Monitoring and Assessment of Marine Reserves at the Northern Santa Barbara Channel Islands: A Multi-Species, Collaborative Trapping Program

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Executive Summary

This report presents the results of collaborative marine protected area (MPA) monitoring and fisheries research that was conducted from 2007-2009 at the Santa Barbara Channel Islands (SBCI). Focal species in this study are the California spiny lobster (*Panulirus interruptus*) and a suite of fishes that comprise the California nearshore fishery (cabezon, *Scorpaenichthys marmoratus*; grass rockfish, *Sebastes rastrelliger*; and California sheephead, *Semicossyphus pulcher*). We worked with California Department of Fish and Game (CDFG) resource managers, senior NOAA fisheries modelers, and local commercial fishermen to plan, design, and execute a progressive research program with three broad objectives:

(1) Expand stakeholder participation in fisheries research and develop functional partnerships among Santa Barbara area fishermen, UCSB scientists, and regional resource managers.
(2) Assess SBCI MPA performance for conservation and fisheries objectives.
(3) Gather management relevant data (life-history data and essential fisheries information - EFI) for use in traditional and alternative (i.e., MPA-based) stock assessment models.

We have successfully achieved each of these objectives, as evidenced by the products generated during our program.

Important products that demonstrate the success of our first objective, *expanded collaboration and partnerships*, are the partnerships that underlie two pending collaborative proposals that our group has submitted. The first builds upon lobster work supported by the current award, and is a joint proposal with CDFG (contact, Dr. Douglas Neilson) and the California Lobster and Trap Fishermen's Association (CLTFA). The purpose of the proposal is to expand collaborative data collection and MPA monitoring statewide in an effort to gain EFI cited as deficient in an MSC pre-assessment for this fishery. The second proposal expands work with the nearshore fishery to conduct an experimental fishery (under a federal exemption) with a high level of fishery coordination. These two proposals indicate that our work to date does not end with the current grant: we have built important bridges towards future partnerships. The degree of fishery collaboration in this project is evidenced by the high number of fishermen that participated in research, as 15 and 11 members of the lobster and nearshore fisheries contributed to some phase of research, respectively.

Objective number two, *SBCI MPA assessment*, continues to be the subject of numerous professional presentations and peer-review manuscripts (page 84, this report). Data collected during our program show that, relative to populations in areas open to fishing, lobster and fish populations in Channel Islands MPAs were characterized by greater mean sizes, trap yields, and/or abundance. Additionally, spatially detailed studies of lobster population structure and movement patterns suggest that reefs immediately inside MPA borders likely experience net export of adult animals (*spillover*) to fished areas. However, spillover was not detectable in research traps or commercial fishery performance outside MPAs. Monitoring results are discussed in detail in this report.

We structured our MPA sampling to allow for collection of data that address our third objective, *management relevant data and models*. Specifically, we have gathered

and are processing data that provide detailed estimates of growth rates, movement patterns, and spatially explicit population structure. In the case of spiny lobster, some data have already been requested for use in a formal stock assessment currently being prepared by CDFG biologist Dr. Douglas Neilson. We maintain open communication with CDFG and Dr. Neilson, such that our work can benefit the assessment process. Additionally, growth and population data for both lobster and nearshore fish are being used to construct MPA-based models that estimate natural mortality and fishery harvest rates. Similarly, nearshore fish data will be used in a model already developed by Wilson et al. (2010) to provide a spatially explicit and low cost MPA-based assessment framework.

Our three broad objectives were supported by defined research goals that were specific to lobster and nearshore fish MPA monitoring and data collection. Specific research activities are detailed for both taxa in this report. The results presented here represent, to some degree, an overview of work accomplished during the study period, and we are actively processing data and preparing manuscripts from this work.

General Introduction

Marine fisheries and ecosystems have been the focus of recent high-profile reports that emphasize management failure and resource collapse (Jackson et al. 2001, Pauly et al. 2002, 2003, Pew 2003, Myers and Worm 2003, Worm et al. 2006). The consequences of failure not only impact human systems through loss of jobs and cultural heritage (e.g., Hilborn and Walters 1992, Hamilton and Otterstad 1998, Milich 1999), but can also impact biodiversity through species depletion and degradation of entire marine ecosystems (Pauly et al. 1998, Jackson et al. 2001, Lotze et al. 2006). Although the scale of this problem and the accuracy of some reports is subject to debate (e.g., Walters 2003, Hampton et al. 2005, Hilborn 2006, Sibert et al. 2006, Murawski et al. 2007), general trends of decline have led to an urgency for management strategies that sustain fisheries and protect biodiversity (Lubchenco et al. 2003, Pew 2003, FAO 2007). Among strategies designed to protect fisheries and biodiversity, marine protected areas (MPAs) that prohibit consumptive activities are perhaps the most globally common and expanding (UNEP-WCMC 2008).

Empirical studies indicate that MPAs are generally effective conservation tools that increase the abundance and mean size of target organisms within their borders (reviews by Côté et al. 2001, Halpern 2003, Lester et al. 2009). These population increases lead to potential fisheries benefits through export of larvae and/or emigration of adults, as predicted by a well formed body of modeling literature (Polacheck 1990, DiMartini 1993, Hilborn et al. 2006, Botsford et al. 2009). Empirical tests of these theoretical predictions are rare and often controversial (e.g., Hilborn 2006), but important case studies are starting to emerge (see Abesamis et al. 2006, Pelc et al. 2009, Goni et al. 2010).

In an effort to realize the potential benefits of marine protected areas for conservation and fisheries, the state of California in 2003 designated a network of 10 no-take MPAs and 2 marine conservation areas in state waters (intertidal to 3 miles offshore) at the northern Santa Barbara Channel Islands (Fig. 1). These MPAs encompass 21% of state waters surrounding the Channel Islands, while the other 79% remains open to commercial and recreational fishing (CDFG et al. 2008). This island chain straddles a major oceanographic and biogeographical transition zone through which the California current mixes with warmer southern waters.

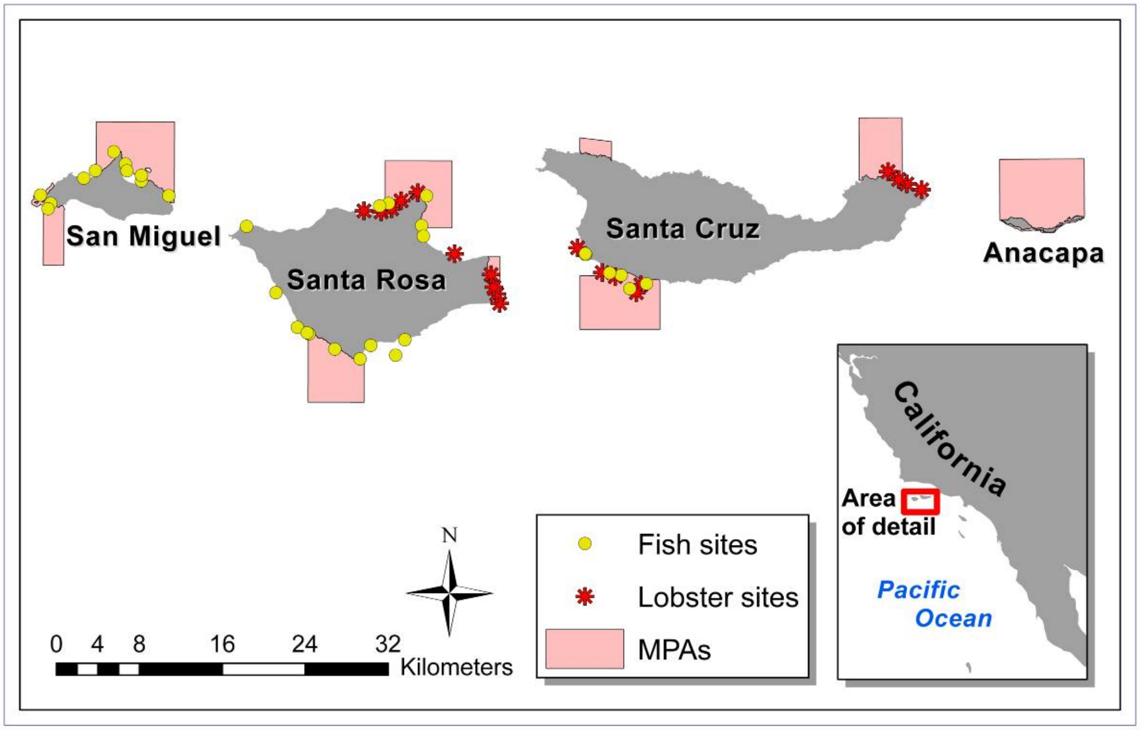


Figure 1. The four northern Santa Barbara Channel Islands, and associated marine protected areas at which lobster and nearshore fish research was conducted (Santa Barbara Island and MPA not shown).

Implementation of the SBCI MPA network was accompanied by a set of well-defined performance objectives, among which was a commitment to monitoring the biological changes of target populations inside and outside reserves. The monitoring plan for tracking biological changes was initially based almost entirely upon visual SCUBA transect surveys conducted by CDFG, National Park Service (NPS), and the Partnership for Interdisciplinary Study of Coastal Oceans (PISCO). Although visual surveys are essential and effective monitoring tools, a monitoring program that is based entirely on this approach suffers two important limitations: (1) stakeholders such as commercial or recreational fishermen are not likely to be involved in scientific SCUBA work, and; (2) visual surveys may be ineffective for cryptic, nocturnal, or deepwater species. As a remedy, the OPC allocated funds for collaborative fixed-gear surveys that are basis of the work reported here.

Collaborative and *cooperative* fisheries research (CFR) are increasingly popular strategies for collecting fishery relevant data, and include fishermen in all, or some, phases of research, respectively (NRC 2004). CFR is an effective way to increase the quality and quantity of data collected for management and policy assessments (NRC 2004, Karp et al. 2001), and the scientific benefits of CFR are complemented by demonstrated social benefits (McCay and Jentoft 1996, Conway and Pomeroy 2006, Hartley and Robertson 2009). Consequently, there are widespread calls for expanded CFR (Ocean Studies Board 2000, Pew 2003, US Commission on Ocean Policy 2004).

The ability of CFR to improve data collection is due to three mechanisms. First, fishermen spend more time at-sea than scientists, and have different opportunities to observe and understand the biological and physical processes that influence fishery performance. This understanding is commonly referred to as fisher knowledge or local ecological knowledge (LEK; Hartley and Robertson 2009). When scientists incorporate LEK they can refine hypothesis formulation, sampling efficiency, and interpretation of results. Secondly, the physical tools and skills of commercial fishermen (in addition to LEK) allow for more efficient and cost-effective sampling. Finally, working with fishermen allows scientists to sample populations in a manner that mimics fishing and therefore produces data are comparable to fisheries-dependent datasets (e.g., logbooks, fishery-mediated data collection and observer programs).

Collaborative MPA monitoring and fisheries research conducted under this proposal was structured to harness the advantages that CFR confers to scientific sampling. Working with commercial fishermen allowed us to sample at multiple sites associated with replicate MPAs, and it helped us select and sample in habitats with similar physical attributes and historical catch characteristics. Handling large numbers of animals on fishing vessels also provided the opportunity to conduct tag-recapture studies. With regard to lobster work, the use of commercial lobster traps as sampling devices, in addition to commercial fishery logbook analyses, allowed us to perform a before-after comparison of trap yield inside and outside reserves. Similarly, we port-sampled the commercial catch of spiny lobster and nearshore fishes from areas surrounding our sampling sites. This port sampling broadened the context of our survey trapping, and would have been irrelevant had we sampled with more traditional and fishery independent techniques. Finally, we have developed collaborative partnerships that have resulted in greater community participation, and our research programs continue to expand these partnerships for improved research capacity and stakeholder involvement.

As mentioned above, work reported here was focused upon two distinct taxonomic groups: (1) California spiny lobster (*Panulirus interruptus*), and; (2) a suite of reef-associated or demersal fishes that comprise the California nearshore fishery (cabezon, *Scorpaenichthys marmoratus*; grass rockfish, *Sebastes rastrelliger*; and California sheephead, *Semicossyphus pulcher*). All research conducted under this proposal was guided by three overarching objectives: (1) expand stakeholder participation in fisheries research and develop functional partnerships among Santa Barbara area fishermen, UCSB scientists, and regional resource managers; (2) assess SBCI MPA performance for conservation and fisheries objectives, and; (3) gather management relevant data (life-history data and essential fisheries information - EFI) for use in traditional and alternative (i.e., MPA-based) stock assessment models. Despite these common objectives, the specific research goals (as stated in our original proposal to OPC) and work accomplished for spiny lobster and nearshore fishes were slightly different. Consequently, our activities and findings for lobster and nearshore fishes are reported here in two separate sections.

Spiny Lobster Research

Overview

As stated in our original proposal of May 23, 2007, field research focused on spiny lobster addressed three specific tasks: (1) Test whether the Santa Barbara Channel Island MPAs influence the population structure (abundance, size, and age class distribution) of spiny lobster within, adjacent, and outside of the protected areas; (2) Examine whether spillover of lobster from within the SMR's to adjacent fishing areas occurs and influences yield of fishermen outside of reserves, and; (3) To further develop our scientist-fishermen-manager "hybrid" form of collaborative reserve monitoring, which can be used in other fisheries throughout CA in the MLPA process.

To accomplish these tasks, the lead lobster biologist (Kay) allocated 118 field days at sites associated with four MPAs at the Santa Barbara Channel Islands (Fig. 1, Table 1). Additionally, commercial catch from Santa Cruz and/or Santa Rosa Islands was port-sampled on 46 occasions (Table 2).

Table 1. Synopsis of lobster field effort at Channel Islands MPAs

Site / Field Season	Trapping events (daily site visits)*	Traps sampled	Lobsters sampled**	Lobsters tagged	SCUBA surveys (dive days)
Gull					
2006-'07	24	1054	3225	2102	
2007-'08	28	768	4393	3349	
2008-'09	11	356	2707	1066	5
Scorpion					
2006-'07	16	340	1527	1103	
2007-'08	26	880	3120	2525	
2008-'09	8	80	697	0	6
Carrington					
2006-'07	6	105	246	230	
2007-'08	15	418	2244	1826	
2008-'09	18	711	5123	2687	5
Skunk					
2006-'07	6	129	259	249	
2007-'08	14	326	1946	1774	
Total	**118**	**5,167**	**25,487**	**16,911**	**16**

* Because two or more MPA sites were often sampled in a single day, the *total field days* does not equal the sum of visits across all MPA sites. Daily site visits include SCUBA dive days.

** Sampled = any lobster measured and tagged, measured but not tagged, or measured upon recapture on a given field day.

Table 2. *CALobster* Port-Sampling 2004-2009

Site	Season	N trips sampled	N lobsters sampled	Total landings (kg)**	Landings port-sampled(kg)	% of total landings port-sampled
Santa Cruz	2004-2005	6	943	24,117.57	724.89	**3.01**
	2005-2006	9	1910	19,285.62	1518.42	**7.87**
	2006-2007			21,284.23		
	2007-2008*	5	1127	22,331.26	896.92	**4.02**
	2008-2009*	14	3125	22,353.79	1902.89	**8.51**
Santa Rosa	2004-2005			41,976.30		
	2005-2006	11	2066	28,535.12	1696.08	**5.94**
	2006-2007			33,025.39		
	2007-2008*	10	1833	22,692.06	1527.98	**6.73**
	2008-2009*	17	4156	27,649.92	3454.56	**12.49**

* Funding period of current report

** For Santa Cruz Island, sum of DFG blocks 685, 686, 687, 708, 709, and ¾ of 710; For Santa Rosa Island, sum of DFG blocks 688, 689, 711, 712, and ¼ of 710

Detailed descriptions and analysis from this work are reported below in four sections. *Section 1* describes a collaborative trapping program and addresses specific research questions (1) and (2); *Section 2* presents habitat and lobster surveys conducted by researchers using SCUBA and directly addresses research question (1); *Section 3* incorporates the SCUBA results from *Section 2*, in addition to higher spatial resolution of trapping data, to revisit research questions (1) and (2) from *Section 1*; and *Section 4* provides a brief overview of growth and movement data that have been analyzed to date.

Although results of spiny lobster research are presented in detail in *Sections 1-4*, the following list is a summary of important findings:

Summary of Important Findings of Spiny Lobster Research

- Compared to areas outside MPAs, trap yield (number of legal lobsters per trap) was ~4-8 times higher and average size of legal lobsters was 5.79-9.52mm larger in 3 MPAs (Carington, Gull, Scorpion) at the Santa Barbara Channel Islands. At a fourth MPA (Skunk Pt, Santa Rosa Island), these differences were much smaller and not statistically significant, but this site was only sampled briefly during one season and the Skunk Pt MPA is much smaller than the other three.

- Historical (i.e., *pre-MPA*) and spatially explicit trap yields were calculated from logbooks for each of the scientific trapping sites in this study, and *pre-MPA* yield estimates at all sites are comparable to trap yields from survey sites currently outside MPAs.

- Compared to areas open to fishing, lobster populations in MPAs had larger proportions of individuals in larger size classes.

- Spillover was not detected in survey trapping outside MPAs, nor was it detected in surveys of commercial fishing effort near MPA borders.

- Spillover of lobsters from reefs immediately inside MPA borders is suggested by trends of increased trap performance at greater distances inside MPAs.

- SCUBA surveys corroborate results of trapping surveys and indicate clear reserve effect inside MPAs.

- SCUBA surveys were used to identify important habitat features that influence lobster abundance on the seafloor, most notable were caves and dens (see *Section 2* for definitions). Site specific habitat characteristics were used in regression analyses to control for their influence on trap performance inside and outside MPAs, which allowed us to more confidently quantify reserve effects.

- Out of 729 tagged lobsters that were recaptured and reported by the commercial and recreational sectors, 58.6% and 72.1% had moved less than 1km and 2km from the location of initial tagging, respectively.

- Growth and mortality estimates are in process from tag-recapture and size structure data. Initial analyses indicate that lobster harvest rates at the northern Channel Islands are lower than coastal regions and measurably less than the 90-100% annual removal rate assumed by some resource managers and marine ecologists.

Lobster Section 1 – Do Channel Islands MPAs influence population structure and contribute to trap yield through spillover?

LS1 - Introduction

This section presents results of collaborative trapping at sites associated with three MPAs during this study (Gull, Carrington, and Scorpion). Collaboration allowed us to fill common gaps associated with measuring conservation and fisheries benefits of MPAs: (1) lack of before vs. after MPA comparisons, (2) lack of replication at MPA level, and; (3) sampling with fishery-relevant metrics. We structured our sampling to address 3 hypotheses: (H_1) The number and mean size of legal (>82.5mm) lobsters captured in traps inside MPAs is greater than in traps placed outside in fished areas; (H_2) The number and mean size of legal (>82.5mm) lobsters captured in traps immediately adjacent MPAs (*Near*) was significantly different (higher or lower) than at locations further from MPA borders (*Far*), and; (H_3) Commercial fishing effort near reserve borders is higher than at more distant sites. Acceptance of H_2 and/or H_3 was defined *a priori* as evidence of spillover and acceptance was predicated upon three patterns: (1)

higher trap yield at *Near* vs. *Far* sites (accept H_2); (2) higher or lower trap yield, *and* higher effort, at *Near* vs. *Far* sites (accept H_2 and H_3), or; (3) higher effort at *Near* vs. *Far* sites and/or an effort gradient near borders within the *Near* sites (accept H_3). By testing these three hypotheses, we address the spiny lobster conservation (H_1) and fishery effects (H_2 and H_3) of Channel Island MPAs.

Spiny lobsters (Palinuridae) are fished intensively across the globe and present many cases where MPAs are part of management (Phillips et al. 2000). This global coverage provides a good model for exploring MPA effects because results of lobster-MPA studies parallel findings of research with other marine taxa. For example, detection of MPA benefits to lobster fisheries is sparse and mostly limited to a handful of tagging studies (Kelly et al. 2002, Kelly and MacDiarmid 2003, Goñi et al. 2006) or catch at MPA borders (Goñi et al. 2006, 2008). Net fisheries benefits (i.e., difference in yield with and without MPAs) are not identified in these studies and may ultimately prove extremely difficult to measure, though Goñi et al. (2010) provide a seminal example. In contrast, a large number of studies confirm that MPAs lead to conservation benefits through increased abundance and average size of lobsters inside MPA borders (Davis and Doldrill 1980, Cole et al. 1990, Hunt et al. 1991, MacDiarmid and Breen 1993, Edgar and Barrett 1997, 1999, Kelly et al. 2000, Bertelsen and Matthews 2001, Eggleston and Dahlgren 2001, Goñi et al. 2001, Lipcius et al. 2001, Davidson et al. 2002, Acosta and Robertson 2003, Cox and Hunt 2005, Iacchei et al. 2005, Parnell et al. 2005, Shears et al. 2006, Babcock et al. 2007, Follesca et al. 2008, Pande et al. 2008, Barrett et al. 2009).

