine reserves in particular. A growing literature, however, demonstrates the relevance of these considerations, and how failure to explicitly address them has led to negative outcomes in ecological and socio-economic terms (Fiske 1992; Alder 1994, 1996; White et al. 1994, 2002; Cocklin et al. 1998, Pomeroy and Beck 1999, Russ and Alcala 1999). The key findings of this work point to the critical participation and multifaceted role of the full range of stakeholders throughout the marine reserve process. Most of this work has focused on evaluating marine reserve processes to determine the social, cultural and political conditions for their successful establishment. Little social science research has been done to inform the design, implementation, management and evaluation of marine reserves. This research focuses on use patterns; perceptions, attitudes and beliefs; economic values, costs and benefits; social and economic relationships; and formal and informal governance institutions (e.g., regulations and norms, respectively).

Yet, there is a critical disconnect between social science research and policy, linked in part to the tendency for marine reserve processes to proceed more rapidly than relevant social science research (Farrow 1996, Badalamenti 2000). This type of social science research is often carried out as an afterthought, if at all, with inadequate time, personnel and financial resources and little or no provision for its integration into the process. In most cases, insufficient social scientific baseline information is generated, making it difficult to develop appropriate measures to evaluate the human dimensions, and how these interact with the ecological dimensions, to influence the effectiveness of marine reserves. This is the case for the four marine reserves addressed in this report.

The first part of this chapter provides an overview of socio-economic considerations for marine reserves. The next section presents the legal and practical rationales for addressing them in the marine reserve process. The third section focuses on the diversity of socio-economic considerations and their relevance throughout the process. The status of social science research that has been conducted on the four marine reserves that are the subject of this report is noted for each topic. The fourth section discusses the social science methods that can be used to generate scientific information, and briefly discusses measures of effectiveness for marine reserves. The last section presents a brief summary and conclusions regarding the relevance of the human dimension, and the critical need for information on this integral aspect of the marine reserve process.

### *Rationale for addressing the human dimension in the marine reserve process*

There are both practical and legal rationales for investigating, understanding and integrating the human dimensions in the marine reserve process. In practical terms, marine reserves are a management tool directed toward governing - and often changing - human behavior. To govern human behavior, it is essential to identify patterns of behavior and what drives them, determine how and why they are problematic, and develop appropriate mechanisms that can facilitate change. Also, because people respond in diverse ways to management interventions, attention to the social, cultural and economic factors that influence behavior is critical, to help insure desired outcomes and prevent unintended negative consequences (Davis and Tisdell 1995). And, because marine reserve goals and objectives are shaped by human perceptions, values and beliefs, understanding these attributes is critical to effective marine reserve design and evaluation.

A second practical argument is that the design, implementation, management and evaluation of marine reserves constitute a process that requires extensive information and other resources over time. Obtaining this information through scientific research and existing management-related data collection can be prohibitively costly and difficult. Incorporating the human dimension in this case involves making use of the diverse resources that participants can bring to the marine reserve process including their knowledge and experience, insights into potential management hurdles, and other resources (Crosby et al. 2000). Active engagement of stakeholders in the marine reserve process provides opportunities to elicit this information and address information gaps and misconceptions held by all stakeholders (i.e., managers, scientists, resource users, environmentalists, the broader public). It thereby can facilitate the development of mutual trust, shared goals and understanding. These, in turn, can enhance support for the marine reserve process and voluntary compliance with management decisions (Fiske 1992).

In legal terms, federal and state mandates require evaluation of the social and economic impacts of marine reserves. The overarching federal statutory requirement is found in the National Environmental Policy Act (NEPA), which mandates the integrated use of the social sciences in assessing impacts on the human environment through attention to "the natural and



physical environment and the relationship of people with that environment" (40 CFR 1508.14; ICGP 1994). Federal environmental impact statements are likened to "full-disclosure procedure[s] for federal decision-makers, who are then expected to consider the negative as well as the positive implications of potential courses of action, and the unintended as well as the intended consequences, before they proceed" (ICGP 1994).