However, some studies indicate that increases in MPAs are not guaranteed (MacDiarmid 1991, MacDiarmid and Breen 1993, Acosta 2001, Lipcius et al. 2001, Mari et al. 2002, Mayfield et al. 2005) and in some cases conclusions should be drawn cautiously since most studies lack replication at the reserve level (but see Edgar and Barrett 1997, 1999, Kelly et al. 2000, and Mayfield et al. 2005) and we know of only two studies in which data were taken prior to reserve implementation (Shears et al. 2006, Follesca et al. 2008).

Because lobster-MPA studies are representative of studies with other taxa, the results we present here have broad pertinence. Furthermore, we demonstrate the benefits of CFR for policy assessment by applying the strengths of CFR outlined above (LEK, superior tools, and access to fishery dependent data) to remedy common limitations in the MPA literature. For example, we sampled with commercial lobster traps and conducted logbook analysis of trap yield prior to reserve implementation such that we could perform a before-after comparison of trap yield inside and outside reserves. Additionally, working with commercial fishermen allowed us to sample multiple reserves and sample in habitats with similar physical attributes and historical catch characteristics. We also conducted port-sampling of commercial catch and we mapped commercial effort distribution to test the theoretical predictions of MPA-fishery dynamics near borders. An important CFR methodological advancement in this study was the reciprocal knowledge exchange between scientists and the fishery. This exchange eased logistical constraints associated with more typical CFR programs (Kay et al. *in press*), and it catalyzed development of a progressive fishery-based data collection program for California spiny lobster. This study provides a detailed case study of MPA effects on California spiny lobster, but also advances CFR methodology and illustrates the power of CFR for assessment of MPAs for conservation and fisheries management.

LS1 - Materials and Methods

Site Description and Selection

Individual trapping sites within and outside each MPA (see *Collaborative Trapping*) were selected in collaboration with 5 commercial fishermen with a combined average of >60 years fishing at each site prior to the 2003 MPA closures. Accessing the local ecological knowledge (LEK) of these fishermen allowed us to identify reefs with similar historical (i.e., pre-MPA) catch dynamics and physical/biological habitat characteristics. Commercial fishery LEK was essential in guiding selection of sites that were spatially distributed inside (referred to in this report as *In*), adjacent outside (*Near*), and ~ 2-4 km farther out (*Far*) relative to a single border of each MPA (Fig. 1). To accomplish this, fishermen worked with scientists to identify 2-4 reef areas inside and outside each MPA that were similar according to the following criteria: 1) historical (i.e., pre-MPA) trap yield; 2) historical population size structure; 3) depth and surrounding bathymetry; 4) physical habitat characteristics; 5) ecological characteristics; and, 6) weather exposure and oceanography.

After candidate reefs were identified through our scientist-fishery LEK collaborative process, the university partners in this research project used SCUBA to qualitatively compare the physical and biological characteristics of each reef. Pre-reserve trap yield was compared for each trapping sites as described below (*Pre-MPA trap yield and effort*). Ultimately, we selected 4 trapping sites associated with the Scorpion MPA (2 *In*, 1 *Near*, 1 *Far*), and we selected 5 (1 additional *In*) at the larger Gull and Carrington MPAs (14 total trapping sites; Fig. 1).

Collaborative Trapping

To test our hypotheses regarding trap yield inside and adjacent MPAs (see H_1 and H_2, *Introduction*), we measured the number and mean size of legal sized lobsters captured in traps placed at each of the *In*, *Near* and *Far* sites associated with each MPA (Gull, Scorpion, Carrington). We deployed ~15 replicate lobster traps at each site (duplicated *In* sites received same approximate effort as a single site; total traps = ~15 traps per site x 3 trapping sites = ~45 traps per MPA). Traps were sampled every 2-4 days for a 1-3 month period at each site from August - October 2007 and 2008.

In each trapping area, initial haphazard deployment of traps was conducted off of a commercial lobster vessel or was guided by commercial lobstermen on the fishing grounds. Trap deployment was not strictly random, but was stratified-haphazard as traps were not positioned directly on the seafloor but were deployed from the surface around designated reefs. Traps were spaced at regular distances from each other (~ 20m) to avoid non-independence of sampling units and minimize placement bias. After each trap was retrieved we recorded the depth, time, date, and GPS coordinates for each trap, as well as the total number, sex, carapace length (nearest mm using vernier calipers), injuries (e.g., missing legs or antennae), and breeding condition of all lobsters in the trap. We minimized stress to lobsters on deck by placing them in standing seawater, returning them to the ocean as quickly as possible, and shading with wet burlap sacks as lobsters were removed from traps. After lobsters had been processed, they were returned to the exact location of capture (GPS coordinates) and released by hand.

Traps employed in this study were nearly identical to those used in the fishery for *P. interruptus* at the Santa Barbara Channel Islands. Traps measured 36 x 48 x 18 inches tall, were constructed of Riverdale® 2 x 4 inch mesh wire and attached at their base to a single 36 x 48 inch rectangular frame constructed from 1inch diameter steel rod, and coated with a hydrocarbon asphalt sealant used by the fishery to prevent corrosion. The only difference between commercial traps with these dimensions and traps used in this study and was the absence of an escape port on research traps (not less than $2^3/_8$ x $11^1/_2$ inches, as mandated by law) which allows escape of sublegal lobsters. Traps were baited with ~1lb. of Pacific mackerel (*Scomber japonicus*) that was replaced in 1-L plastic bait capsules (1 per trap) after each sampling event. Each trap was connected to a 3/8" polypropylene line and surface buoy that for allowed rapid location and retrieval as in the commercial fishery.

LEK Exchange for trapping

A unique and logistically advantageous aspect of our collaborative program was the transfer of fundamentally important LEK from fishermen to the lead field biologist in this project. This LEK transfer began when the field biologist (Kay) received extensive training from a veteran lobsterman (Miller) prior to beginning this project. During this training, Kay worked as crew during commercial lobster fishing trips in and around the sites selected for study. Additionally, other lobstermen on the fishing grounds provided support during the project, such that the biological sampling was facilitated by a community-supported LEK transfer from the fishery to biologist. Consequently, after traps were initially deployed and critical safety and fishery information about each site had been communicated, the biologist possessed the skills to sample and re-deploy traps from a university-owned vessel retrofitted with a commercial grade trap hauler.

This arrangement had strong logistical advantages over repeated sampling from commercial vessels (Kay et al *in press*). Continued oversight from commercial vessels on the fishing grounds was a critical element of the collaboration. Therefore, there were four distinct but related collaborative interactions undergirding the trapping program: (1) the initial transfer of LEK from fishery to biologist; (2) training of biologist as crew on an actively fishing commercial vessel; (3) initial trap deployments from commercial vessels, and; (4) continued oversight from commercial lobstermen on the fishing grounds.

Fishery-dependent data I: Pre-MPA trap yield and effort, and addressing caveats

To ensure that study sites (*In*, *Near*, *Far*) at each MPA had comparable trap yields prior to reserve implementation, and to conduct a before-after analysis of trap yield, we calculated area-specific daily yield-per-trap prior to MPAs from analysis of commercial logbooks. Briefly, commercial fishermen are required to report daily catches in logbooks in which the number of traps and legal lobsters retained are reported near a given landmark. Thus, one of us (Guenther) calculated the daily average number of lobsters retained near landmarks coinciding with our trapping areas. Extensive fishermen interviews provided geo-referencing of areas reported in logbooks with our trapping sites.

Trap yield is a potentially confounded measure of area-specific productivity in fisheries where effort is spatially heterogeneous and causes CPUE to equilibrate across space in accord with the *equal gains* predictions of ideal free-distribution (e.g., Swain and Wade 2003). To ensure that our CPUE estimates were reliable for analysis and not confounded by spatially differential effort trends, we measured effort levels in each research trapping area prior to MPA establishment. To accomplish this, and because detailed effort records are not recorded in logbooks, we again consulted the traditional knowledge of fishery partners. Specifically, one of us (Kay) interviewed fishermen to determine the density of traps present at each trapping site for the ~5 year period immediately preceding reserve implementation. In these interviews, fishermen were provided a map of trapping areas and asked to report the average number of total traps that they recall seeing in each area during October-November (timing of commercial season time most closely corresponding to our surveys) from 1998-2002. Fishermen were readily able to provide this historical and seemingly arcane information due to the small number (<5) of individuals that fish in a given area, the familiarity among these individuals, strong inter-annual site fidelity, the spatial precision of retrieving and resetting lobster traps, and the frequency with which commercial traps are serviced (at least every 96 hours, by law). Estimates for each area were averaged from all respondents (n = 2-5).

In addition to the caveat that differential effort might homogenize trap yield and thereby render pre-MPA trap yield unreliable to our analysis, we tested the caveat that research trapping and fishery trapping might be inherently different and thereby preclude comparisons of commercial trapping before MPAs (logbooks) and research trapping after MPAs. To test this caveat we compared logbook data and research trapping that took place simultaneously at the Scorpion *Near* and *Far* sites during the 2006-07 and 2007-08 seasons. We compared each season in separate analyses to effectively 'double check' that research and logbook trap yield data were comparable.

Fishery-dependent data II: Current effort distribution and size composition of catch

To test the hypothesis that MPAs in this study effect fisheries through spillover and therefore effort outside MPAs is highest near borders (H_3, *Introduction*), we mapped the distribution of commercial effort (trap buoys) within our *Near* and *Far* research trapping sites. Effort was mapped on four dates during the 2008 fishing season: 1 October (Carrington and Gull), 1 November, and 3 December 2008 (Scorpion, Carrington, Gull) and additionally on 19 January 2009 (Gull and Scorpion). This allowed us to examine effort profiles on the scale of 10's of meters in the *Near* sites and also on larger (km) scales by comparing effort levels at the *Near* and *Far* sites.

Finally, we compared the size composition of our research trapping data to larger sample sizes that were gathered in collaboration with commercial fishing partners. During the 2007-08 and 2008-09 fishing seasons, we conducted port sampling to measured the size composition of commercial catches. During port sampling, biologists met commercial fishing partners at Santa Barbara Harbor on 19 dates (Santa Cruz Island) and 27 dates (Santa Rosa Island) and recorded the carapace length, sex, and reproductive condition of all lobsters landed.

Data analysis

The number of legal sized lobsters (> 82.5mm) captured in research traps placed at the *In*, *Near* and *Far* sites was compared with a mixed effect 3-way ANOVA in which *time* (before vs. after MPAs) and *site* (*In*, *Near*, *Far*) were fixed factors, and *MPA* (Gull, Scorpion, Carrington) was a random factor. ANOVA was followed by a Student-Newman-Keuls *post-hoc* test. As described above, logbook catch data are recorded as the average number of lobsters caught from multiple traps in a given area on a single fishing day. To ensure consistency between logbook and research data, our research trapping data were averaged across all traps at an individual site for each daily sampling event. Consequently, the standardized unit of replication in this analysis is the yield per trap averaged across all traps at a given site for a given sampling (or fishing) event. Data gathered to test the two potential caveats we identified (heterogeneous effort distribution, comparability of fishing vs. survey trapping) were analyzed with separate 1-way ANOVAs.

Mean sizes of all legal sized lobsters per trap were compared using a mixed effect 2-way ANOVA in which *site* and *MPA* were fixed and random factors, respectively. Time was excluded from this analysis because size is not recorded in logbooks and therefore size data prior to MPA establishment were not available. ANOVA was followed by a Student-Newman-Keuls *post-hoc* test. In this analysis, the unit of replication was the mean size of all lobsters in an individual trap.

Length frequency data from each site (*In*, *Near*, *Far*) were compared at each individual MPA using Kolmogorov-Smirnov tests (KS). Survey trapping data at the Gull and Scorpion sites were then compared (KS) to port sampling data from Santa Cruz Island, and survey data from Carrington sites were compared (KS) to port sampling data from Santa Rosa Island.

The location and density of buoys in the *Near* and *Far* sites, as well as the distance of each buoy from the nearest MPA border, was examined using ArcGIS 9 (ESRI 2009). Density of commercial effort in the *Near* vs. *Far* sites was compared with ANOVA. The distribution of traps within the *Near* sites was examined with linear regression, where distance from MPA border (measured at the midpoint of sequential 50m along-shore segments) was the independent variable and the dependent variable was the number of traps in each 50m segment. Buoy data were pooled from all surveys (N = 3 at Carrington and Scorpion, N = 4 at Gull) at each *Near* site and separate regressions were run for each *Near* site.

For all ANOVA analyses, data were log transformed (ln[Y+1]) to correct for heterogeneity of variance. Homogeneity of variance after transformation was confirmed with Levene's test ($P = 0.05$) or inspection of residual plots (Ramsey and Schafer 2002). Only data describing scientific vs. commercial trap yield in active fishing grounds (caveat test) failed to meet standards for parametric analysis, and in those cases we report results from Welch's ANOVAs. Significance levels in all tests was $\alpha = 0.05$. Data were analyzed with JMP 8.0 (SAS Institute, Inc 2008) and program R (R Development Core Team 2005).

LS1 – Results

H_1 and H_2: Difference in numbers and mean size of lobsters in traps at In, Near, Far sites

Based on logbook analysis, the commercial trap yield (# of legal lobsters per trap) during the five year period prior to reserve implementation ranged from 0.06-3.12 and was spatially heterogeneous about Santa Cruz and Santa Rosa Islands (Fig. 2). At the sites where we conducted post-MPA trap surveys, logbook analysis indicates that pre-MPA trap yields ranged from 0.59-0.99 legal lobsters trap^{-1}. Compared to these values, trap yields for post-MPA trap surveys were statistically different but were significantly influenced by the interaction term in our model (3-way ANOVA; Time (before vs. after) x MPA x location (In, Near, Far); $F_{4,845} = 3.6657$; $P = 0.0057$; Fig. 3). This interaction precludes discussion of the main affects as independent factors, but important trends emerge from the overall model. Most strikingly, trap yields at the Gull and Carrington *In* sites after MPA

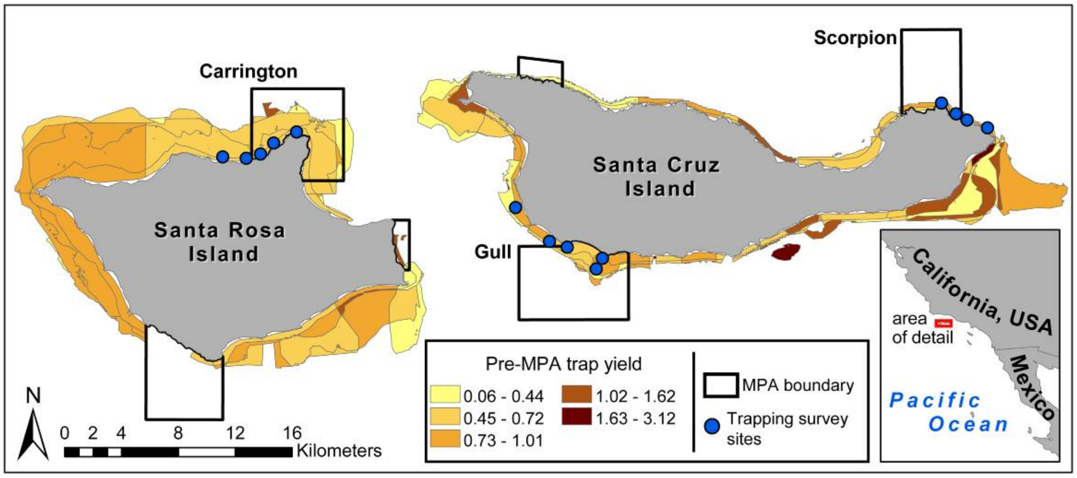


Figure 2. Sites for collaborative trap monitoring (blue dots) and historical (pre-MPA; 1998-2003) trap yield (# of legal lobsters per trap pull) as calculated from analysis of commercial logooks.

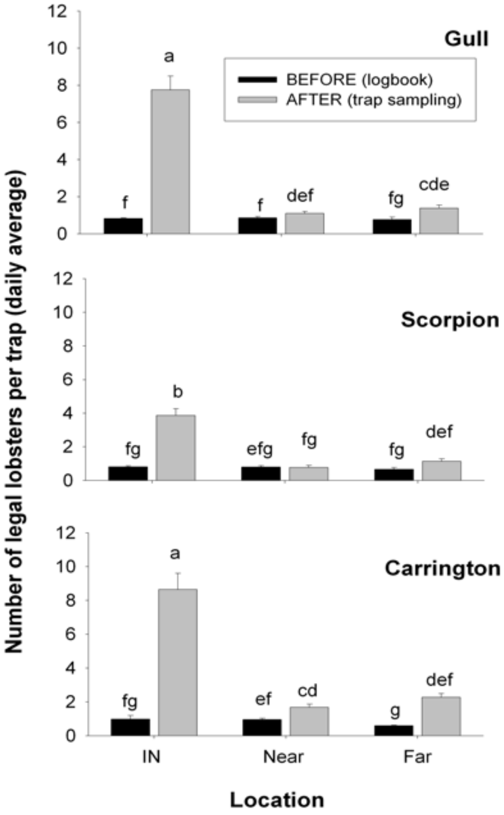


Figure 3. Mean number (± SE) of legal sized (=82.5mm) lobsters caught in replicate traps at sites within (In), immediately adjacent (Near), and 2-6km distant (Far) the three MPAs in this study. Data describing conditions before and after MPA implementation are from analysis of commercial logbooks and collaborative trapping surveys, respectively. Same letter above bars indicates means that are not statistically different (SNK, P < 0.05).

establishment (henceforth: *In-after*) were significantly higher than any other *In, Near* or *Far* sites sampled before or after MPA implementation ($P < 0.05$; Student-Newman-Keuls). Trap yield at the Scorpion *In-after* site was also statistically greater than any other *In-before*, *Near* and *Far* sites, but was statistically lower than daily mean trap yields at Carrington and Gull *IN-after* sites (Fig. 3; $P < 0.05$, SNK).

Aside from *In-after* sites at each MPA, the daily mean trap yields across all sites (*In*, *Near*, *Far*) and times (*before, after*) for each MPA were statistically indistinguishable with a few relatively minor exceptions. At Gull, the *Near-after* and *Far-after* sites were indistinguishable from each other ($P > 0.05$, SNK) but the *Far-after* site was significantly higher than all Gull sites before MPAs ($P < 0.05$, SNK). At Scorpion all sites other than *IN-after* were statistically indistinguishable ($P > 0.05$, SNK). Finally, trap yields at Carrington *Near-after* and *Far-after* were significantly higher than any *before* sites while *Far-before* was lower than *Near-before* ($P < 0.05$, SNK). Similarly, differences in trap yield across MPA's were limited in number and degree as compared to the differences between the *IN-after* sites and all other sites (Fig. 3).

The mean size of legal sized lobsters in each trap during our surveys was influenced by the interaction of MPA and location (2-way ANOVA; MPA x location; $F_{4,1823} = 4.4497$; $P = 0.0011$; Fig. 4). This interaction precludes discussion of the main affects as independent factors, but important trends emerge from the overall model. At Scorpion and Carrington, the mean size of lobsters trapped at *In* sites was statistically greater than at any other site across all MPAs. At Gull, mean size was statistically smaller than Carrington and Scorpion *In*, but statistically bigger than all other sites except for Carrington *Near* ($P < 0.05$, SNK; Fig. 4). Mean size at *Near* and *Far* sites for both Scorpion and Carrington were statistically indistinguishable, but mean size at Gull *Near* and *Far* was smaller than any Carrington site and size at Gull *Near* was smaller than Scorpion *Near* (P < 0.05, SNK). Mean size of animals at *Near* and *Far* sites at any single MPA was statistically indistinguishable (P > 0.05, SNK).

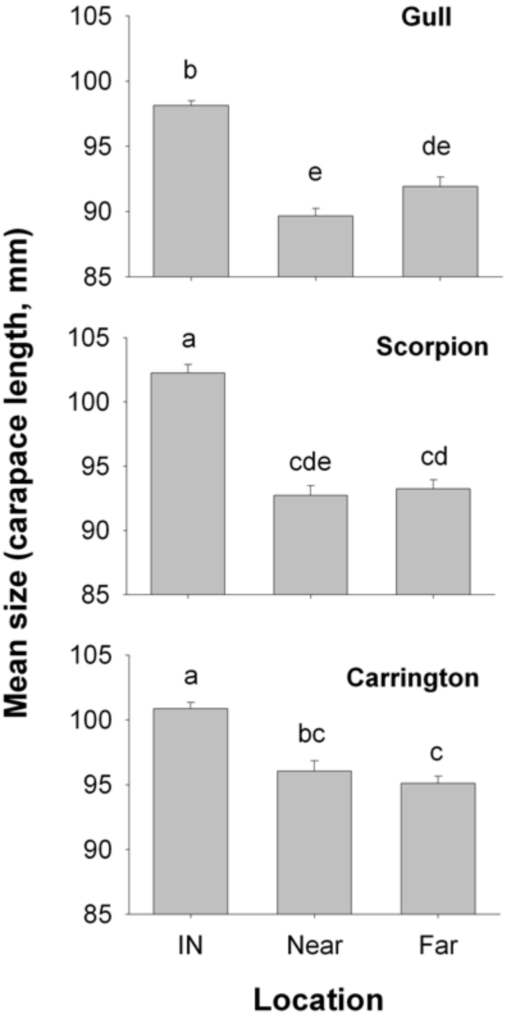


Figure 4. Mean size (± SE) of all legal (=82.5mm) lobsters caught in replicate traps at sites within (In), immediately adjacent (Near), and 2-6km distant (Far) the three MPAs in this study. Same letter above bars indicates means that are not statistically different (SNK, P < 0.05).

Size structure and port sampling

Lobster populations sampled inside MPAs had very apparent and significantly larger proportions of individuals in larger size classes when compared to populations outside MPAs ($P < 0.001$ for all *In* vs *Near* and *In* vs *Far* tests; KS; Fig. 5). In contrast, KS results for *Near* vs *Far* site comparisons indicate no statistical difference at Scorpion or Gull (KS, $P = 0.133$ and 0.182, respectively) and only a slight difference at Carrington ($P < 0.05$). Size frequency data from port sampling (fishery-dependent) showed similar patterns to data from trap surveys at *Near* and *Far* sites, but port-sampling data from

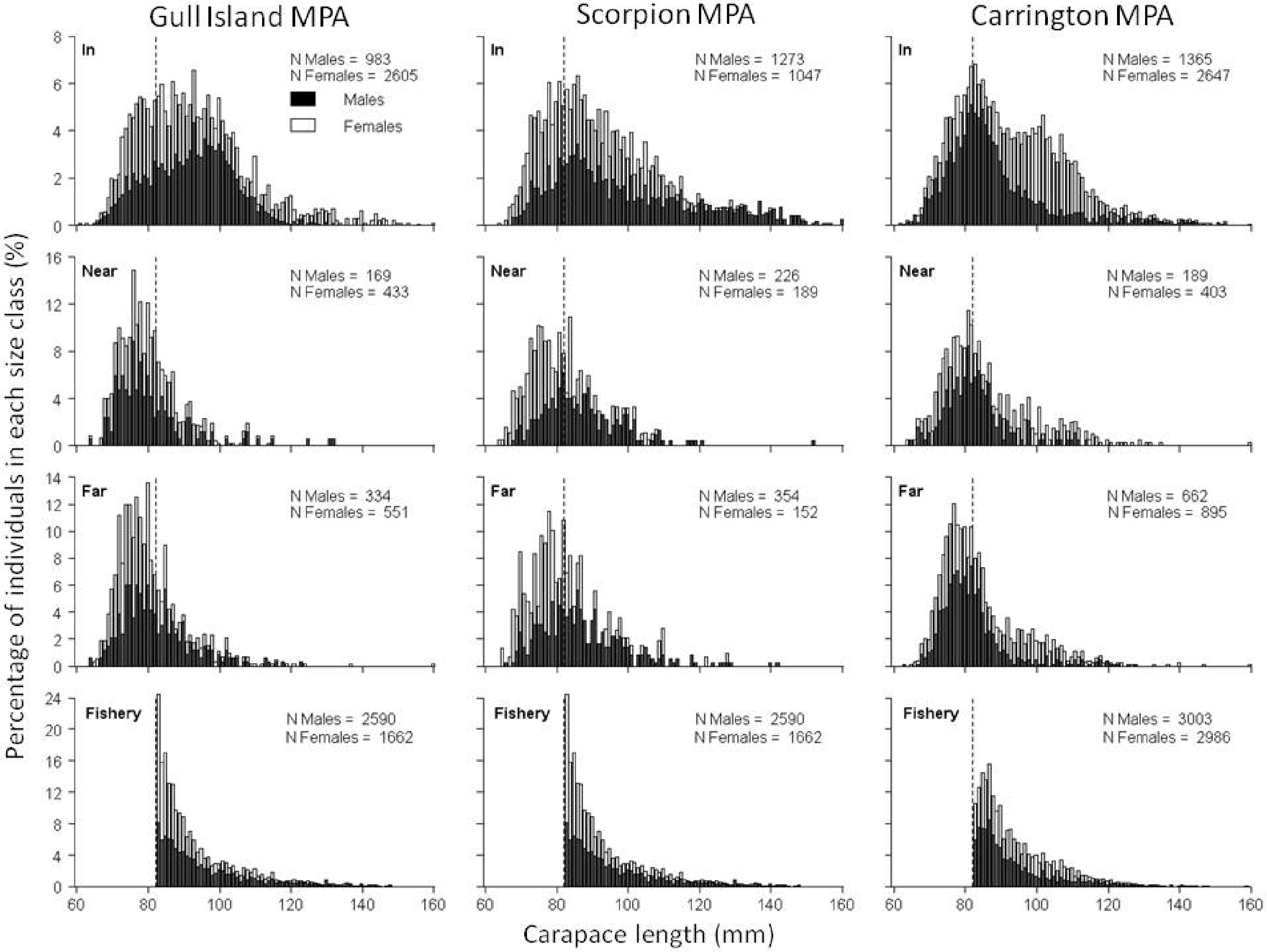


Figure 5. Length frequency histograms for lobsters caught during collaborative trapping surveys (top three rows) and concurrent port sampling of commercial catch (bottom row) from Santa Cruz and Santa Rosa Islands during the 2007-08 and 2008-09 field and fishing seasons. Dashed lines represent the legal size limit (82.5mm).

Santa Cruz and Santa Rosa were statistically different from individual and pooled *Near* and *Far* data at all three reserves ($P < 0.001$).

Addressing caveats

Examination of the potential caveats we identified for comparison of fishery-dependent data (logbook, before MPAs) against survey data (after MPAs) reveals that the caveats are unlikely to undermine our comparisons and analyses. Specifically, research traps and commercially fished traps (reported through logbooks) that were deployed in the same area during the 2006-07 and 2007-08 fishing seasons had no significant difference in catch (2006-07; $F_{1,49} = 0.0069$, $P = 0.934$: 2007-08; $F_{1,53} = 0.1.367$, $P = 0.2513$). With regard to the potential for effort heterogeneity to confound our use of trap yield as a metric for pre-reserve conditions (see *Materials and Methods* for rationale), fisherman interviews contained no statistically significant pre-MPA effort heterogeneity across our survey sites ($F_{2,25} = 1.188$, $P = 0.331$).

(H_3): High fishing effort near borders

We mapped the location of 617 total buoys at all *Near* and *Far* sites on four dates during the 2008-09 fishing season. Trap densities within our *Near* and *Far* sites were not statistically different ($F_{1,18} = 1.61$; $P = 0.221$; Fig. 6). Within our *Near* and *Far* sites, traps were generally distributed along-shore but the total number and position changed with time, but we did not observe concentration of

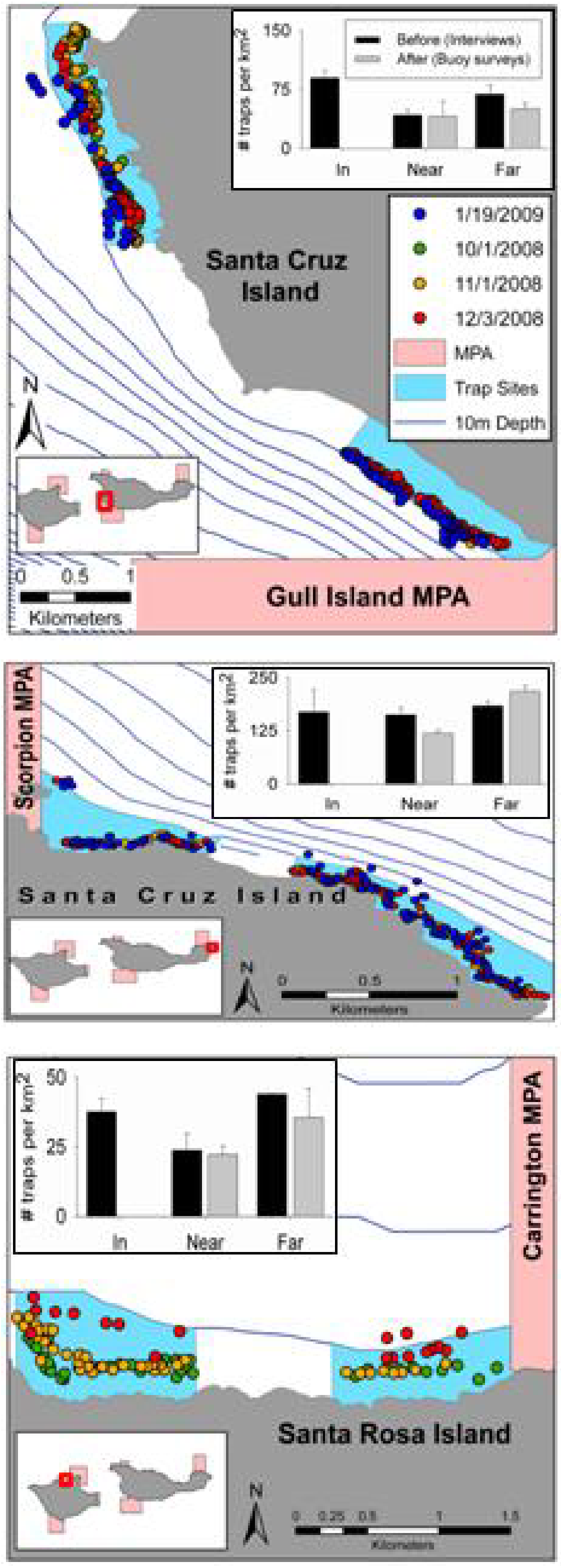


Figure 6. Commercial effort (buoy) distribution at each MPA site on four (Gull) or three (Scorpion, Carrington) dates during the 2008 season. Blue polygons are areas where collaborative trapping took place. Commercial effort was not sampled between the Near and Far polygons (sites), but was qualitatively similar. Insets show effort patterns (mean ± SE) before and after MPA implementation, as measured from fisherman interviews and buoy surveys, respectively.

commercial fishing effort near MPA borders (Fig. 6). At two of three MPAs (Scorpion and Carrington), traps were consistently absent immediately adjacent reserves, and regression analysis reveals no relationship between distance from MPA border (predictor) and the number of traps (response) within any of our three *Near* sites ($P > 0.05$ for all tests). Although we lacked the power to quantitatively compare effort at each MPA before and after MPAs, qualitative comparison of pre-MPA effort (fishermen interviews) and post-MPA effort (buoy surveys) indicates that effort at each site has not drastically shifted since reserve implementation (Fig. 6, insets).

LS1 - Discussion

Because the number and mean size of legal (≥82.5mm) lobsters captured inside MPAs was greater than in traps placed outside in fished areas, we accept our H_1 and conclude that Channel Island MPAs have conservation benefits for spiny lobster. In contrast, we did not observe any of the three patterns that were defined *a priori* as evidence of adult spillover: (1) higher trap yield at *Near* vs. *Far* sites (accept H_2); (2) higher or lower trap yield, *and* higher effort, at *Near* vs. *Far* sites (accept H_2 and H_3), or; (3) higher effort at *Near* vs. *Far* sites and/or an effort gradient near borders within the *Near* sites (accept H_3). Due to the absence of these patterns we are unable to accept hypotheses H_2 and/or H_3, and we conclude that spillover did not influence fishery dynamics during our study. This statement should not be read to infer that no spillover is occurring, for as we show in *Sections 3* and *4*, net movement of adults from within MPAs is likely. However, the number of animals moving out of reserves at this early stage after implementation does not appear, as we state here, to influence fishery dynamics.

The conservation benefits we observed in this study developed rapidly (our data are pooled from surveys 5 and 6 years after implementation) and included larger mean size, shifts in population structures towards larger size classes, and ~4 - 7.5 times more lobsters/trap inside MPAs. Such rapid responses have been observed across many taxa (Halpern and Warner 2002) and have been reported for other spiny lobster species: (a) MacDiarmid and Breen (1993) observed a 4.5 fold increase in density of *Jasus edwarsii* in a New Zealand MPA during the 2-7 year period after implementation; (b) Goñi et al. (2001) report 5 fold increases in catch for *Palinurus elephas* from a Mediterranean Sea MPA 8 years after implementation; (c) Follesca et al. (2008) observed *P. elephas* CPUE ~7.5 times greater inside a different Mediterranean MPA after 5 years, and; (d) Pande et al. (2008) provide a review of studies (most previously unpublished) that reveal similar and rapid increase trajectories for *J. edwarsii* throughout New Zealand. More broadly, MPA conservation benefits such as those we observed are reported for palinurids worldwide: *J. edwarsii* in New Zealand (Cole et al. 1990, Kelly et al. 2000, Davidson et al. 2002, Shears et al. 2006) and Tasmania (Edgar and Barrett 1997, 1999, Barrett et al. 2009), *Panulirus argus* from the Bahamas (Lipcius et al. 2001), Belize (Acosta and Robertson 2003) and Florida, USA (Davis and Doldrill 1980, Hunt et al. 1991, Bertelsen and Matthews 2001, Eggleston and Dahlgren 2001, Cox and Hunt 2005), *Panulirus interruptus* from California, USA (Iacchei et al. 2005, Parnell et al. 2005), and *Panulirus cygnus* in Western Australia (Babcock et al. 2007).

Despite the large volume of spiny lobster-MPA conservation studies, relatively few are replicated at the MPA level (but see Edgar and Barrett 1997, 1999, Kelly et al. 2000, Mayfield et al. 2005), and to our knowledge only two include data prior to reserve implementation (Shears et al. 2006, Follesca et al. 2008). Replication and pre-MPA data are critically important because they control for spatial-temporal variability that must be considered when interpreting the magnitude and direction of perceived *reserve effects* (Edgar and Barrett 1997, Willis et al. 2003, Shears et al. 2006). This is suggested by prior studies that do not show uniform spiny lobster population increases in MPAs (MacDiarmid 1991, MacDiarmid and Breen 1993, Acosta 2001, Lipcius et al. 2001, Mari et al. 2002, Mayfield et al. 2005), and is demonstrated by the differences we observed in trap yield and mean size across sites (Figs. 3 and 4).

In our study, two instances of spatial heterogeneity warrant discussion: (1) trap yields at Scorpion *In* sites were ~50% lower than *In* sites at Carrington and Gull, and; (2) mean size of legal lobsters at Gull *In* sites was statistically smaller than Carrington and Scorpion *In* sites. Elucidation of the drivers of such variability is an important frontier, and we suspect three causes: (1) differential habitat quality and connectivity at borders, (2) habitat-mediated *catchability* (probability of capture), and; (3) long-term heterogeneity of fishing effort prior to MPAs. Such variability clearly impacts conservation objectives but is also important for fisheries. Specifically, the potential for MPAs to increase fisheries yield through larval export is dependent upon increased lifetime egg production of populations inside MPAs (Beverton and Holt 1957, Guénette et al. 1998, Botsford et al. 2009). However, reproductive output and lifetime egg production from MPAs is influenced by population structure (Tetreault and Ambrose 2007, Taylor and McIlwain 2010), which is shown to vary across reserves in this study (Figs. 3-5). Additionally, there is growing interest in use of MPA populations as proxies for unfished stocks in fishery assessments (e.g., Morgan et al. 2000, Millar 2005, Wilson et al. 2010, Willis,), but our results indicate that care should be taken in selecting reference sites.

In contrast to the conservation benefits we identify for *P. interruptus* at Santa Barbara Channel Islands MPAs, we were unable to detect fishery benefits of spillover at reserve borders. Although we did not detect gradients in catch or effort near MPA borders, recent empirical studies identify such patterns as evidence of spillover. For example, Goñi et al. (2006) found that 75% of commercial effort in a *P. elephas* fishery occurred within 1km of an MPA boundary, such that catch rates precipitously declined immediately outside the MPA (due to high effort) but then increased a short distance further outside and ultimately declined linearly with distance. In another study of multiple species (including *P. elephas*) at six MPAs, Goñi et al. (2008) found statistically significant patterns of increased effort and catch near reserve borders. Similar patterns have been attributed to spillover in studies of reef fishes in the Philippines (Russ et al. 2004, Abesamis and Russ 2005, Abesamis et al. 2006) and Kenya (McClanahan and Mangi 2000, Kaunda-Arara and Rose 2004), artisanal fisheries in the Mediterranean (Harmelin-Vivien 2008, Forcada et al. 2009, Stobart et al. 2009), and a trawl fishery in northeastern USA (Murawski et al. 2005).

The absence of spillover patterns in this analysis can be explained by a number of factors. The most likely explanation is that the Channel Islands MPAs had been in place only 5-6 years prior to our study, and had not yet experienced biomass accumulations sufficient to initiate density dependent emigration. A related explanation is that lobster biomass inside MPAs may have indeed approached carrying capacity, but resource limitation that might cause emigration had not yet reached critical levels (e.g., Sanchez-Lizaso et al. 2000, Shears et al. 2006). This hypothesis is supported by recent work suggesting that, unlike the rapid responses of target organisms inside MPA's, indirect affects such as trophic cascades typically develop over decadal time scales (Babcock et al. 2010). Finally, lobster habitat at our sites is slightly less favorable immediately outside reserve borders (Kay et al. *pers. obsv.*), which might discourage spillover for reserve populations that are not critically resource limited. Research from older reserves in New Zealand suggests that lobster biomass will continue to increase in Channel Island MPAs (Kelly et al. 2000, Shears et al. 2006), such that spillover may become more likely and detectable in coming years. Finally, spillover patterns such as those we sought in this study are most prevalent for stocks that are heavily exploited. Given the fact that average sizes in areas open to fishing are well above the legal size limit (according to initial growth studies, average sizes of ~90mm (Fig. 4) correspond to animals that have been in the fishery for 2^+ seasons), and given the presence of large size classes in commercial catches (Fig. 5), it is highly unlikely that lobster stocks at the Santa Barbara Channel Islands are fully exploited (i.e., nearly all legal animals are taken in a single season).

An alternative interpretation of our results is that we may have not observed spillover because lobsters move on spatial or temporal scales that our sampling failed to detect (i.e., Type I error). The first assertion (inadequate spatial coverage) is not valid because tagging studies at our sites indicate that ~65% and 75% of *P. interruptus* recaptured by commercial fishermen had moved less than 1 and 2km from sites of tagging, respectively (Kay et al. 2008). These distances are adequately bracketed by our experimental design (Fig. 6). With regard to temporal coverage, it is possible that we may have not captured seasonal migration events. Such events are often associated with winter storm events in California, and most of our work was conducted in late summer and fall (but commercial effort surveys were made during late winter). However, in this study we explicitly test the predictions of spillover as it is typically defined and modeled as a process that is driven by non-seasonal movement due to density dependence (Polacheck 1990, DiMartini 1993, Sanchez-Lizaso et al. 2000), diffusion (Hilborn et al. 2006, Kellner et al. 2007, Walters et al. 2007), or home ranges that cross MPA borders (Moffit et al. 2009). Exploration of temporally dynamic (e.g., seasonal, ontogenetic) emigration from MPAs is newly developing (Botsford et al. 2009).