Although marine reserves are often construed as a broad ocean management tool, they are explicitly or implicitly a fishery management tool, because they directly affect fishing activity. The 1976 Magnuson Fishery Conservation and Management Act (MFCMA) established legal requirements for considering the human dimensions of fishery management, especially in the national standards for fishery management [MFCMA section 301(a)]. The MFCMA requires the use of "a systematic, interdisciplinary approach which will ensure the integrated use of the natural and social sciences ... in planning and decision-making" [NEPA section 102(2)(a)] in the preparation and implementation of federal fishery management plans (FMPs). Because marine reserves are a "system for limiting access to the fishery in order to achieve optimum yield" [MFCMA section 303(b)(6)], the Secretary of Commerce and fishery management councils are required to consider their economic and social impacts. This requirement was strengthened by 1990 amendments to the MFCMA, which specified that FMPs must "assess, specify, and describe the likely effects of conservation and management measures on participants in the affected fishery, and the effects on participants in other fisheries that may be affected directly or indirectly" [MFCMA section 303(a)(9)]. The 1996 Sustainable Fisheries Act added national standard 8, which requires conservation and management measures to minimize adverse economic impacts on fishing communities, consistent with the Act's conservation requirements (NOAA 1997). Fishing communities are defined as "substantially dependent on or substantially engaged in the harvest or processing of fishery resources to meet social and economic needs" (NOAA 1997). As a result, ecological, economic and social impact assessments are necessary to meet MFCMA and NEPA requirements, based on the rationale that "the more comprehensive the information base and analysis, the more objective and defensible will be the decision-making process" (ICGP 2001).

In California, legal mandates to consider the human dimensions in the design and use of management tools such as marine reserves are found in the California Environmental Quality Act (CEQA) and the Marine Life Management Act (MLMA). In passing CEQA, the Legislature noted that "it is the policy of the state to ... require governmental agencies at all levels to consider qualitative factors as well as economic and technical factors and long-term benefits and costs, in addition to short-term benefits and costs and to consider alternatives to proposed actions affecting the environment" [CEQA § 21001 (g)]. Following the MLMA, California FMPs are required "to summarize information on economic and social factors in the fishery" [7080(e)]. If an FMP includes new management measures, it must analyze their anticipated effects on fishermen as well as coastal communities and businesses that rely on the fishery [7083(b)]. The MLMA directs FMP developers and managers to minimize adverse impacts on small-scale

fisheries, coastal communities, and local economies [7056(j)] (although these concerns are secondary to the broader mission of fostering sustainable fisheries), and to allocate increases or restrictions on catches fairly among recreational and commercial fishermen [7072(c)] (Weber and Haneman 2001).

### *Socio-economic considerations for the marine reserve process*

The features of the human environment that are relevant to marine reserves include use patterns; perceptions, attitudes and beliefs; costs and benefits; social and economic relationships; and formal and informal institutions related to marine resources, their use and management. Understanding these as they interact with the biophysical environment is critical to all stages of the marine reserve process, and to the achievement of marine reserve goals and objectives.

### *Use patterns*

Use patterns comprise the spatial and temporal aspects of human activities in the marine environment. These activities include both consumptive (e.g., fishing, kelp harvesting, oil and gas drilling) and non-consumptive (e.g., diving, whale-watching, shipping) activities. Use patterns vary considerably within and among groups, across locations and over time, and are influenced by a range of environmental, economic, socio-cultural and regulatory factors. Research on use patterns associated with the four Central California marine reserves addressed in this report has been done only for Big Creek Ecological Reserve (Pomeroy in press, Wilcox and Pomeroy in press).

Understanding use patterns and the forces that underlie them is critical to the effective design, management and evaluation of marine reserves (Walters 2000). Information on use patterns can be used to identify potential marine reserves sites. For example, if a marine reserve is to be used to protect a particular habitat type, information on use patterns can inform the selection of sites that seek to minimize and equitably distribute the displacement of users while maximizing habitat protection. Consideration of use patterns, and the factors that influence them, is also important for anticipating potential changes in use patterns such as displacement of consumptive users following the establishment of marine reserves. Such shifts in activity can result in negative and perhaps unintended ecological, social and economic consequences such as localized habitat damage and overfishing, conflict over access to open sites, increased financial and safety risks as fishermen travel to more distant or hazardous areas, and undesired changes in the distribution of costs and benefits of marine resource management.