The results of *Section 1* are important because they are replicated across multiple MPAs, include data before and after MPA implementation, and because they address both conservation and fisheries affects of an MPA network. Another important aspect of *Section I* is our collaborative approach and use of fishery dependent data to inform an MPA assessment. To be sure, we are not the first to use fishery-dependent data in an MPA assessment: Murawski et al. (2005) compared catch and effort data in their seminal study of closures on the grand banks off of the northeastern USA, and Goni et al. (2006) relate survey catch data to commercial effort in a manner similar to our approach here. Similarly, the use of commercial fishing equipment to sample in and around MPAs is not new (e.g., Rowe 2002, Goni et al. 2006). However, the combination replication, data prior to reserve implementation, use of fisheries-dependent data, exploration of both conservation and fishery effects, and our CFR approach distinguish this study from all other MPA assessments.

Perhaps the most important aspect of our collaborative approach was the directed transfer of LEK from the fishery to research scientists. This had significant logistical advantages over more traditional collaborative research (Kay et al. *in press*), and was balanced by training of fishermen (by scientists) as part of a broader program to develop fishery-based data collection for the California spiny lobster fishery. Through this reciprocal knowledge exchange we achieved the scientific benefits of CFR, as demonstrated by the progress in MPA monitoring specified in this report, but we also built capacity for broader adaptive learning and management for the California spiny lobster fishery.

Lobster Section 2 – SCUBA surveys: How does habitat influence lobster distribution on the seafloor inside vs. outside MPAs?

LS2 - Introduction

Spiny lobsters in temperate oceans are typically associated with rocky reefs for most of their adult lives. However, rocky reef habitats are highly variable with regard to physical structure and ecological characteristics that influence abundance and demography. Furthermore, rocky habitats (and their variability) are heterogeneously distributed on the seafloor. Consequently, a fair criticism of results such as those presented in *Section 1* is the inability to define and control for the influence of habitat heterogeneity. The absence of habitat considerations from MPA studies represents a fundamental source of uncertainty when quantifying perceived *reserve effects*.

To address this shortcoming, we conducted SCUBA surveys at each of the sites in *Section 1*, and we measured the number of lobsters present on transects relative to habitat features. In this section, we present the results of our SCUBA surveys. In *Section 3* we use the habitat measurements taken at each site as a factor to help explain the variation in lobster trapping data presented in *Section 1*.

LS2 – Materials and Methods

SCUBA surveys

We performed SCUBA surveys at each trapping site and entered these measurements into a regression analysis in order to identify the potential influence of habitat features on lobster abundance on transects. During summer 2008, we conducted a grand total of 80 SCUBA transect surveys across 13 of the 14 trapping sites inside and outside of the 3 MPAs (Gull, Scorpion, Carrington). At each site we conducted a minimum of 6 transect surveys, and transects were 45m long x 10m wide (450m^2 x 6 transects per site = ≥ 2700m^2 of reef surveyed at each site). Divers recorded the total number of lobsters encountered on each transect, and also recorded the number of dens, caves and extent of linear cracks that could be occupied by lobsters. Dens were defined as any cave-like open space in solid rock or boulders that was enclosed on all sides except for 1-3 openings (openings no greater in diameter than the depth of the cave-like structure), and which 1-3 legal-sized lobsters (82.5mm carapace length) could occupy and use as defensible space in the presence of predators. Caves were defined as openings whose inner diameter exceeded the diameter of the entrance and that could serve as refuge for 3^+ legal-sized lobsters. Linear crack was defined as any fissure that was deep enough to be occupied by a legal sized lobster but lacked the walls of a den, such that a potential predator could approach a lobster from any direction in a 180^O arc (in two-dimensional space) as opposed to the aperture of a den entrance.

In addition to dens, caves and linear crack, we also measured the substrate type and relief on each transect at 45 points spaced every 1 meter along a transect tape and an additional 15 points at ½ meter increments on the transect tape (60 sampling points on each transect). At each sampling point divers recorded the substrate type (boulder, cobble, bedrock, sand or the habitat forming tubeworm *Phragmatapoma californica*) and measured relief as the greatest difference in reef height in a square 1m wide by 1/2m long (four categories: 0-10cm, 10cm-1m, 1-2m, and >2m). In addition, divers counted the total number of purple sea urchins (*Strogylocentrotus purpuratus*) and stipes of giant kelp (*Macrocystis pyrifera*) on each transect.

Data analysis

To test whether and how lobster abundance on transects was influenced by MPA protection, fishing, and the habitat features described above, we specified the following multivariate regression model:

$$Y_n = \alpha + \beta X + \gamma MPA + \varphi Gull + \delta Carrington + \varepsilon_s \qquad (1)$$

where response variable Y_n is the number of lobsters recorded on each transect; α is the intercept (constant); X is a column vector of the independent variables (depth; # of caves; # dens; total meters of linear crack; proportion of the substrate that was bedrock, boulder, cobble, or sand; total # of stipes of giant kelp *Macrocystis pyrifera*; # of purple sea urchins, and; # red sea urchins); MPA is a dummy variable for traps inside any MPA; Gull and Scorpion are dummy variables for sites (inside or outside MPA) associated with the Gull and Carrington MPAs, respectively.

The coefficients in row vector β describe the influence of each independent variable (column vector X) on the number of lobsters observed on each transect. Similarly, coefficient γ describes the influence of MPA (inside vs. outside any of the three MPAs) on the number of lobsters observed; coefficients φ and δ account for the average differences in response variables at Scorpion (the omitted category) vs. Gull and Carrington, respectively. Parameter ε_s is the error term (variance not explained by the model).

LS2 – Results

Model (1) accounts for 76.7% of the variability in the number of lobsters observed on SCUBA transects ($r^2 = 0.7670$), and five of the predictor variables had statistically significant influences on lobster abundance (Table 3). Transects inside MPAs had an average of 7.915762 more lobsters that transects outside of MPAs, and this average difference was highly statistically significant ($P < 0.001$). Coefficients for Carrington and Gull indicate that transects associated with these MPAs had and average of 10.77463 and 5.325978 more lobsters per transect than transects at Scorpion sites.

Interestingly, this average difference was statistically significant for Carrington ($P = 0.007$), whereas the average difference between Gull and Scorpion was not statistically significant ($P = 0.081$).

Data for the predictor variable "# Dens" are continuous and recorded as the total number of individual dens on each transect, such that the coefficient value of 0.4249668 is interpreted as: "For every den that is added to a transect, 0.4249668 more lobsters are present." This relationship is statistically significant ($P < 0.001$). The number of caves (predictor variable "# Caves") per transect also had a large and statistically significant impact ($P = 0.002$) on the # of lobsters per transect (coefficient interpretation: for every cave that was on a transect, 2.25112 more lobsters were present). For every 1meter of linear crack on a transect, 0.1735959 more lobsters were present. This relationship was considerably smaller than for the "# Dens" and "# Caves" predictors and was not statistically significant ($P = 0.101$). Clearly, "# Dens" and "# Caves" are superior to "Crack (meter)" as predictors of lobster abundance.

Table 3. Results for equation 1, in which the response variable is the number of lobsters present on a 450m^2 SCUBA transect. Model $R^2 = 0.7670$; $F_{(13,42)} = 10.64$; # observations = 56.

Predictor variable	**Coefficient**	**Std. Err.**	**t**	**P**	**95% Confidence Interval**	
In MPA (dummy)	**7.915762**	1.984756	3.99	**0.000**	3.910363	11.92116
# Cave	**2.25112**	0.6729691	3.35	**0.002**	0.8930136	3.609227
Crack (meters)	0.1735959	0.1033797	1.68	0.101	-0.0350328	0.3822246
# Dens	**0.4249668**	0.0858373	4.95	**0.000**	0.2517401	0.5981936
% bedrock	3.082825	20.81632	1.48	0.146	-1.118078	7.283728
% boulder	3.75944	22.00847	1.71	0.095	-.6820481	8.200929
% cobble	3.690699	25.14551	1.47	0.150	-1.38387	8.765268
% sand	2.187709	23.19816	0.94	0.351	-2.493869	6.869288
# Kelp stipes	-0.0088444	0.0070201	-1.26	0.215	-0.0230116	0.0053227
# Purple urchins	**-0.0109916**	0.0038724	-2.84	**0.007**	-0.0188064	-0.0031767
# Red urchins	-0.003954	0.0075352	-0.52	0.603	-0.0191606	0.0112527
Scorpion	(dropped)					
Carrington	**10.77463**	3.799566	2.84	**0.007**	3.106797	18.44246
Gull	5.325978	2.98386	1.78	0.081	-0.6956956	11.34765
Constant (α)	-36.22271	19.79592	-1.83	0.074	-76.1725	3.727082

For every 1% increase in the occurrence of benthic habitat predictor variables "% bedrock", "% boulder", "% cobble", and "% sand", the number of lobsters per transect increased by 3.082, 3.759, 3.691, and 2.187, respectively, but none of these relationships were statistically significant (all $P >> 0.05$, Table 3).

The ecological predictor variables "# of Kelp stipes" and "# Red urchins" did not significantly influence the number of lobsters observed on SCUBA transects ($P = 0.215$ and 0.603, respectively). In contrast, the addition of one purple urchin was associated with -0.0109916 fewer lobster (coefficient is negative), and the relationship was statistically significant, $P = 0.007$).

LS2 – Discussion

Higher lobster abundance on transects inside SBCI MPAs corroborates the results of trapping surveys presented in *Section I.* The impact of reserves was similar in magnitude in both survey types: SCUBA transects inside MPAs had an average of 7.92

more lobsters than those outside, and trap yields inside reserve were ~4-8 times higher than yields outside (*Section I*, Figure 3). The consistency between survey results provides compelling evidence for SBCI MPA conservation benefits. The most important contribution of SCUBA surveys, however, is that they permitted identification of habitat features that influence localized lobster population abundance.

The multivariate regression model that we specified (equation 1) accounted for a high degree (76.7%) of the variation in lobster abundance observed on transects. Consequently, individual predictor variables in the model that were statistically significant are important factors that govern local abundance patterns. Among the significant predictor variables, the most important were the number of dens and the number of caves on transects, as well as transect location relative to an MPA (i.e., inside vs. outside).

Among the two habitat variables that were significant, the number of caves had a higher coefficient than the number of dens, and so had a larger marginal effect (i.e., effect per cave or den) on lobster abundance. However, when the abundance of each habitat feature is taken into consideration dens have a greater impact on lobster abundance because they are more numerous on transects (average per transect = 1.3 caves and 15.8 dens). The interaction between habitat (dens) and MPA protection as factors in this model is illustrated by the relative influence of den abundance inside vs. outside MPAs. Not surprisingly, den abundance inside MPAs had a steeper relationship with lobster abundance than den abundance outside MPAs (Figure 7).

An interesting extension of the relationship between dens and lobsters

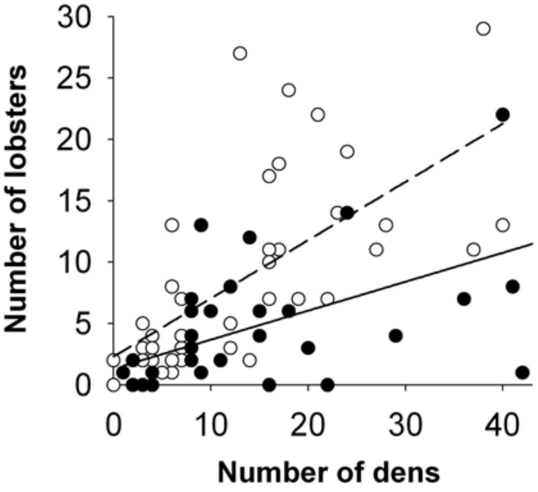


Figure 7. Relationship between the number of lobsters on a transect, the number of dens per transect, and transect location inside (open white circles) or outside (closed black circles) MPAs. N = 40 for each location (In vs. out, total N = 80). Equations for lines are: MPA sites (dashed line) $y = 0.475x + 2.311$, $R^2 = 0.4198$; fished sites (solid line) $y = 0.236x + 1.331$, $R^2 = 0.3591$.

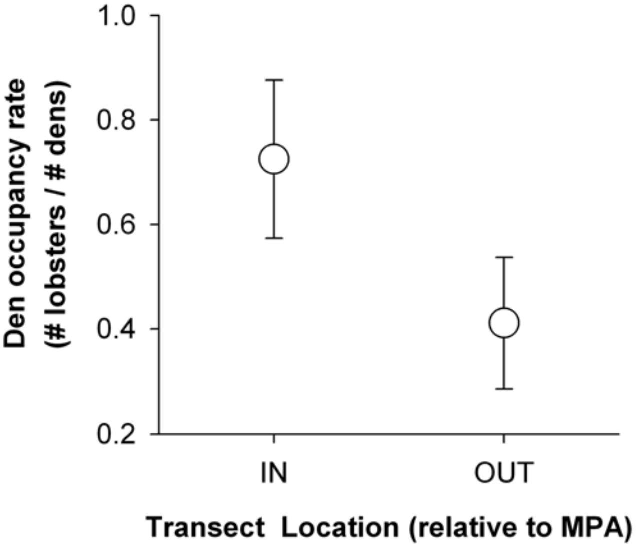


Figure 8. Den occupancy rate (# lobsters / # dens on each transect) for transects located inside and outside MPAs. Error bars are 95% confidence intervals. N = 40 for each location (80 total transects).

on transects is the occupancy rate of dens (i.e., the number of dens that were occupied by a lobster, calculated here as total lobsters divided by total dens per transect). Transects in MPAs had a significantly higher occupancy rate than dens outside reserves (Figure 8, $P < 0.05$). The high occupancy rate inside reserves might reflect lobster populations that are approaching den saturation or carrying capacity with regard to shelter space, but this is highly speculative. It will be interesting to track these changes in the future and determine if lobster abundances continue to increase even as dens become saturated.

The only ecological variable that was significantly related to the number of lobsters per transect was the number of purple urchins. Interestingly, this relationship was negative, indicating that higher purple urchin abundance was correlated with lower lobster abundance. This relationship raises a challenging problem when interpreting ecological data in the context of regression models. Specifically, interpretation and assignment of causality is confounded because the response variable (number of lobsters) can influence the abundance of the predictor variables (lobsters eat urchins, urchins eat kelp).

Section 3 – How are the MPA effects from *Section I* influenced by habitat and distance from MPA borders?

LS3-Introduction

This section presents an additional analysis of trapping data from *Section I*. However, the regression model employed here controls for the influence of habitat features measured in *Section 2* at each trapping area. Additionally, we use the distance of each trapping area from the nearest MPA border as a predictor for trap performance. The disadvantage of this approach, and the reason that *Sections 1* and *3* are not combined into a single analysis, is that the current analysis is incompatible with the fishery dependent data (logbooks and interviews) from *Section 1.* Thus, *Section 3* accounts for spatio-temporal variability by controlling for site specific differences in habitat and distance from MPA borders, whereas *Section I* addressed this variability via a before vs. after MPA comparison of trap yield at all sites. The importance of accounting for spatio-temporal variability in MPA studies is presented in *Section I* (*LS1 – Discussion*).

LS3 – Materials and Methods

Data collection

Trapping data were collected as described in *Section 1*, and habitat data were collected as described in *Section 2*

Statistical approach

The first objective of this section of the report is to compare trap performance (number and mean size of legal lobsters) inside vs. outside the three MPAs while controlling for important habitat features that might vary across site and influence trap performance. To accomplish this, we specified the following multivariate regression models:

$$Y_n = \alpha + \beta X + \gamma MPA + \varphi Scorpion + \delta Carrington + \rho 2006 + \omega 2007 + \varepsilon_n \quad \text{(equation 2)}$$

$$Y_s = \alpha + \beta X + \gamma MPA + \varphi Scorpion + \delta Carrington + \rho 2006 + \omega 2007 + \varepsilon_s \quad \text{(equation 3)}$$

where response variables Y_n and Y_s are the number and mean size of legal sized lobsters in traps, respectively; α is the intercept (constant) in each model; X is a column vector of the independent variables (trap depth, number of nights traps were deployed, linear crack (meters), caves, and dens); MPA is a dummy variable for traps inside any MPA; Scorpion and Carrington are dummy variables for sites (inside or outside MPA) associated with the Scorpion and Carrington MPAs, respectively; 2006 and 2007 are dummy variables for data collected during those years; and ε_n and ε_s are the error terms (variance not explained by the models).

The coefficients in row vector β describe the influence of each independent variable (column vector X) on the number (equation 2) and mean size (equation 3) of lobsters caught. Similarly, coefficient γ describes the influence of MPA (inside vs. outside any of the three MPAs) on the number and mean size of lobster caught; coefficients φ and δ account for the average differences in response variables at Gull (the omitted category) vs. Scorpion and Carrington, respectively; and ρ and ω account for the average differences in response variables during 2008 (the omitted category) vs. 2006 and 2007, respectively.

Our second objective was to test the relationship between trap performance and spatially explicit distances of traps from MPA borders. To address this, we measured the midpoint distance of each trapping area from the nearest MPA border and used these values as predictors of trap performance at each area. These measurements replaced the MPA dummy in equations (2) and (3), such that two new models were specified:

$$Y_n = \alpha + \beta X + \gamma outdist + \Phi indist + \varphi Scorp + \delta Carr + \rho 2006 + \omega 2007 + \varepsilon_n \quad \text{(equation 4)}$$

$$Y_s = \alpha + \beta X + \gamma outdist + \Phi indist + \varphi Scorp + \delta Carr + \rho 2006 + \omega 2007 + \varepsilon_s \quad \text{(equation 5)}$$

where outdist and indist are dummy variables for trapping areas outside and inside any MPA, respectively. and γ and Φ describe the influence of distance from MPA borders on trap performance in those areas. All other terms in the equation are as described above for models (2) and (3).

Distances of trapping areas from the nearest MPA border were measured as straight-lines that paralleled the shoreline from the center of each trapping area to the intersection of the border and shoreline. For all models, numerical values for habitat features that were measured during SCUBA surveys (crack, caves, dens) are the averages from six transects at each trapping area, such that all traps at within an area were associated with a common score for each habitat variable. Due to this averaging of trap distance and habitat scores, standard errors during the analyses were clustered for the 14 areas. These models were specified to be robust to heterogeneity of variance and non-normal distribution of data. Statistical significance was determined at $\alpha = 0.05$.

LS3 - Results

As in *Section 1*, models 2 and 3 in this section indicate that traps placed inside MPAs caught significantly more and larger sized lobsters than traps placed outside (Table 4). Specifically, an average of 4.88 more lobsters per trap were caught inside MPAs, and the mean size of individuals was 8.77mm larger inside MPAs, and both effects were highly significant (Table 4). Unlike *Section 1*, however, models 2 and 3 in this section explicitly control for the influence of other predictor variables in determining the significance of the “IN/OUT MPA” variable. Other predictors that significantly influenced the response variables in model 2 included “# of Dens”, individual MPA (Carrington, Scorpion and Gull), and “Year 1/Year 2/Year 3”. Significant predictor variables in model 3 include “# of Caves”.

Table 4. Results of multivariate linear regression tests 2-5 (see text for model specifications) for the influence of experimental and environmental factors (predictor variables) on the number and mean size (response variables) of legal-sized lobsters that were captured in traps. The response variable in each model is listed (in parentheses) in the top row. Values reported for each predictor variable include the coefficient, t-statistic robust to heteroscedasticity (in parentheses), and statistical significance (P-value or 1, 2, or 3 asterisks to indicate statistical significance at α = 0.05, 0.01, and 0.001, respectively).