Information on use patterns before and after marine reserve implementation is critical for evaluating the effectiveness of marine reserves, in both ecological and socio-economic terms. The permeability of marine reserve boundaries means that activities, conditions and outcomes within a reserve (and changes therein) will influence those outside its boundaries, and vice versa

(Pomeroy 1999, Walters et al. 2000). For example, fishermen may aggregate at the edge of a marine reserve, enabling them to take advantage of potential spillover, but precluding the broader distribution or recruitment of fish from the reserve to the areas beyond the reserve edge, as McClanahan and Kaunda-Arara (1996) documented in Kenya. They argue that fishermen experienced high catches at the boundary of the marine reserve, but that the concentration of fishing effort likely prevented the broader distribution of the spillover from the reserve into the larger surrounding area. These ecological impacts also have socio-economic implications through the differential distribution of costs and benefits among resource users who fish the reserve edge and those who do not. Shifts in use patterns can have broader implications for marine reserve effectiveness if they result in excessive concentration of fishing in areas that remain open. The interaction between this human dimension and the ecological dimensions of marine reserves is evident in the concern that locating a reserve in a sink area may increase fishing pressure on source populations and thereby prompt or exacerbate population decline (Crowder et al. 2000 in NRC 2001).

Knowledge of use patterns and how they change is also critical for evaluating the effectiveness of marine reserves in socio-economic terms. Most marine reserves will displace fishing effort, which in turn will have social and economic impacts on resource users and related communities. Fishermen may find adequate substitute sites to replace those lost to marine reserves, although their operating, safety and social psychological costs may increase as a result of having to transit further to and from these sites. Because the quality of fishing sites varies in time and space, not all areas that remain open will afford commensurate fishing opportunities (Kenchington 1995, Hilborn 2001). Moreover, quality fishing areas that remain open may become crowded, and result in social conflict and other increased costs of fishing. In the Central California context, legal mandates require the evaluation of these costs and benefits and their distribution, and the design of reserves that seek to allocate these equitably.

Non-consumptive use patterns also warrant attention. Understanding non-consumptive use patterns prior to marine reserve establishment can inform site selection and design, and provide a baseline against which to evaluate marine reserve outcomes in terms of desired recreation, tourism and educational benefits. Monitoring shifts in these use patterns can facilitate detection of problems such as crowding, which can diminish the social and economic benefits to these users and result in ecological damage (Davis and Tisdell 1995, Boersma and Parrish 1999, Crosby et al. 2000, Dobrzynski and Nicholson 2001).

Changes in use patterns on the water can have shoreside repercussions as well. Marine reserves, to the extent they limit or displace fishing, may result in losses of fishery-related activity for ice and bait providers, harbors (which depend on commercial activities to qualify for federal dredging funds), and other providers of local goods and services. Losses of local fresh fish supply can also negatively affect local restaurants, markets and consumers who may in turn become dependent on external supplies of seafood. These losses may be offset - or more than

compensated for - by increases in non-consumptive activities. Alternatively, they may be exacerbated if these activities are already near or at their ecological or socio-economic carrying capacity (Davis and Tisdell 1995). A second issue is the extent to which increases in non-consumptive uses can be supported by existing or latent infrastructure and providers of goods and services. Rapid increases in the demand for such infrastructure, goods and services may provide a welcome stimulus to coastal communities, or they may overtax them. Reserve-induced shifts in fishing effort to other regions may generate demand for and stimulate - or overburden - coastal communities in those areas. Across sites, these shifts also can lead to the redefinition of community identity, for example, from fishing towns to recreation centers, or from non-fishing towns to fish producing centers.

### *Perceptions, attitudes and beliefs*

Perceptions, attitudes and beliefs are components of individuals' views of the world that shape their preferences, choices and actions. In the context of marine reserves, perceptions, attitudes and beliefs of particular interest include those related to the marine environment, its management and use, as well as marine reserves per se (Fiske 1992, Crosby et al. 2000). Among the four Central California reserves, directed research on perceptions, attitudes and beliefs has only been done with regard to Big Creek Ecological Reserve (Pomeroy and Beck 1998, Pomeroy in press).

Shared perceptions of a problem (or opportunity) in the marine environment, positive attitudes toward management, and beliefs that marine reserves are an appropriate management tool can translate into support for them. A process that fully engages stakeholders can engender these, and a sense of ownership in the process and its outcomes (Crosby et al. 2000). Such support facilitates, and reduces the costs of, design and implementation (Alder 2002). Perceptions, attitudes and beliefs that differ may result in opposition to marine reserves that can be costly (Fiske 1992, White et al. 1994, Crosby et al. 2000). At the same time, divergent opinion and action can catalyze more careful planning and action to address these differences, resulting in a more robust outcome in ecological, social and economic terms.

It is also critical to consider the diversity of expectations for marine reserves. Unrealistically high expectations of marine reserve benefits can lead to dissatisfaction with marine reserve outcomes, and unfounded rejection of marine reserves as a management tool. With knowledge of the potential mismatch between expectations and likely outcomes, efforts can be made to inform people and help make these expectations more realistic (Wolfenden et al. 1994). Acute concerns about negative social and economic impacts and their distribution also require consideration (Suman et al. 1999). Failure to address these can result in unrealistic expectations, opposition and reduced compliance to marine reserves once they are established (Fiske 1992, Alder 1996, Suman 1997, Suman et al. 1999).

### *Economic values, costs and benefits*

Economics of marine reserves pertains to market and non-market values, benefits and costs, and positive and negative impacts of marine reserves, which allocate access to marine places and resources. Although marine reserves are theorized to generate substantial social and economic benefits (Hannesson 1998, Sanchirico and Wilen 1999), limited attention has been directed toward their systematic, empirical evaluation to date (Badalamenti 2000, Alder 2002). Where such evaluations have been done, they have focused primarily on net economic impacts (e.g., Dixon et al. 1993), have been limited by a lack of adequate baseline information (e.g., McClanahan and Kaunda-Arara 1996, Roberts et al. 2001), or have not adequately addressed interactions with factors such as other (non-reserve) regulations (NOAA et al. 2002). A notable exception is the research on Apo and Sumilon Island marine reserves in the Philippines (Alcala and Russ 1990; White et al. 1994, 2002; Russ and Alcala 1999). Study of the economics of the four Central California marine reserves has been limited to work done at Big Creek (Pomeroy in press), and that directed more toward the larger fishery rather than the Big Creek reserve per se.

Part of the challenge in assessing costs and benefits follows from the difficulty of assigning market values to the intangible aspects of the marine environment. Economic costs, benefits and impacts such as landings of fish or increases in non-consumptive diving may be readily measured and assigned a market value. Other qualities such as the aesthetic or social psychological value of fishing for sport, subsistence or income, or existence value (knowing that an area in the marine environment is protected from direct human disturbance) are not readily assigned a market value. Moreover, it is difficult to tease out the economic effects of marine reserves from those of other forces in the marine environment and the interactions among them.

A separate and seldom considered economic dimension of marine reserves is the cost of the marine reserve process, from design through evaluation. These costs include time and funding to support agency and other stakeholder participation, the collection, analysis and reporting of quality biophysical and social scientific information, and the integration of scientific and local knowledge throughout the process. These costs are not trivial, especially given agencies' limited staff, considerable workload, and often scant financial resources, and the fact that other stakeholders often participate without compensation for the time, travel costs, foregone income and the considerable knowledge and social resources that they contribute.

Understanding the costs and benefits of marine reserves, and the distribution of these impacts, is essential to reserve design. Together with data on use patterns, this information can be used to design marine reserve options that seek to minimize negative socio-economic impacts while maximizing ecological and other socio-economic benefits. This approach was used by a group of commercial fishermen involved in the Channel Islands Marine Reserves Working Group (MRWG) process. In March 2001, several fishermen, each representing a different fishery or gear type, held an informal meeting where they drafted a marine reserve scenario to present to

the MRWG. They drew upon their own ecological and social knowledge and information generated during the MRWG process, and were guided by ecological and socio-economic goals and objectives developed by the MRWG. The result of their efforts was a proposed network of marine reserves that addressed several (but not all) of the MRWG's ecological goals while seeking to minimize costs to and distribute them equitably among commercial fishery sectors. Although their proposal was not formally considered by the MRWG, a modified version is now before the Fish and Game Commission as one of six marine reserve alternatives.