Predictor	Model 1 (# of lobsters)	Model 2 (mean size)	Model 3 (# of lobsters)	Model 4 (mean size)
In MPA (dummy)	**4.88 (17.88)*** **	**8.77 (21.02)*** **	---	---
Distance In MPA	---	---	**0.002 (2.92)* **	**0.002 (2.19)* **
Distance Out MPA	---	---	-0.0005 (2.12) 0.054	**-0.002 (-3.70)** **
Depth	-0.01 (-0.72) 0.483	0.03 (1.57) 0.140	-0.013 (-0.83) 0.422	0.03 (1.43) 0.176
Nights	0.28 (1.84) 0.089	-0.11 (-1.81) 0.093	0.28 (1.82) 0.092	-0.109 (-1.88) 0.082
Linear Crack	0.02 (1.67) 0.119	0.05 (1.64) 0.125	0.05 (1.26) 0.229	**0.25 (2.58)* **
Caves	-0.005 (-0.04) 0.97	**-1.13 (-4.23)** **	0.28 (0.60) 0.557	-1.43 (-1.63) 0.126
Dens	**0.08 (3.88)** **	-0.01 (-0.04) 0.770	**0.25 (2.83)* **	**0.54 (3.70)** **
Gull	Dropped	-0.96 (-1.61) 0.132	Dropped	3.56 (1.58) 0.138
Scorpion	**-4.04 (-13.13)*** **	Dropped	**-3.98 (-2.74)* **	Dropped
Carrington	**4.38 (0.66)*** **	2.15 (1.86) 0.085	-1.11 (-0.76) 0.459	-4.11 (-1.42) 0.182
Year 1	**-2.06 (0.82)* **	-0.22 (-0.20) 0.846	**-1.85 (-2.27)* **	0.04 (0.03) 0.975
Year 2	**-1.15 (0.029)* **	-0.44 (-0.54) 0.601	**-1.09 (-2.31)* **	-0.36 (-0.04) 0.667
Year 3	Dropped	Dropped	Dropped	Dropped
Constant	**-32.70 (-8.95)*** **	**97.89 (15.83)*** **	-10.75 (-1.17) 0.263	**116.77 (7.57)*** **
Observations	3361	2312	3361	2312
r^2	0.36	0.19	0.35	0.19

Among trapping areas located inside MPAs, there was a significant positive relationship between the distance inside MPA borders and both the number and mean size of lobsters captured in traps (Table 4, Figs. 10A, B). Models 4 and 5 indicate that for every meter that a trap is placed further inside an MPA (i.e., alongshore and towards the center of the MPA), trap yield increased by 0.002 lobsters per trap and size (carapace length) increased by 0.002mm ($P < 0.05$ for both response variables; Table 4). Among trapping areas outside of MPAs, there was a negative relationship between distance from nearest MPA border and the response variables (Table 4, Figs. 10A, B). Models 4 and 5 indicate that for every meter that a trap is placed further outside an MPA border, yield decreased by 0.0005 fewer lobsters (but the relationship was marginally insignificant, $P = 0.054$), and size significantly decreased by 0.002mm ($P < 0.01$).

LS3 – Discussion

Two important results of this analysis warrant emphasis: (1) significant differences in trap performance inside vs. outside MPAs, which were previously identified in *Section 1*, persist in a model that controls for site specific differences in habitat and location relative to MPA borders, and; (2) trap performance inside MPAs

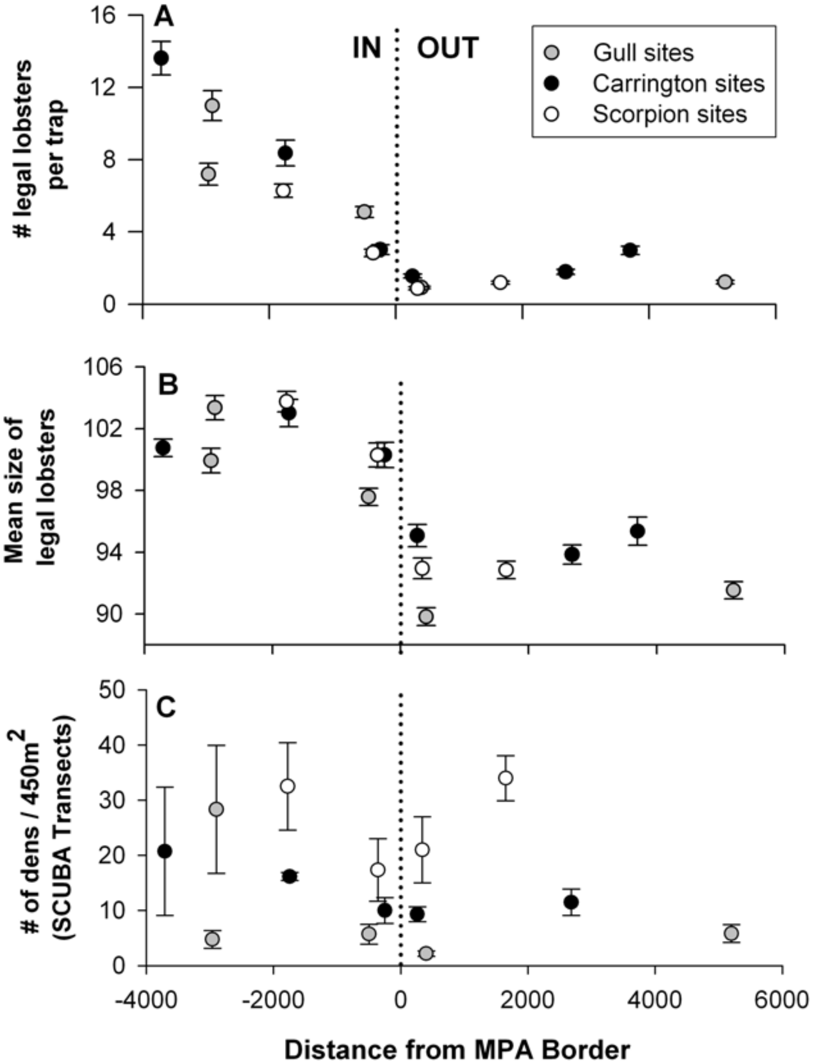


Figure 9. The influence of distance from MPA border on the trap yield (A) and mean size (B) of legal sized (≥ 82.5mm) lobsters captured in traps at 14 monitoring sites inside and outside of three Channel Island MPAs. The average number of lobster dens per site (C) per 450m^2 transect (see *Section 2*) was highest at sites far inside reserves, and was correlated with high trap yield and mean size. Such habitat effects are statistically controlled for in models 2-5 (Table 4). Error bars are 95% CI's.

improved (more and larger lobsters) as traps were set further inside (i.e., towards the center) of MPAs.

Spatio-temporal variability in abundance and size of individuals can confound assessment of reserve effects if sources of variability are not identified and controlled for during analysis. The Santa Barbara Channel Island MPA network provides an example of the importance of controlling for such variability, because many high productivity fishing sites were located inside MPAs (e.g., Morse Point, Gull Island and Carrington Point). Consequently, it has been argued that high trap performance at SBCI MPA sites may be due to inherently higher productivity and not reserve effects, *per se*. Identifying sources of spatial variability, and controlling for them in this analysis, has facilitated improved assessment of MPA effects in this study. As we demonstrate, heterogeneously distributed habitat features do indeed influence lobster size and abundance at sites inside and outside MPAs, but across all sites the implementation of protected areas has lead to higher abundance, trap yield, and/or average size inside MPAs.

The relationship between trap performance and distance inside MPAs suggests that some spillover may be occurring at Channel Islands even though it was not detected in *Section 1*. This is true because among conspecifics inside MPAs, those that are closer to borders are more likely to move outside MPAs and become captured in a fishery. As a result, theoretical studies predict that the abundance of target organisms inside MPAs is likely to be lowest near borders and highest at the center of reserves (Hilborn et al 2006, Moffitt et al 2009). These simplistic patterns are likely to be modified by habitat heterogeneity and connectivity. Because we controlled for habitat variability in our analysis, however, the most likely explanation of increased trap performance at MPA centers is differential spillover of individuals near borders. This conclusion is supported by recapture of lobsters that were originally tagged inside MPAs but later recaptured in areas open to fishing: The proportion of animals tagged inside reserves and later recaptured in the commercial fishery decreased with distance towards the center of MPAs (Figure 11). Furthermore, tagged animals were most often recaptured < 1km from the original tagging site. Although spillover is likely occurring near borders, it bears repeating that spillover was not evident outside MPAs during research trapping surveys or commercial effort surveys (see *Section 1*).

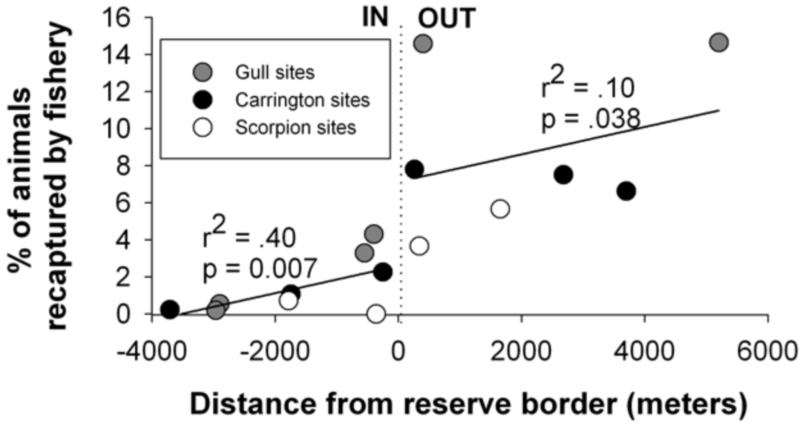


Figure 10. The percentage of lobsters tagged at each site during 2006-07, 2007-08, or 2008-09 summer field seasons (prior to opening of fishing season) that were later recaptured in the recreational or commercial fishery in any year 2006-2009.

Lobster Section 4 – Growth and movement

LS4 - Introduction

In addition to the assessment of Santa Barbara Channel Island MPAs described in *Sections 1-3*, this project has generated important life history data and essential fishery information (EFI) for California spiny lobster. This information includes the population structure data presented in *Section 1*, as well as research focused upon lobster growth rates (i.e., change in size or weight of individual lobsters though time) and movement patterns. Because there is no known method for determining the age of juvenile and adult lobsters that are captured from the wild, growth and movement information can only be collected through tagging studies. This section describes the processes and initial results from tag recapture studies we conducted in collaboration with commercial fishermen at the Santa Barbara Channel Islands.

Growth rates are critical for many traditional stock assessment models that relate temporal changes in population size (numbers or biomass), catch, and recruitment. The California Department of Fish and Game is currently in the initial phases of preparing a formal stock assessment for the California spiny lobster fishery. Growth rates from our study may be useful to the DFG assessment, and we are in communication with the assessment's chief analyst, Dr. Doug Nielson. In addition to supporting traditional stock assessment models, our growth data for spiny lobster are intended for use in a low cost MPA-based harvest rate model that we are developing.

Growth is a physiological process that is influenced by environmental conditions such as food availability and temperature. Because ecological and oceanographic conditions are geographically variable along the California coast, it is reasonable to expect that growth rates might mirror this variability. At the very least, our growth rate estimates will provide data from areas that are at the northernmost (and coldest) portion of spiny lobster range and that are ecologically distinct from many mainland reefs.

Movement of adult animals is the process by which adult biomass inside MPAs is redistributed to fished areas. The distances that individual animals are likely to move, and the time scales associated with those movements, govern important MPA-fishery interactions associated with spillover. These interactions include the export of yield form with MPAs to adjacent fisheries, effort distribution outside the MPAs, and gradients in organism abundance near reserve borders. Understanding movement patterns is therefore a powerful tool for predicting influences of MPAs on target populations and fisheries performance. The ability to make such predictions is especially important as California (and the world) continues to expand marine reserve implementation as a form of management.

LS4 – Materials and Methods

Lobsters were trapped and sampled as described in *Section 1* (*LS1 – Materials and Methods*). Briefly, as traps was retrieved we recorded the depth, time, date, and GPS coordinates for each trap, as well as the total number, sex, carapace length (nearest mm using vernier calipers) injuries (e.g., missing legs or antennae), and breeding condition of all lobsters in the trap. In addition, each lobster was tagged with an individually numbered T-bar tag (TBA-2 standard, Hallprint tags Australia) applied through a thin membrane on the ventral surface between the tail and carapace, such that the "T" portion of the tag was anchored in muscle and would persist through molting. After lobsters had been processed, they were returned to the exact location of capture and released by hand.

Movement of adult lobsters was calculated from tagged lobsters that were recaptured by commercial and recreational fishermen during the fishing season. Although we routinely captured tagged animals during scientific sampling, our distribution of traps was spatially confined and would have underestimated movement because animals travelling outside our research trapping areas would not have been recaptured. In contrast, commercial and recreational fishing effort around the Channel Islands is much more broadly distributed and provides the best and least biased coverage for recapturing tagged animals.

Prior to each fishing season, one of us (Kay) met with commercial fishermen and operators of commercial passenger fishing vessels (CPFV) who fish or lead lobster charters at the Channel Islands. During these meetings, skippers were provided a tag recapture kit that included vernier calipers and a data sheet with instructions to record the tag number, carapace size (in mm), sex, exact GPS coordinates, date, physical/reproductive condition, and nearest geographical landmark for recaptured lobsters. Data sheets were collected at the conclusion of each fishing season.

The GPS positions of tagging and recapture sites for all individual lobster was entered into a GIS database (ArcGIS 9, ESRI) for analysis. The distance moved by individual lobsters was calculated using the Hawth's Tools extension for ArcGIS (downloaded online from: http://www.spatialecology.com/htools/tooldesc.php). Hawth's Tools calculated straight lines connecting tagging and recapture locations in ArcGIS. Straight lines that crossed land (and of the Channel Islands) were re-routed through manually placed nodes in ArcMAP, such that movement paths for each animal could now consist of two or more straight segments. Movement paths and lengths that were atypical and outliers, and which contained additional discrepancies in the tagging and/or recapture data (e.g., sex or size), were excluded from analysis to avoid bias due to recording error.

Unlike movement data, growth data reported here were collected exclusively during research trapping. The purpose was to limit the error associated with many individuals recording sizes to the nearest millimeter.

LS4 – Results and Discussion

Movement

Among commercial and recreational fishermen combined, a total of 239 and 490 recaptured lobsters were reported during the 2007-08 and 2008-09 lobster seasons, respectively. During both seasons, tag returns were reported from tagging locations throughout our field sites at Santa Rosa and Santa Cruz islands (Figs. 11 A, B). Although a few lobsters were recaptured at great distances (e.g., up to 30km) from release sites, more than half (58.7%) of all individuals were recaptured within 1km of their initial tag and release site (Fig. 13). There was no relationship between the distance animals traveled and the time elapsed between release and recapture (Fig. 14B).

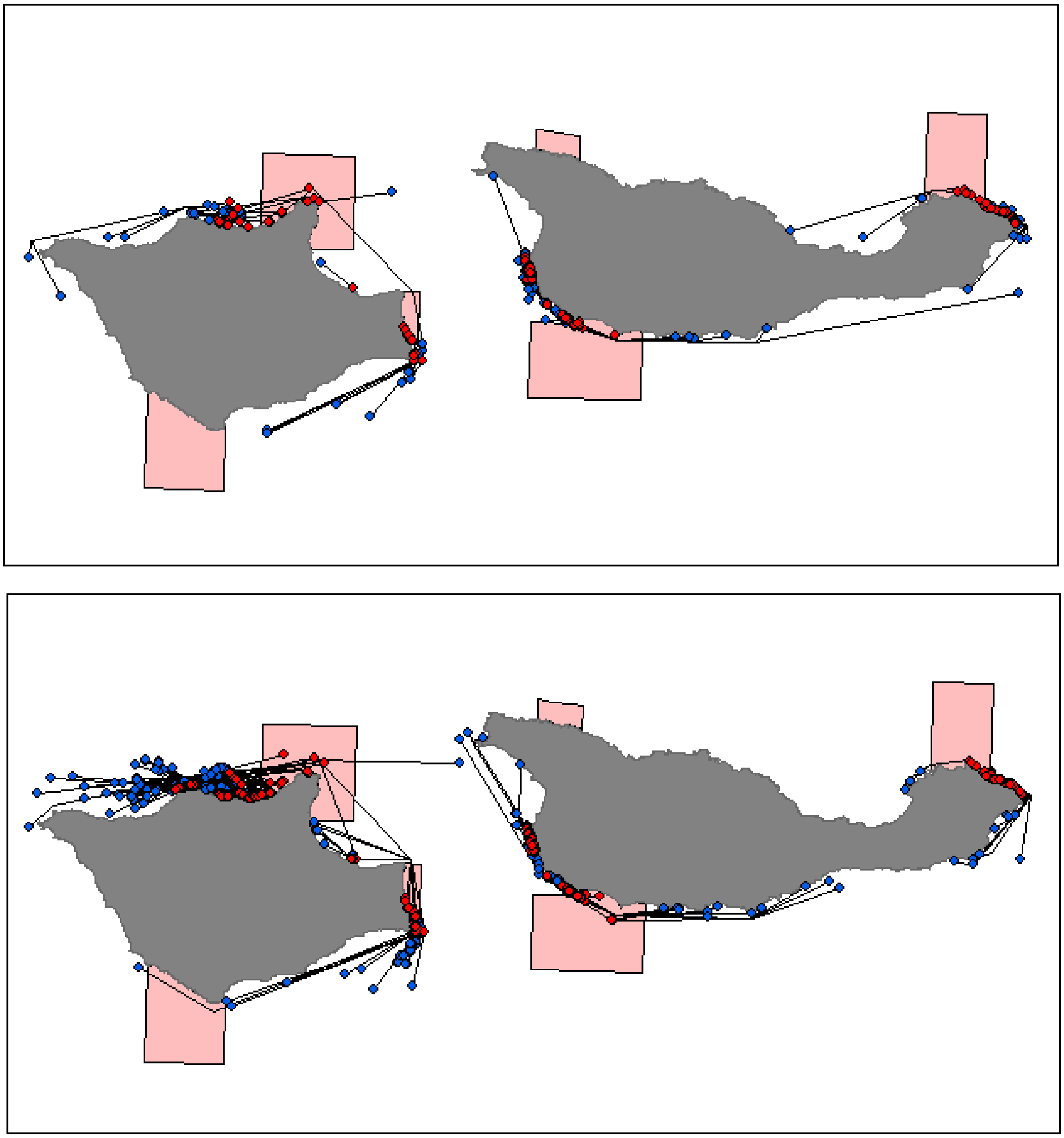


Figure 11. Map of Santa Rosa and Santa Cruz Islands showing the release (red dots) and recapture points (blue dots) for tagged lobsters that were recaptured during the 2007-08 (top panel) and 2008-09 (bottom panel) fishing seasons. Lines connecting the release and recapture points represent shortest path distances.

Distances travelled by lobsters recaptured in this study were, on average, much shorter than the dimensions of MPAs at the Santa Barbra Channel Islands. Consequently, one might conclude that animals tagged within MPA borders are likely to remain inside reserve borders. This prediction is consistent with the higher trap yield, mean size, and abundance inside MPAs reported in *Sections 1,2* and *3* of this report.

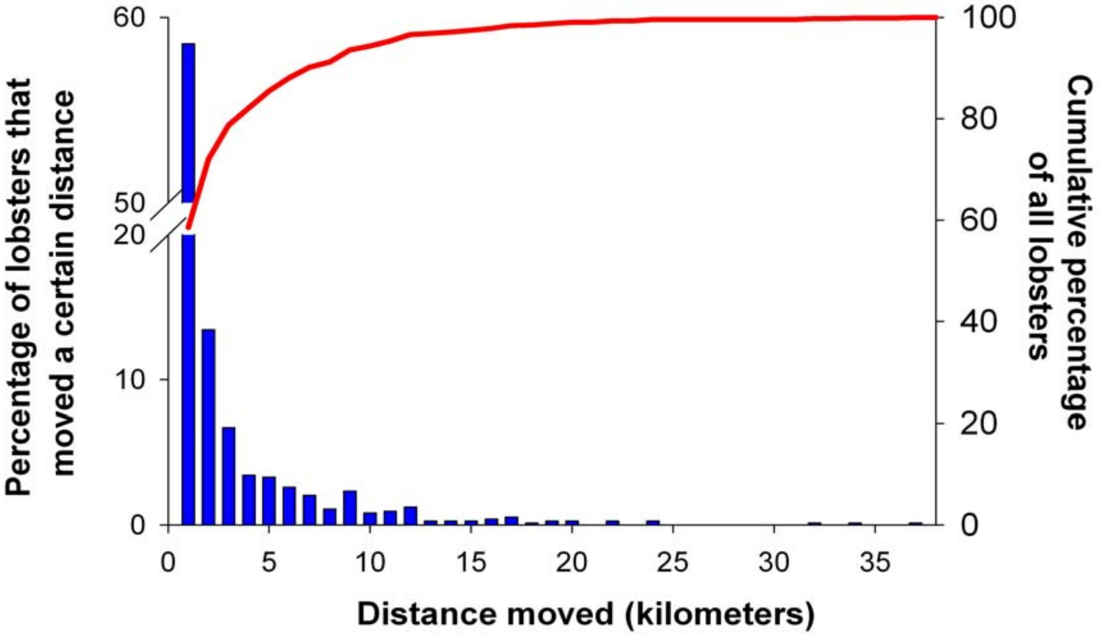


Figure 12. Distance moved by lobsters that were tagged from 2006-2008 and recaptured by recreational and commercial fishermen during the 2007-08 and 2008-09 fishing seasons. Bars correspond to the primary y-axis, and the red line corresponds to the cumulative percentage on the secondary (right) y-axis. Number of lobsters reported here = 729.