### *Social and economic relationships*

Social and economic relationships consist of the linkages among those who use and otherwise value the marine environment and associated shoreside socio-economic systems. Such linkages are evident, for example, in the social, cultural and economic ties between (commercial, recreational and subsistence) fishermen on the one hand, and receivers, processors, harbors, support businesses and their communities on the other. Other users of the marine environment likewise have extensive shoreside linkages. These linkages have been documented for the fishery associated with the Big Creek reserve (Pomeroy and Beck 1998, Pomeroy in press), but not the other three Central California reserves.

These linkages convey information that can be used to inform, support or oppose the marine reserve process. While not all community members may be willing or able to participate in the marine reserve process<sup>1</sup>, well positioned representatives can convey information between members of their social and economic networks and other participants in the process. In addition, the social and economic forces manifest in these relationships influence attitudes, perceptions, beliefs, use patterns and other behaviors related to marine reserves. Attention to these relationships (along with other socio-economic considerations) contributed to the successful and locally supported designation of the Fagatele Bay National Marine Sanctuary in Samoa (Fiske 1992) and Apo Island Reserve in the Philippines (White et al. 1994). The neglect of these relationships contributed to the failure to establish a National Marine Sanctuary at La Parguera, Puerto Rico (Fiske 1992), to the failure of a marine managed area in St. Lucia (Sandersen and Koester 2000), and to the demise (at least temporarily) of the reserve at Sumilon Island, Philippines (Alcala and Russ 1990; White et al. 1994, 2002; Russ and Alcala 1999).

These relationships also constitute the network through which the positive and negative impacts of marine reserves circulate. Inattention to these linkages can result in lost opportunities to realize broad benefits, or in greater than expected costs as negative impacts reverberate among individuals and communities.

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<sup>1</sup>In all except the most localized situations, it is impractical for all of those with an interest in the marine reserve process to participate in all aspects of it.

### *Formal and informal governance institutions*

Institutions are the shared norms, rules and strategies that manifest the social and economic relationships discussed in the previous section and govern individual and collective behavior (Ostrom 1990). Formal institutions include the structures of government (e.g., legislatures, agencies) and associated rules and regulations (e.g., NEPA, MFCMA). The jurisdictions, mandates and actions of these institutions vary, overlap and at times conflict with one another. Informal governance institutions include locally devised rules and shared understandings that govern behavior complementary to, in the absence of, or in spite of formal government. Formal and informal institutions may be complementary, compatible or contradictory, but inevitably interact with one another. In the context of marine reserves, these interactions may facilitate and support, or hinder and undermine the effectiveness of marine reserves (Fiske 1992, Johannes 1998, Pomeroy and Beck 1999, Pomeroy 1999). For example, the cooperative arrangement at Big Creek Reserve, California, where the manager of the (terrestrial) Landels-Hill Big Creek (LHBC) Reserve and local fishermen established an informal marine reserve, is an example of an informal, local institution (Pomeroy and Beck 1999, Pomeroy 1999). It was formalized as a state Marine Ecological Reserve in 1994, and is co-managed by the LHBC reserve manager (through UC Santa Cruz) and the state Department of Fish and Game (DFG). A memorandum of understanding between UC Santa Cruz and DFG provides for the LHBC reserve manager's oversight of day-to-day operations, and thereby recognizes the cooperative arrangement between the fishermen and the reserve manager (Pomeroy and Beck 1999). The institutional arrangements at the other three Central California reserves have not been studied.

Informal institutions can serve as mechanisms for support and management of marine reserves, although not all localities have the capacity to fulfill these functions (Johannes 1998, King and Faasili 1999). On the other hand, efforts to establish marine reserves may be perceived as a threat (or contradictory) to local institutions, and lead to resistance or efforts to undermine them (Fiske 1992, White et al. 1994, Alcala and Russ 2000). Knowledge and understanding of the formal and informal institutions that govern resource management and use and how they do or might interact with one another are important and often neglected components of the marine reserve process. In the U.S., for example, different federal and state agencies have authority over different places and activities in the marine environment. Legally and practically, efforts to establish marine reserves must work within these institutional arrangements. Failure to do so can result in incompatibilities between marine reserve design and existing management that undermine the effectiveness of both. Rules that govern the use of marine resources, whether in the form of marine reserves or traditional fishery management, may interact with one another to limit the extent to which those affected can adapt (Pomeroy and Hunter 2001, Pomeroy in press). Coordination of marine reserves with other management measures is important in order to guard against excessive effort and user conflicts in areas that remain open (NRC 2002), and to help insure that activities within marine reserves are conducted in a way that is consistent with their goals and objectives.

### *Approaches and methods for incorporating the human dimension*

Two general approaches are useful for incorporating socio-economic considerations into the marine reserve process: public participation and social scientific research. These two mechanisms differ in their utility and their limitations, but can complement one another.

Forms of public participation vary in terms of the nature and extent of communication between government and the larger public. Common mechanisms used in marine resource management include public hearings and the solicitation of written comments to obtain input and feedback on proposed management actions. These methods are consistent with centralized, top-down systems in which government retains full authority and responsibility for management, informing the public about its decisions once they have been made. However, several factors have led to the growing interest in more cooperative forms of governance, and the use, for example, of small group meetings and workshops in which government consults with non-government stakeholders, giving the latter a greater say in management decisions, with more positive ecological and socio-economic outcomes (Berkes 1989, Pinkerton 1989, White et al. 1994, Johannes 1998). These forms of public participation can be effective for eliciting perceptions, values, beliefs, local ecological knowledge, opinions and other information from a broad range of stakeholders throughout the marine reserve process. They are also valued because they can enhance communication, generate mutual understanding and trust, and provide social and other resources to facilitate and enhance marine reserve design, management and evaluation.

However, public participation alone is not sufficient for addressing socio-economic considerations, nor is it a substitute for social scientific research for marine reserves (ICGP 1994, Cocklin et al. 1998). Public participation does not afford full representation of all relevant stakeholders, nor does it provide systematic, reliable and valid information on the full range of human dimensions topics discussed above. Social science research can address these limitations, and provide critical information on socio-economic considerations for marine reserves.

Several social science approaches and methods can be used to inform the marine reserve process. Archival research entails the systematic review and analysis of previous studies, landings and other quantitative data, and other historical materials. It is useful for assessing trends (e.g., in use patterns) and developing an understanding of the environmental, socio-economic and regulatory context (e.g., relationships and institutions). Archival research alone, however, usually cannot provide a complete understanding of these dimensions, because the data were not generated with the marine reserve process in mind, and quickly become outdated. For example, in the Dry Tortugas, Florida and the California Channel Islands marine reserve processes, existing landings data (which illustrate the distribution of fishing effort) had been collected at too gross a scale to enable meaningful evaluation of marine reserve alternatives. Moreover, the data were insufficient to inform assessment of potential interactions between local



fisheries and marine reserves. In the Channel Islands case, a recent study of the social and economic organization of the California market squid fishery (Pomeroy and FitzSimmons 2001) afforded considerable information on general use patterns, social and economic relationships, attitudes, perceptions and beliefs, informal and formal institutions and the economics of the fishery. This work had also generated social resources (e.g., fishing industry participants' and others' good will, active support and constructive and timely input into the research effort) essential to designing and conducting research to inform the MRWG process, even with inadequate time and funding. Similarly, archival information was insufficient to efforts to understand the Big Creek reserve process and the reserve's performance. Directed field research was needed to address these critical information needs.

An important complement to archival research, field research entails the systematic collection and analysis of primary data using methods and tools such as surveys, ethnography (i.e., in-depth interviews and observation) and focus groups. Each of these has its strengths and weaknesses. Mail, phone and in-person surveys, which usually employ statistical sampling and a highly structured set of questions to enable analytical generalization, are useful for collecting quantitative data from large numbers of people (Yin 1989, Babbie 1998). While surveys are viewed as being more cost-effective than other field data collection methods, they tend to produce less valid and reliable information (Babbie 1998). Ethnography entails the use of in-depth interviews and observation (Spradley 1980) to gain in-depth understanding of, in this case marine reserves, in local context. Ethnographic research tends to produce more reliable and valid results than survey research, but requires considerable time and effort (Spradley 1980). Focus groups and workshops, which bring individuals together to discuss a well defined set of topics, can generate information on shared understandings about issues and ideas for resolving them. They are vulnerable, however, to the effects of interpersonal and group dynamics (Babbie 1998), as when more vocal or powerful members of the group dominate the discussion and thereby suppress input from others.