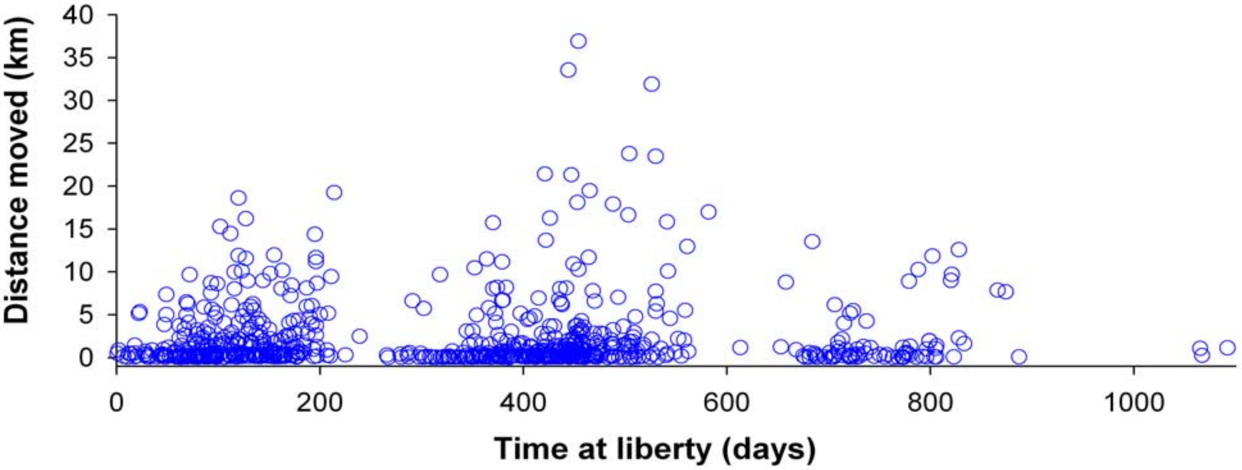


Figure 13. Distance moved vs. time at liberty (days between initial tagging and recapture) for all 729 lobsters reported from the 2007-08 and 2008-09 fishing seasons.

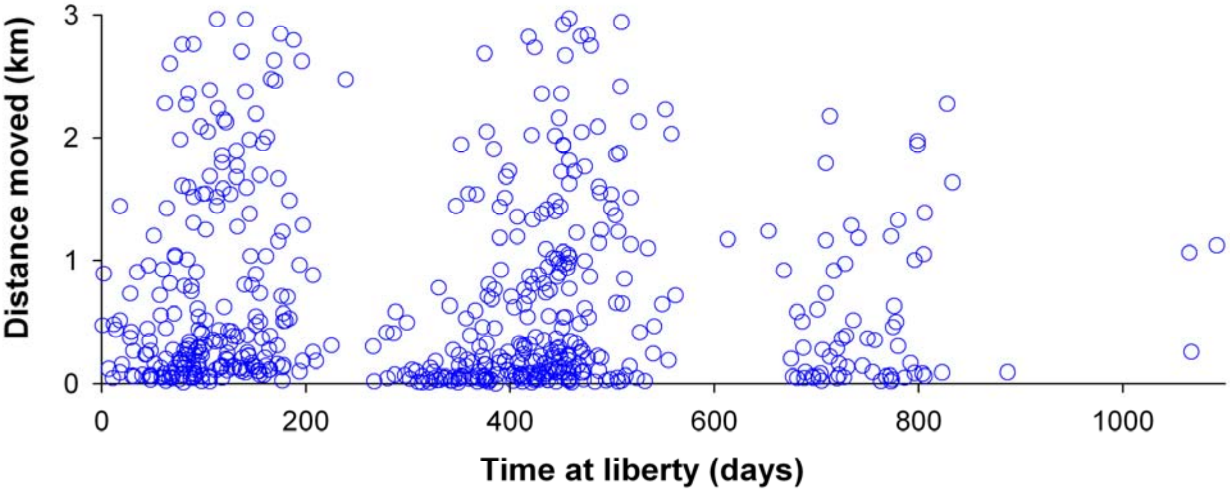


Figure 14. Distance moved vs. time at liberty (days between initial tagging and recapture) for the 78.7% of lobsters from figures 12, 13, and 14 that were recaptured <3km from release points.

Although some animals tagged within MPAs were indeed recaptured in fished areas, the majority were from reef areas immediately inside reserve borders (Fig. 10, *Section 3*). This pattern helps explain the gradients of trap yield that we observed inside MPA borders (Fig. 10, *Section 3*). From a conservation perspective, it seems reasonable to expect that the loss of animals from inside MPAs due to spillover may be limited to reef areas that are near borders. It bears repeating, however, that trap yield and mean size at sites immediately inside borders were significantly higher than in fished areas. A fisheries-relevant corollary to this dynamic is that lobster populations at the center of Channel Island MPAs may make extremely limited contributions to fishery yield via spillover. Such dynamics are predicted by theoretical MPA models (). Scales and patterns of movement are key determinants of MPA performance, and they underscore the importance of movement studies such as ours.

Growth

Initial results and analyses from growth studies suggest sexually dimorphic growth rates, which are especially pronounced at sizes >80-90mm (Fig.15). Because there is no methodology (known to us) for determining the age of wild caught *P. interruptus*, modeling growth rates proceeds as with most other molting crustaceans and is dependent upon analysis of growth increment data from tagging studies. Derizo and Quinn (1999) present a formulation of the von Bertalannfy growth model to accommodate tag-recapture data:

$$\Delta L_i = (L_\infty - L_{1i})(1-e^{-k\Delta ti}) + e \qquad \text{(equation 6)}$$

where ΔL_i is the observed change in size of an individual, L_∞ is the theoretical average maximum size for individuals in a population, L_{1i} is the length of an individual at tagging, k is the growth constant, and Δti is the time elapsed between tagging and recapture. In this model L_∞ and k are free parameters. Our current (but very initial) fit of the von Bertalannfy growth equation to annual growth increments from this study and data from Engle (1973; included for very small individuals that we were unable to recapture) indicates that female lobsters have an average maximum carapace length (L_∞) of 103.3mm (Fig. 16; k = 0.2409), and that they reach legal size at an age of ~7years (Fig. 17). These growth results are initial and include data from only one field site. More comprehensive analyses are pending, and results will be published in peer-review journals.

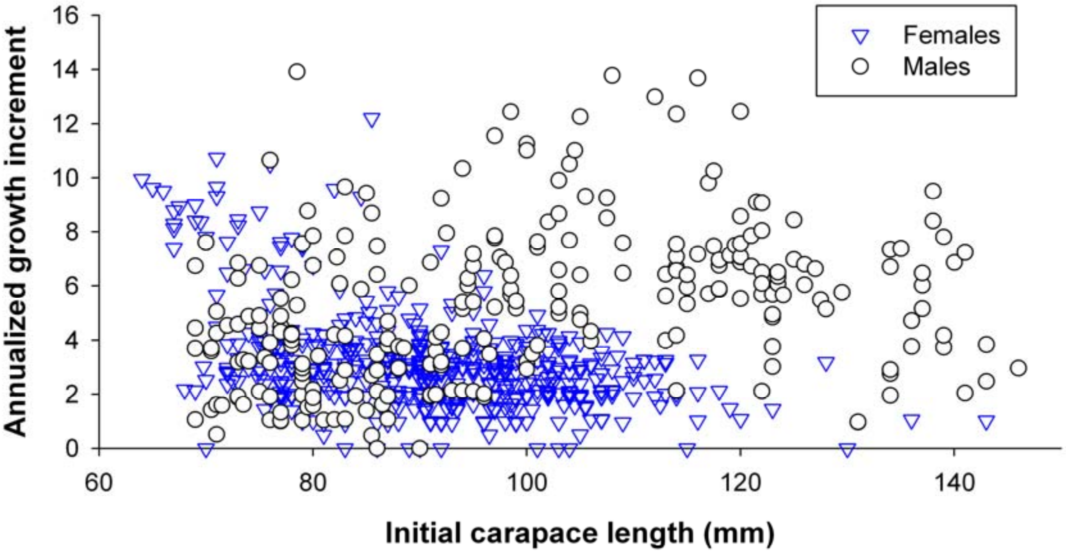


Figure 15. Growth increments for male (circles) and female (triangles) lobsters recaptured during scientific trapping. Individuals reported here were captured within 4 weeks of the anniversary of initial tagging. Growth increments were then annualized by dividing by the decimal years elapsed between tagging and recapture. Because lobsters growth (molting) is periodic and not continuous, animals captured outside the 4 week window are excluded in order to avoid biased growth estimates.

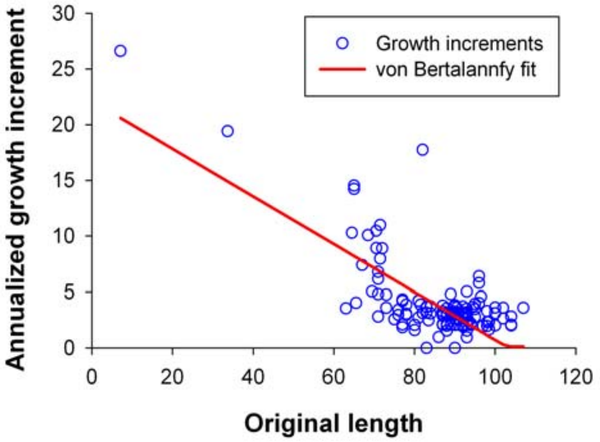


Figure 16. Growth increments (blue circles) for female lobsters recaptured at Morse Point (inside Gull Island MPA, Santa Cruz Island) and the best fit for the von Bertalannfy growth model (red line) written to accomodate growth increment data (see text). The two smallest increments were calculated from Engel (1973).

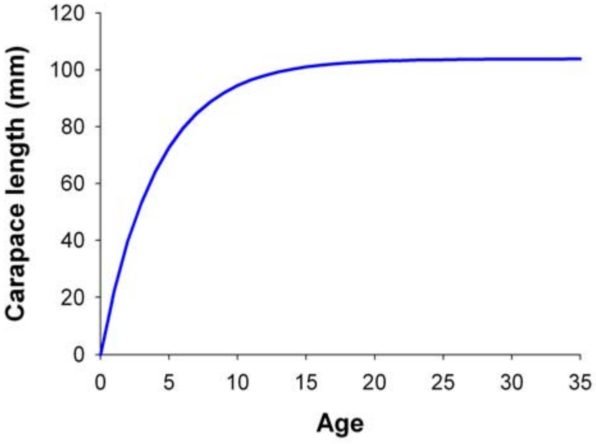


Figure 17. Estimated size at age for female *P. interruptus* tagged and recaptured at Morse Point (Gull Island MPA), and based upon von Bertalannfy growth parameters estimated from growth increment data collected during this study (see text).

Lobster Research Conclusions

As demonstrated in *Sections 1-3*, MPAs at the Santa Barbara Channel Islands have resulted in increased trap yield, body size, and abundance inside vs. outside MPA borders. Because we replicated across multiple reserves, worked with commercial fishermen to sample similar habitats, and performed analyses that control for habitat features that influence localized lobster abundance, we conclude unequivocally that the increases we observed are indeed reserve effects.

Continued spiny lobster CFR at Channel Islands marine reserves may be important because neither the fishery nor lobster populations inside reserves are likely to be at equilibrium. In particular, LEK of senior fishermen at Channel Islands suggests recently increased effort that is likely to intensify as fishermen along the California mainland are displaced by an imminent network of marine reserves. Concern regarding such effort increases in the early 1990's motivated an industry request to DFG to adopt restricted access and a trap limit for the California spiny lobster fishery. Although trap limits were not adopted due to enforcement limitations, restricted access policy has resulted in a limited entry and transferrable permit process for the California spiny lobster fishery. This action on behalf of industry reflects a commitment to long-term sustainability and responsiveness to temporal changes in the fishery.

With regard to temporal changes in lobster populations, research from older reserves in New Zealand suggests that lobster biomass will continue to increase in Channel Island reserves (Kelly et al. 2000, Shears et al. 2006), and this increase may enhance spillover. Due to this temporal dynamism, the relationships among effort distribution, reserve population structure, spatially explicit catch rates, and fishery-reserve interactions should be monitored carefully, and a CFR approach such as we present here is an important tool. The use of CFR to advance reserve assessment suggests that this approach may be successful for broader study and adaptive management of many California nearshore fisheries. The ability to manage adaptively will be critical as humans reach (or surpass) sustainable yields for most fisheries (Hilborn et al. 2003, Mullon et al. 2005) but historic levels of ecosystem services (e.g., fisheries yield) become less certain due to anthropogenic changes to marine ecosystems.

Nearshore Fish Research

Fish Section 1 - Do Channel Islands MPAs influence size structure and CPUE of nearshore fishes?

FS1 - Introduction

In August 2007, the fish component of the research grant began. The program consisted of several modes of research. The first and primary mode of research was the mark and recapture of commercially important nearshore reef fishes in collaboration with commercial fishermen. All fishing for this program was conducted on board one of three commercial vessels – The Ma'alaea, owned and operated by John Colgate, The Janice Anne, owned and operated by Mark Brubaker, and the Sashay, owned and operated by Chris Hoeflinger. We obtained scientific collecting permits from the California Department of Fish and Game (CDFG) to mark and release all commercially important nearshore fishes taken in the live fish fishery of southern California. Our research program set out to evaluate the size structure, CPUE and spatial patterns of population dynamics across the Santa Barbara Channel, inside and outside of no-take Marine Protected Areas (MPAs). We fished inside and outside of six MPAs (Figure 18).

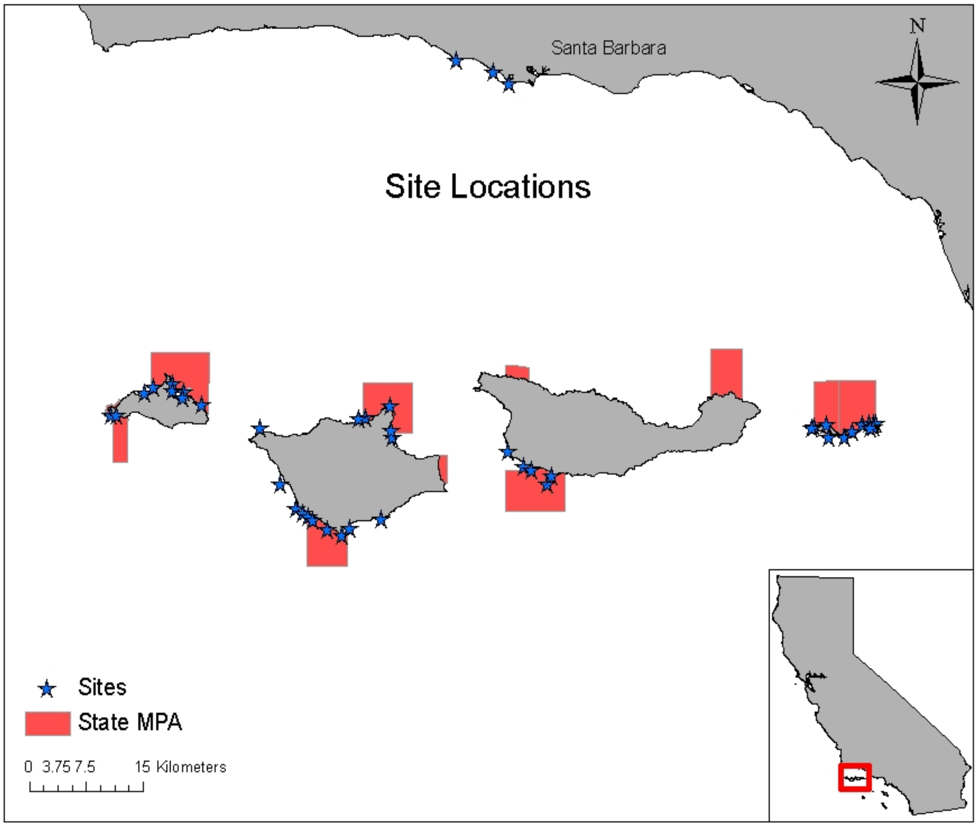


Figure 18. Map of northern Channel Islands study region. Stars indicate sampling locations.

FS1 – Materials and Methods

Our primary means of fishing involved the use of set hook and line gear known as "sticks". Sticks are constructed of five foot sections of ¾ inch weighted PVC pipe. Strung between the two ends of each stick are five leaders with circle hooks, evenly spaced along the length of the pipe. The sticks were consistently baited with thawed market squid (*Loligo* spp). Sticks were baited, and set on the seafloor in shallow rocky reef habitat, and attached to a surface buoy by a thin line. All sticks were set by means of a 14 foot aluminum vessel that allowed for ease of operation in shallow water. Most fishing was conducted in one to ten meters of water depth. Sticks were allowed to soak on the seafloor for approximately one hour. After soaking, the sticks were pulled, and all fish caught were georeferenced, measured (total length), and tagged in the dorsal musculature using individually numbered t-bar tags with reporting information inscribed All tags were inserted under the second spine of the dorsal fin, between the second and third pterygiophore (Figure 19). All fish were released at the location of capture.


Figure 19. grass rockfish with green tag.

Additional fishing operations were conducted using baited trap gear on board commercial vessels. Traps were constructed of vinyl coated wire and measured 18" x 24" x 36". A bait portal was positioned in the center of the trap and two funneled openings measuring 8" in diameter were positioned on either side of the trap. Traps were baited with a combination of market squid and Illex squid (*Illex* spp.), and were allowed to soak for approximately 2 hours before pulling. All fish were similarly tagged and released.

For both trapping and stick fishing, we deployed gear in a stratified non-random design. In consultation with 3 commercial fishermen with a combined 90 years of fishing experience at the Channel Islands, we identified comparable habitats inside and outside of 6 Channel Island MPAs. These included Anacapa Island MPA, Gull Island MPA (Santa Cruz Island), South Point MPA (Santa Rosa Island), Carrington Point MPA (Santa Rosa Island), Harris Point MPA (San Miguel Island), and Judith Rock MPA (San Miguel Island). To effectively sample fish populations inside and outside of each MPA we strategically chose habitats that had historically yielded high abundances, and restricted our sampling to these high quality habitats, much like commercial fishermen do in their daily operations.

For each stick fishing trip, we fished 27 pieces of gear at a time, and allowed each to soak one hour before pulling, re-baiting and resetting. In any given day we were able to set between 108-189 sticks. All sticks were set by commercial fisherman John

Colgate, and all data were recorded by Jono Wilson. At each MPA we fished at least two reefs inside the MPA and at least two sites outside the MPA at differing distances from the border (Figure 18).

Over the course of two years, we spent a total of 48 days on the water for both types of fishing. 36 days were spent stick fishing, and 12 days were spent trap fishing. We caught a total of 27 species of nearshore fishes stick fishing and a total of 33 species trap fishing (Tables 5 and 6). We set 4499 sticks, and 691 traps.

Table 5. Percentage of fish species caught using stick gear.

Species caught on sticks	Number caught	% of catch
Total fish	3262	100
grass rockfish	1514	46.4
cabezon	1282	39.3
gopher rockfish	130	3.9
copper rockfish	46	1.41
California sheephead	36	1.1
lingcod	33	1.01
kelp rockfish	28	0.86
treefish	7	0.22
kelp bass	27	0.83

Table 6. Percentage of fish species caught using trap gear.

Species caught in traps	Number caught	% of catch
Total fish	1096	100
grass rockfish	13	.01
cabezon	162	14.8
gopher rockfish	169	15.4
copper rockfish	62	5.6
California sheephead	164	14.9
lingcod	61	5.5
kelp rockfish	10	.91
brown rockfish	45	4.1
black & yellow rockfish	5	0.46
blue	10	0.91
treefish	27	2.46
kelp bass	146	13.3
black perch	83	7.57

Our first objective was to evaluate the size structure of the population and quantify the Catch per Unit Effort (CPUE) inside and outside of 6 Channel Islands MPAs. Due to logistical constraints, we were unable to sample consistently at Judith Rock MPA so we do not present results for this MPA. For this section we also only present results for stick fishing, and we did not use stick gear at Anacapa Island. Thus we present results from stick fishing for four MPAs: Gull Island, Carrington Point, South

Point and Harris Point MPAs. At each of these sites, our collaborator, commercial fisherman John Colgate took detailed recordings of all fish caught during his commercial fishing operations. Because John used the same gear, and fishing techniques as when he was fishing for science in collaboration with our research program, we present results for the size structure data that includes John's data as well. The CPUE data did not include his commercial data as it is assumed that CPUE may not scale linearly between commercial and scientific fishing ventures.