The results of social science data analyses are reported in a variety of ways, and may be descriptive, explanatory or predictive of individual and collective processes and potential behaviors. In the marine reserve process, case studies can be done to describe and explain current (social, economic, political) situations and trends. Social and economic impact assessments that build upon such contextual understanding can be used to predict and compare marine reserve scenarios' potential and actual outcomes. These outcomes are expressed in terms of 1) absolute or relative benefits and costs, in social, economic and ecological terms, 2) how behavior and associated impacts change with reserve implementation, and 3) how these are distributed within and among human and ecological systems. Cost-benefit analyses can be used to weigh the costs and benefits of marine reserves prior to and following implementation. Cost-effectiveness analyses can be used to select the least costly option to achieve a given set of objectives (Crosby et al. 2000).

Social science research to address marine reserve information needs requires adequate time, funding and social resources. The researcher uses these to build working relationships with study participants, develop research tools, carry out quality data collection and analysis, and prepare results that are responsive to management and public needs and concerns. In addition, the collection, analysis and reporting of human dimensions data to inform marine reserve processes raise social and ethical issues that must be addressed. Information on use patterns, economics and other aspects of the human environment is personal, potentially proprietary and sensitive. Social scientists are obligated to operate by principles of ethical research, whereby they insure that participation is voluntary and anonymous, and individuals' information remains confidential. However, access to and use of this aggregated information for the marine reserve process raise social and ethical issues that need to be explicitly addressed in the public arena.

*Measures of effectiveness for marine reserves: The human dimensions*

Just as it is vital to evaluate marine reserves for their ecological effectiveness, it is also critical that they be evaluated for their socio-economic effectiveness. Effectiveness may pertain to the marine reserve process itself (i.e., the means), or its outcome (i.e., the ends). Goals and objectives of marine reserves in most cases include some combination of maintaining ecosystem functions and conditions, maintaining (or increasing) resource abundance and diversity, promoting sustainability (in ecological, social and economic terms), and providing opportunities for (non-consumptive) recreation, tourism, education and scientific research.

Criteria for evaluation - or measures of effectiveness - should be simple, measurable, cost-effective, and reflective of these goals and objectives (Alder et al. 2002). They should also be clearly defined and understood in common by all participants. Alder et al. (2002) adapted RAPFISH (Pitcher et al. 1998), a fishery management evaluation model that uses multidimensional scaling, to develop a marine protected area (MPA) evaluation model (MPAEM), and pilot tested it at 20 MPA sites. To apply the model, they identified and defined attributes associated with MPAs such as maintenance of habitat, biodiversity and resources, sustainable exploitation, economic benefits, and social features (e.g., equity, stewardship, management resilience, efficiency). They then had marine reserve managers and scientists knowledgeable of the test cases evaluate the MPAs on each attribute using an ordinal score to indicate whether that attribute had improved, declined or remained unchanged since reserve establishment. The ordinal scores on each of several attributes were then arrayed for comparison, and combined to provide a qualitative evaluation of marine reserve effectiveness. Although Alder et al. (2002) report that this pilot test demonstrated the potential utility of the model, they caution that it still needs to be evaluated by other stakeholders involved in those MPA processes. The authors note, however, that its utility for stakeholders other than managers and scientists has yet to be demonstrated.

The foregoing discussion of the socio-economic considerations for marine reserves suggests several attributes that could be integrated with Alder et al.'s MPAEM to evaluate marine reserve effectiveness from a human dimensions perspective. It is possible to measure marine reserve effectiveness by asking stakeholders and expert observers how they would rate the reserve(s) on each attribute after marine reserves have been in place for some time. Although perceptions of effectiveness are critical, a more valid and robust evaluation would also entail the development of baseline human dimensions data well before marine reserve implementation, and continued monitoring and consideration throughout the marine reserve process.

MPAEM constitutes one approach to marine reserve evaluation, and can stimulate more directed attention to the need for tools and measures to evaluate MPA processes and outcomes from a human dimensions perspective. There is a critical need for measures of effectiveness that explicitly address both qualitative and quantitative phenomena and the interactions between them, as well as the interactions between marine reserves' effectiveness in achieving ecological and socio-economic goals. In addition, these measures need to address marine reserve processes and outcomes as they interact with and compare to other marine resource management strategies.