<u>Grass Rockfish Population Size Structure</u>

Size structure of grass rockfish was recorded inside and outside of four Channel Islands MPAs. Size frequency distributions describe the frequency of individuals in the populations, and when coupled with age at length functions can inform alternative and traditional stock assessment models.

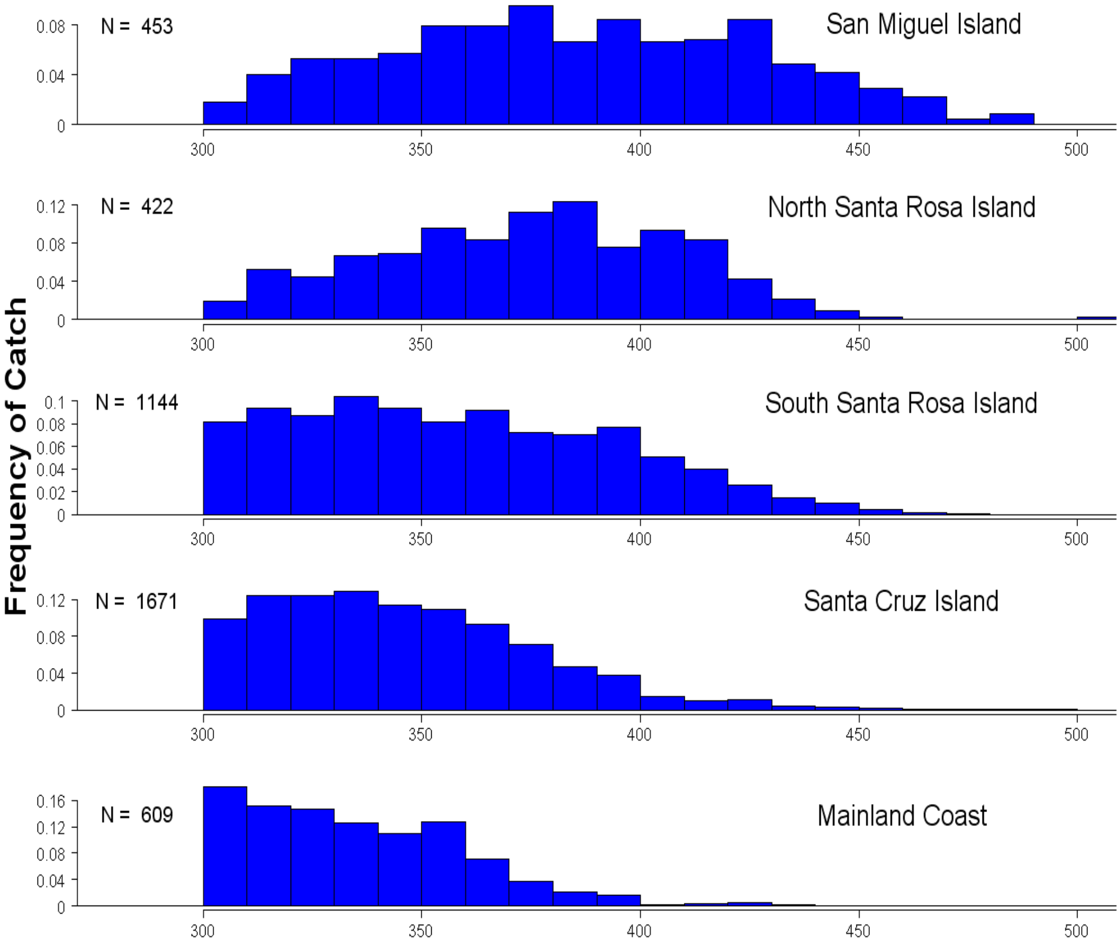


Figure 20. Size frequency distributions for grass rockfish at five separate sites in the Santa Barbara Channel, extending from west to east, including the mainland coast.

Our results show that size frequency distributions were variable across sites along an environmental gradient that naturally extends from the west Santa Barbara Channel to the east. This environmental gradient results in changes in sea surface temperature and primary productivity that contributes to dramatic differences in growth rates, and size frequency distributions (Fig. 20). Figure 20 shows size frequency distributions for all grass rockfish measured inside and outside of MPAs. To determine whether size of fish was driven by longitudinal variation, we ran a generalized linear model with longitude as the predictor variable. Results indicate that grass rockfish mean total length increases from east to west channel (Table 7). The adjusted R-squared is 0.1335 and the F statistic is 168 on 2 and 2128 DF.

Table 7. Results of linear regression for grass rockfish.

Model	Estimate	Error	p-value
Intercept	-9069.307	568.468	<0.001
Longitude	-78.836	4.739	<0.001
Reserve Status	-19.877	1.863	<0.001

We next compared size frequency distributions for all fish measured inside MPAs with all fish measured outside MPAs for all sites (Fig. 21).

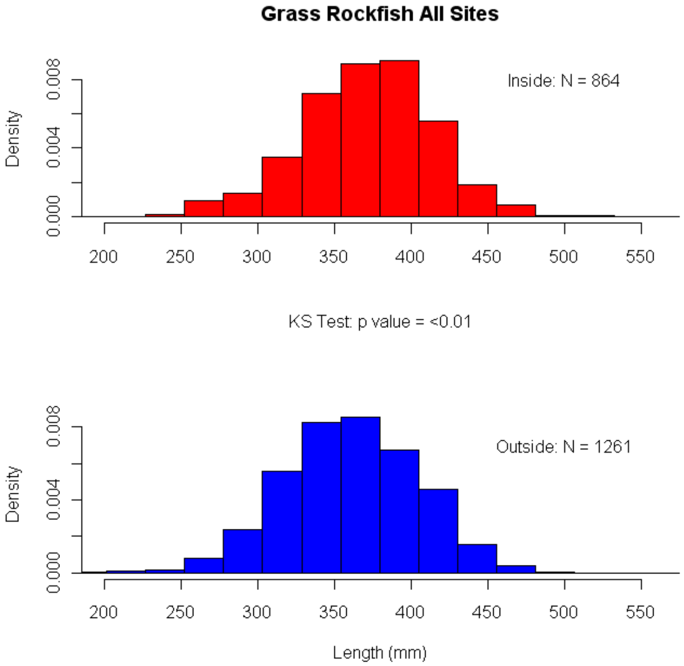


Figure 21. Grass rockfish size structure at all sites

We performed Kolmogorov-Smirnov tests (KS tests) to determine whether the distributions are significantly different from one another. It is apparent from the visual inspection of the plots that the distribution inside reserves is slightly shifted to the right and the KS test indicates significant increases in size structure for those fish measured inside MPAs in the Channel Islands. When size structure plots are examined at individual sites, KS tests support the hypothesis that populations of grass rockfish residing within MPAs are significantly larger than those residing outside of the boundaries (Figures 22-25).

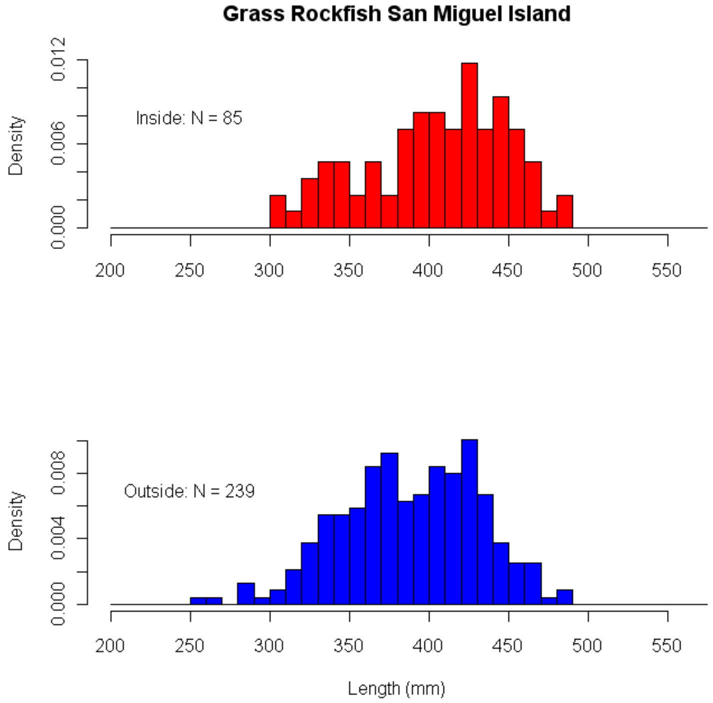


Figure 22. Grass rockfish size structure at Harris Point MPA.

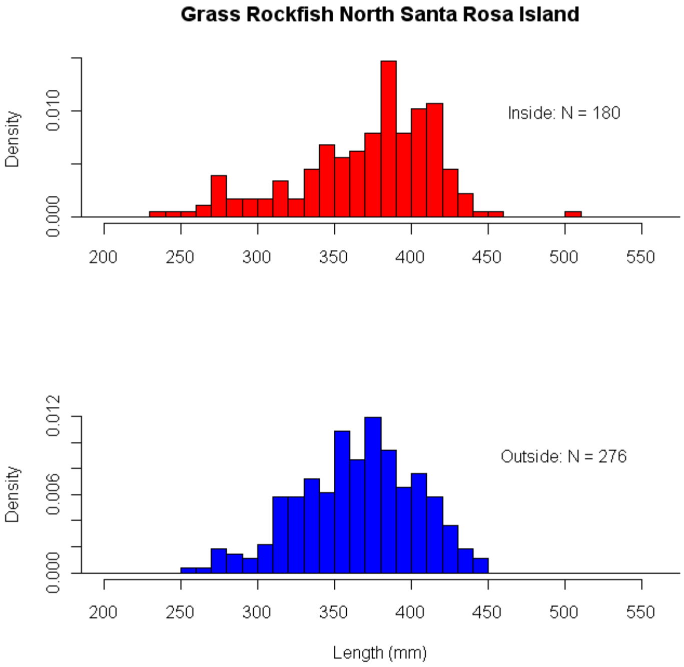


Figure 23. Grass rockfish size structure at Carrington Point MPA.

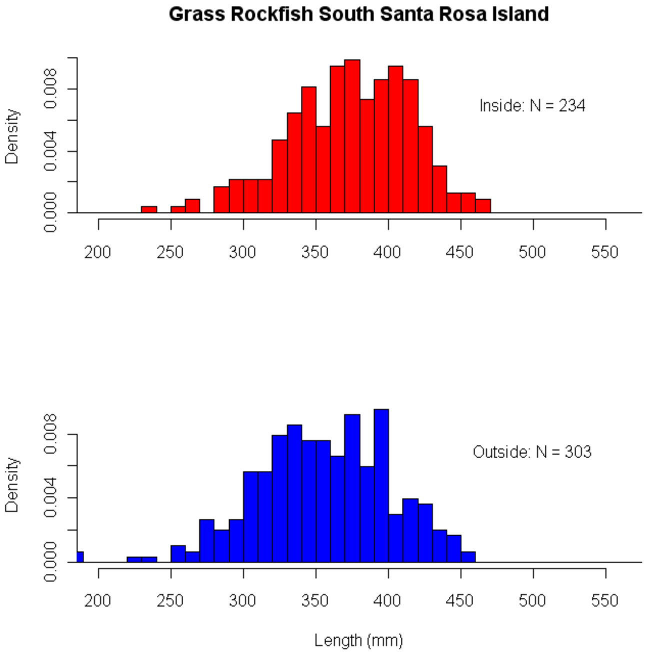


Figure 24. Grass rockfish size structure at South Point MPA.

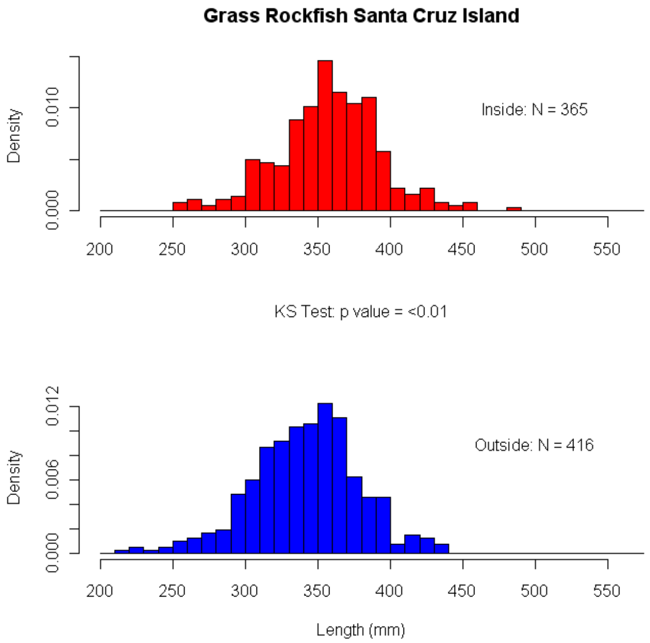


Figure 25. Grass rockfish size structure at Gull Island MPA.

In most cases, the increases in size frequency from outside MPAs to inside MPAs are minimal, albeit significant. We suggest that fishery management regulations enacted in 1999 in the state of California, including a minimum size limit of 12 inches, and subsequent reductions in Total Allowable Catch have reduced fishing pressure to such an extent that the size frequency distributions observed inside and outside of MPAs are only marginally different. Table 8 shows KS test results for each inside-out comparison of size frequency distributions at each site.

Table 8. Results of KS tests for grass rockfish size frequency distributions at each site. The values represent the D statisitic, and the significance levels are as follows: * = p<0.05, ** = p<0.01, *** = p<0.001.

Sites	All Sites Out	SMI Out	N. SRI Out	S. SRI Out	SCI Out
All Sites In	0.1193***				
SMI In		0.1824*			
N. SRI In			.1775**		
S. SRI In				0.1903***	
SCI In					0.205***

Grass Rockfish CPUE

Grass rockfish catch per unit effort (CPUE) was recorded as the number of fish per stick (5 hooks). We recorded CPUE for each site, inside and outside of the MPAs. To determine whether there was a significant effect of MPAs on the CPUE of grass rockfish across all sites, we fit a poisson generalized linear model (GLM) for a full model that included MPA categorical variable, and the site. We then dropped site from the model and performed a chi squared test to compare the two models with an analysis of deviance. Results indicate that there is a significant effect of MPAs on the CPUE of grass rockfish (Table 9).

Table 9. Results of the chi squared analysis of deviance test to determine the effects of an MPA on CPUE for grass rockfish.

Site	Df	Deviance	p
Full model vs MPA only	1	-5	.025

To determine site specific differences in CPUE inside and outside of MPAs, we performed individual poisson GLMs with MPA as the categorical variable. Results were variable for each site. San Miguel Island had greater CPUE inside than outside, but differences were insignificant. North Santa Rosa Island and South Santa Rosa Island had significantly larger CPUE inside than outside. Santa Cruz Island had lower CPUE inside than outside, but results were insignificant (Figure 26).

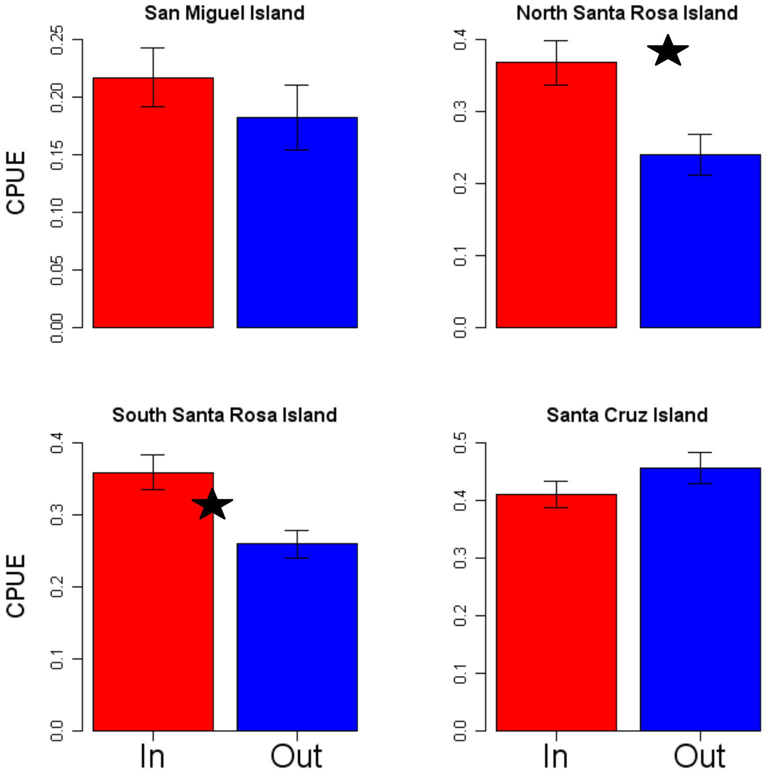


Figure 26. CPUE of grass rockfish. Stars indicate significant differences ($p < 0.05$).

Cabezon Population Size Structure

Size structure of cabezon was recorded inside and outside of four Channel Islands MPAs. Size frequency distributions describe the frequency of individuals in the populations, and when coupled with age at length functions can inform alternative and traditional stock assessment models. Our results show that size frequency distributions were variable across sites along an environmental gradient that naturally extends from the west Santa Barbara Channel to the east (Fig. 27).

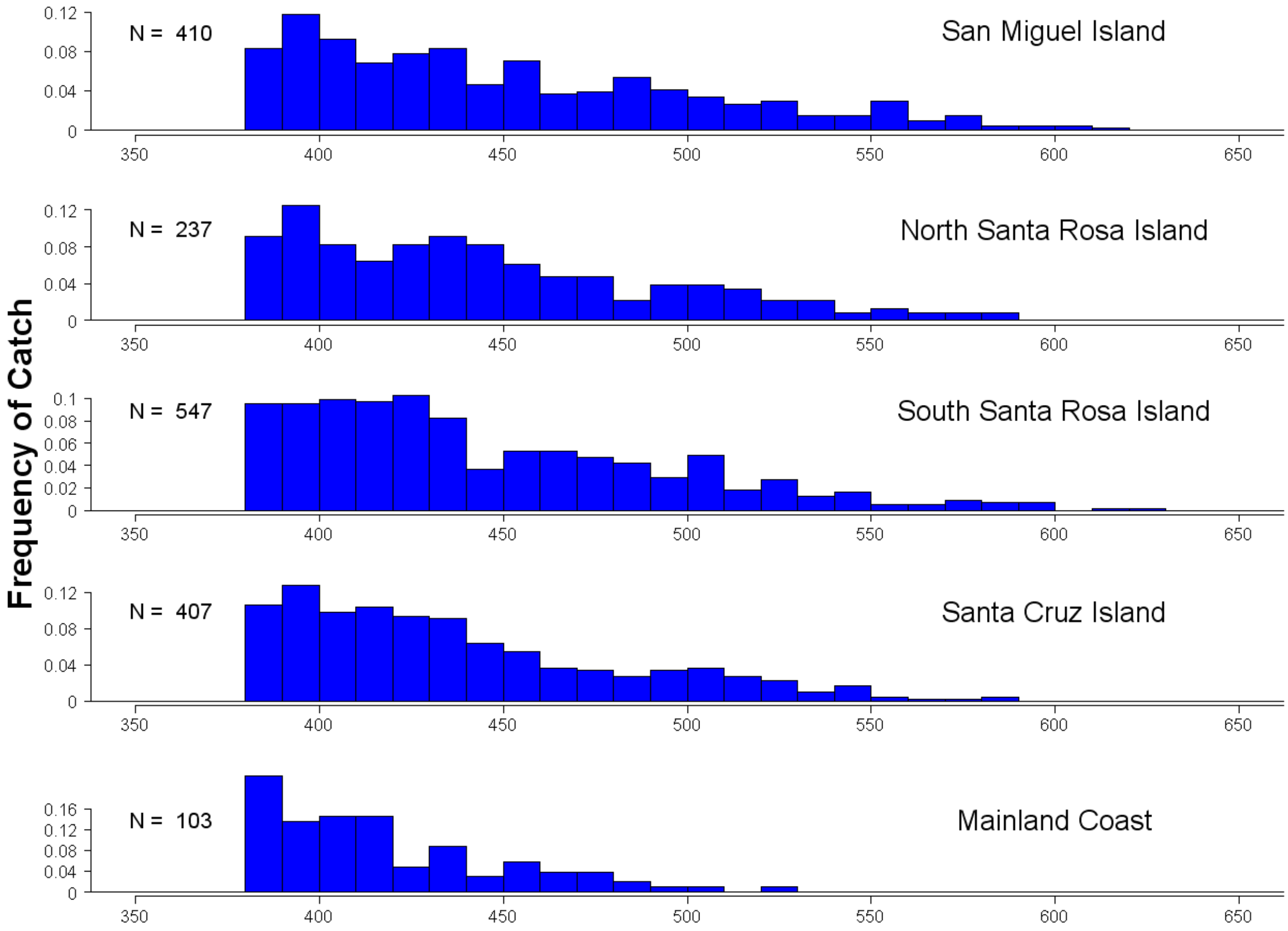


Figure 27. Size frequency distributions of cabezon at five separate sites in the Santa Barbara Channel, extending from west to east, including the mainland coast.