## **Summary and Conclusion**

Commonly viewed as a management tool, marine reserves are also a management process that requires careful consideration of a range of socio-economic, or more accurately, human dimensions factors. Among these are how people and communities perceive, value and use the marine environment, and the informal and formal rules and structures that influence these values and uses. The rationale for attending to these is practical as well as legal. Marine reserves generate social and economic impacts which both individuals and society must bear. Moreover, human responses and adaptations to marine reserves have ecological as well as socio-economic implications. Attention to the human dimensions can facilitate adequate planning to take advantage of the opportunities and mitigate the challenges that marine reserves pose. NEPA and other federal and state statutes provide the legal foundations and some guidance for addressing the human dimensions for marine reserves in Central California.

Public participation and social science research are complementary means for addressing social and economic considerations for marine reserves. Public participation is especially useful for addressing social and economic concerns associated with marine reserves, and generating trust, support and other social resources to assist in their design, management and evaluation. It is not, however, a substitute for systematic collection and analysis of socio-economic information to inform the process. The social sciences offer diverse methods that can be adapted to particular stages in the process and associated information needs. Yet to date, limited social science research has been done, with the majority of this work focused on evaluating marine reserve processes to identify the necessary and sufficient conditions for the establishment of

marine reserves. Little research to inform and evaluate marine reserve design, implementation, management and outcomes from a human dimensions perspective has been done. This is especially the case in Central California, where directed social science research has only been done on the Big Creek Ecological Reserve, and that hindered by a lack of adequate baseline information.

Measures of effectiveness relevant to the human dimensions of marine reserves are not well developed, even as the theory of marine reserves posits that social and economic benefits will outweigh their costs. Basic questions about changes in use patterns, perceptions, attitudes and beliefs, economic values, costs and benefits; social and economic relationships and institutions can be used as the basis for developing such measures of effectiveness. Reliable and valid answers to these questions, however, require baseline information on the human dimensions of marine reserves, monitoring of these attributes throughout the marine reserve process, and insuring their integration into management as it changes over time.

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## SUMMARY

The exceedingly small size of existing marine reserves in Central California prevents them from achieving many of the goals and benefits attributed to marine reserves in the scientific literature. The number of fish and invertebrates inhabiting existing reserves is small, compared to the total population sizes of species in Central California. Existing reserves in Central California protect a variety of shallow water habitats and species, but do not provide reserve benefits for animals living in deeper water, unless they reside in existing reserves during a portion of their life. The older marine reserves in Central California show some of the primary benefits associated with protection from exploitation, including modest increases in size and abundance of fishes, but it is difficult to assess the degree to which these benefits represent pristine conditions. This is to be expected, as the primary fish species inhabiting these reserves (rockfishes) are slow growing and exhibit sporadic recruitment. Also, new scientific theories suggest that substantially altered habitats may or may not return to pre-existing states after the disturbance has been removed.

Marine reserves in other temperate and tropical oceans, and theoretical models of marine reserves, show substantial conservation and some potential fishery benefits. For these reasons, we expect marine reserves created in Central California for conservation purposes would accrue many of the benefits predicted by reserve theory. The extent to which reserves in Central California would successfully benefit fisheries, however, would depend on a large number of social and biological factors, such as social acceptance of reserves, fishery effort shifts, catch regulations, enforcement levels, the proportion of a stock protected in a reserve, rates of movement and larval production of protected species, and reserve size and location. Currently, only a small proportion of fished species are protected in reserves. To be an effective fishery management tool, more area would need to be placed in reserve status, but not so much as to preclude viable fisheries. If marine reserves are to be developed and successfully used in Central California as a tool for fisheries management, however, they will need to be integrated into existing fishery management processes. A structured and well-supported monitoring program, which clearly identifies a set of effectiveness parameters, will also need to be established to measure how well reserves achieve stated objectives.

Effective natural resource management requires public participation and buy-in to management goals, objectives, and regulations. Thus, just as it is vital to evaluate marine reserves for their ecological effectiveness, it is also critical that they be evaluated for their socio-economic values. In this respect, the use of marine reserves is a public policy decision that must be made with consideration of human activities. For marine reserves to be an effective public policy tool in Central California, human use patterns, perceptions, attitudes, and beliefs will need to be incorporated into the design process. Information about social and economic costs and benefits should also be incorporated to maximize the effectiveness of a reserve system. Ultimately, an understanding of how people interact with the biophysical environment is integral to the design and development of marine reserve goals and objectives.

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## APPENDIX 1

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