To determine whether size of fish was driven by longitudinal variation, we ran a generalized linear model with longitude as the predictor variable. Results indicate that grass rockfish mean total length increases from east to west channel (Table 10). The adjusted R-squared is 0.1335 and the F statistic is 168 on 2 and 2128 DF.

Table 10. Results of linear regression for cabezon.

Model	Estimate	Error	p-value
Intercept	-9069.307	568.468	<0.001
Longitude	-78.836	4.739	<0.001
Reserve Status	-19.877	1.863	<0.001

To examine size frequency distributions at individual sites, we plotted distributions inside and outside of each MPA across all sites (Fig. 28), and then at each site (Figs. 28-32), and ran KS tests to determine if there were differences in population size structure inside and outside of MPAs (Table 11). Cabezon size structure was greater at all reserve sites save for Santa Cruz Island (Table 11).

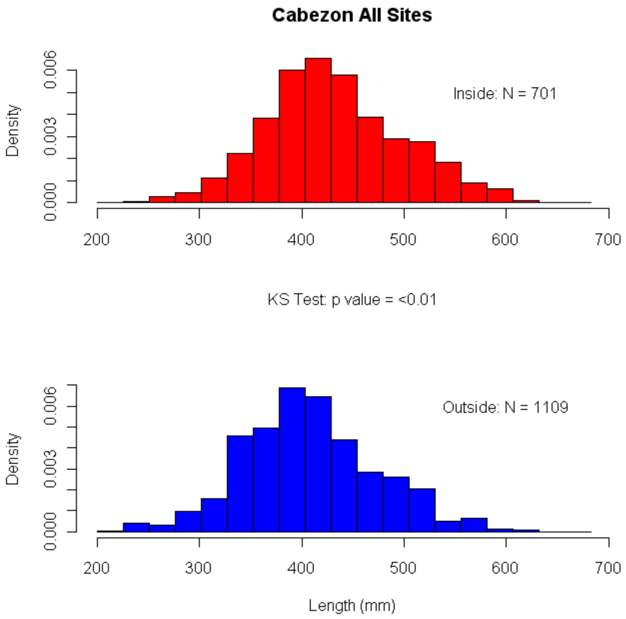


Figure 28. Cabezon size structure across all sites.

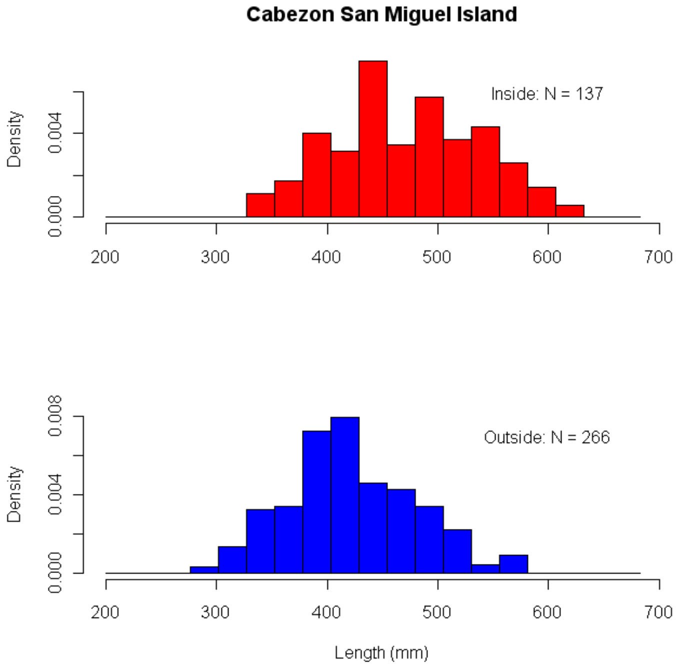


Figure 29. Cabezon size structure at Harris Point MPA.

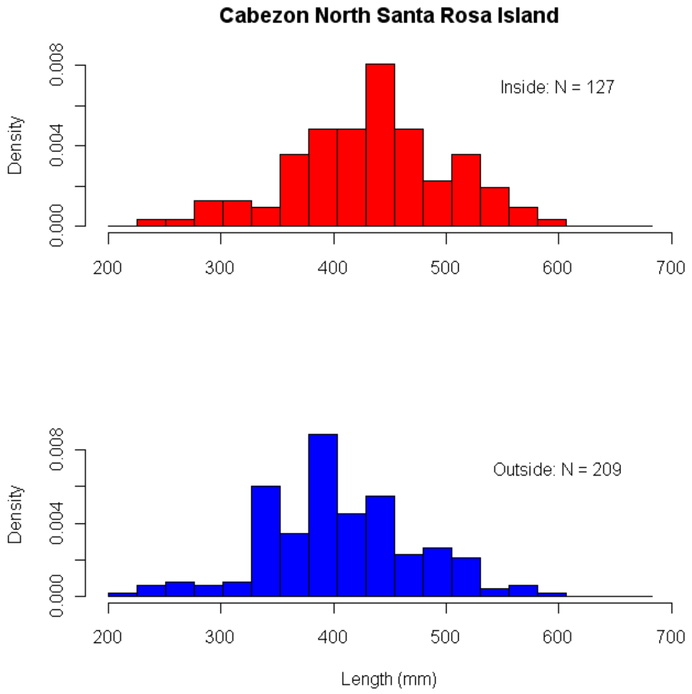


Figure 30. Cabezon size structure at Carrington Point MPA.

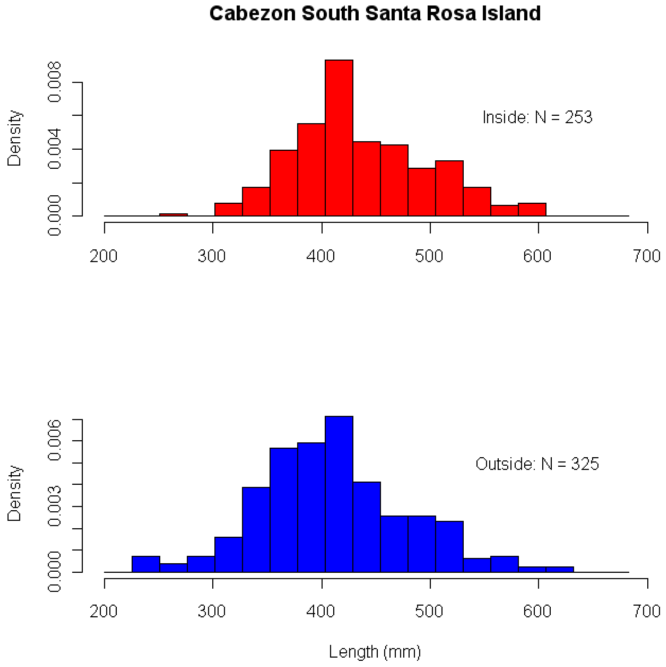


Figure 31. Cabezon size structure at South Point MPA.

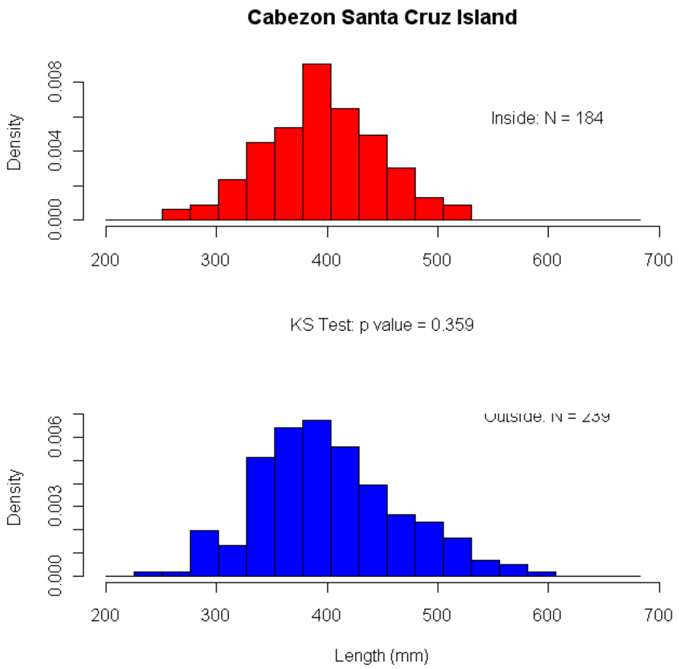


Figure 32. Cabezon size structure at Gull Island MPA.

Table 11. Results of KS tests for cabezon size frequency distributions at each site.

Sites	All Sites Out	SMI Out	N. SRI Out	S. SRI Out	SCI Out
All Sites In	0.159***				
SMI In		0.3615***			
N. SRI In			0.2355***		
S. SRI In				0.1837***	
SCI In					0.0909

Cabezon CPUE

To determine differences in CPUE for cabezon inside and outside of each reserve location, we also performed an analysis of deviance test. Results indicate that there is also a significant reserve effect on CPUE for cabezon (Table 12). We then performed individual poisson GLMs for CPUE at each site. Cabezon CPUE was also variable across sites. CPUE was significantly higher inside the MPAs at San Miguel Island and North Santa Rosa Island, but insignificant at South Santa Rosa Island and Santa Cruz Island (Fig. 33).

Table 12. Results of the chi squared analysis of deviance test to determine the effects of an MPA on CPUE for grass rockfish.

Site	Df	Deviance	p
Full model vs MPA only	1	-4.5	.028

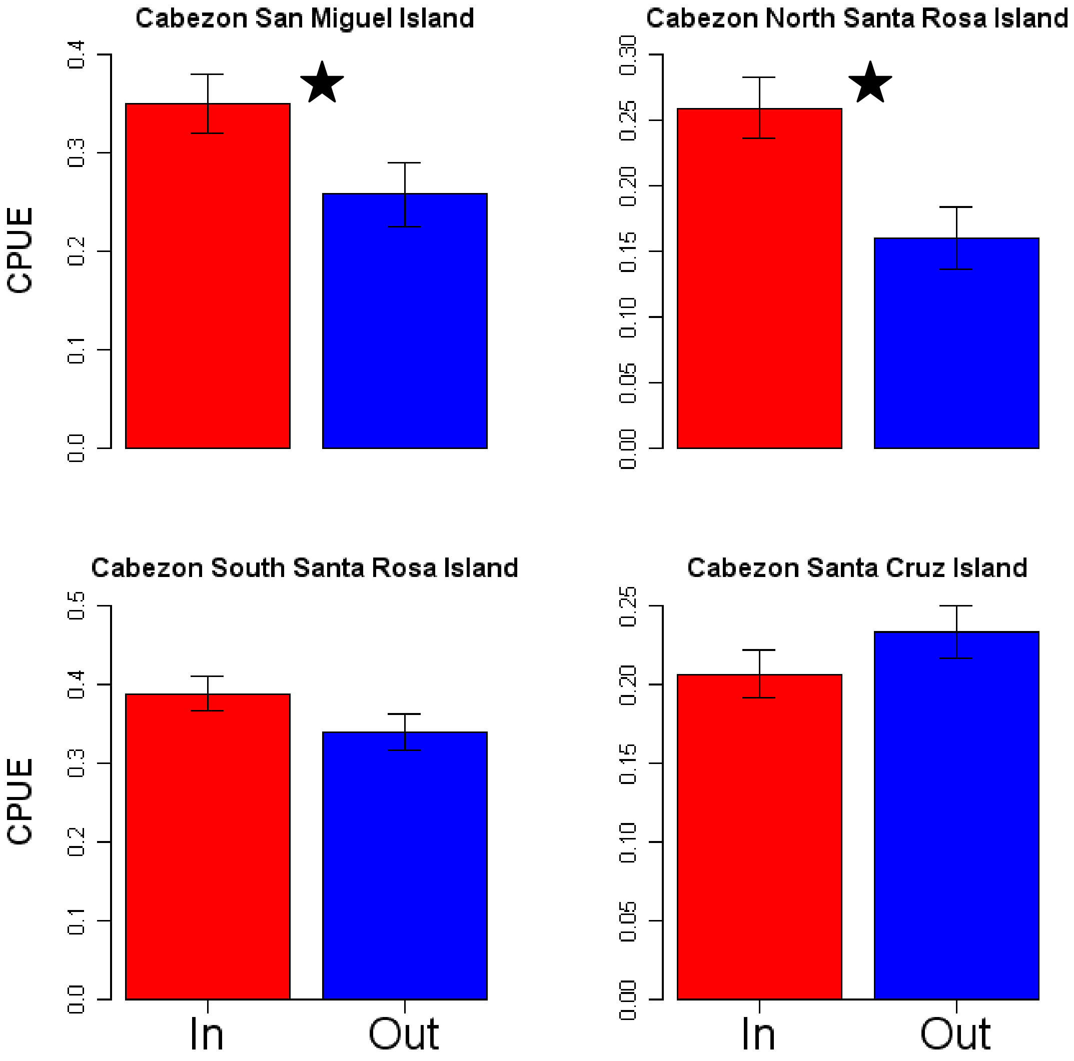


Figure 33. CPUE of cabezon. Stars indicate significant differences ($p < 0.05$).

Fish Section 2 - Grass Rockfish (*Sebastes rastrelliger*) Life History

FS2 – Growth Rates of grass rockfish

We aged 173 total grass rockfish from three different sites in the Santa Barbara Channel; San Miguel Island, East Santa Cruz Island, and the mainland coast, west of Santa Barbara. All *S. rastrelliger* samples were analyzed by Fish Aging Services PTY of Port Arlington, Australia. All otoliths were aged by reading transverse otolith sections. Sagittal otoliths were embedded in rows of five in blocks of clear casting resin ensuring that the primordium of each otolith is line. Each row of otoliths was sectioned on the transverse axis. Four sections, approximately 300µm thick, were cut through the otoliths centers with a modified high speed gem-cutting saw with a 250µm thick diamond impregnated blade. Four sections were taken to ensure that the primodium of the otolith was taken. Sections from each block were cleaned, rinsed in alcohol, dried and mounted on glass microscope slides (50 x 76 mm) under glass cover slips using resin. Using these methods, 5 otoliths were prepared for ageing at one time.

The age of a sample was determined by counting the sequence of alternating translucent and opaque zones from the primordium (the biological centre of the otolith) to the otolith edge. Otoliths were read using research grade Leica stereo dissecting microscopes (MZ80's). Sectioned otoliths were examined using transmitted light. Each section of the otolith was inspected, and the section with the clearest increments was chosen for ageing. This was usually, but not necessarily the section closest to the primordium. To avoid the introduction of bias, the magnification at which otoliths were read remained constant. Occasionally, some samples were examined at higher magnifications to resolve the structure of on the edge of the sample. To avoid the potential for biasing age estimates, all counts were made without knowledge of fish size, sex and location. Once age estimates were completed, the ageing data were combined with the biological information for subsequent analyses. These analyses included generation of a von Bertalanffy growth curve, examination of the otolith weight age relationship and age composition.

Repeated readings of otoliths provide measures of intra-reader and inter-reader variability. The purpose of the re-reading otoliths was to provide an indication of error associated with the estimates, not an agreed age. Re-reads do not validate the assigned ages but provide an indication of magnitude of the error to be expected within a set of age estimates. These differences are due to variations in interpretation of the otolith zones. Beamish and Fournier (1981) developed an index of average percent error (IAPE), which has become a common method for quantifying this variation. A sub-sample of more than 25% was re-aged for this species and the IAPE calculated. Age bias plots, age difference tables and regression analysis were also examined to investigate precision.

A bootstrap technique was applied to individual error estimates to establish confidence intervals around the IAPE. Five thousand data sets (each the same size as the original) was generated with replacement from the original repeat reading data set, and an IAPE calculated for each group. Confidence intervals around the bootstrapped IAPE were determined by using 5% and 95% quantiles from the array of bootstrapped IAPE's.

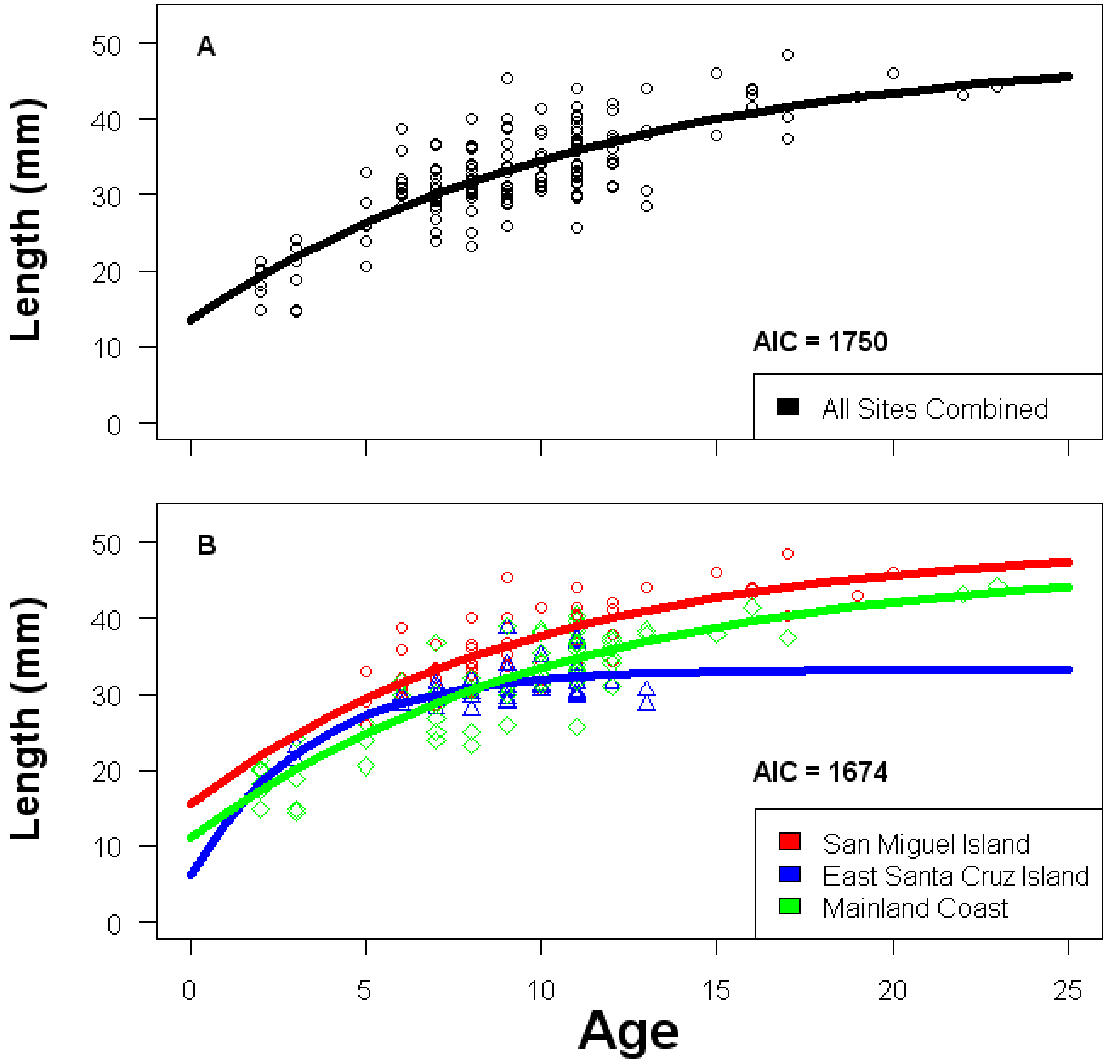


Figure 34**.** MLE estimate of von Bertalanffy growth curves. Figure A represents all fish combined, whereas figure B shows separate growth curves for each of the three sites. T0 is allowed to be estimated as a free parameter.

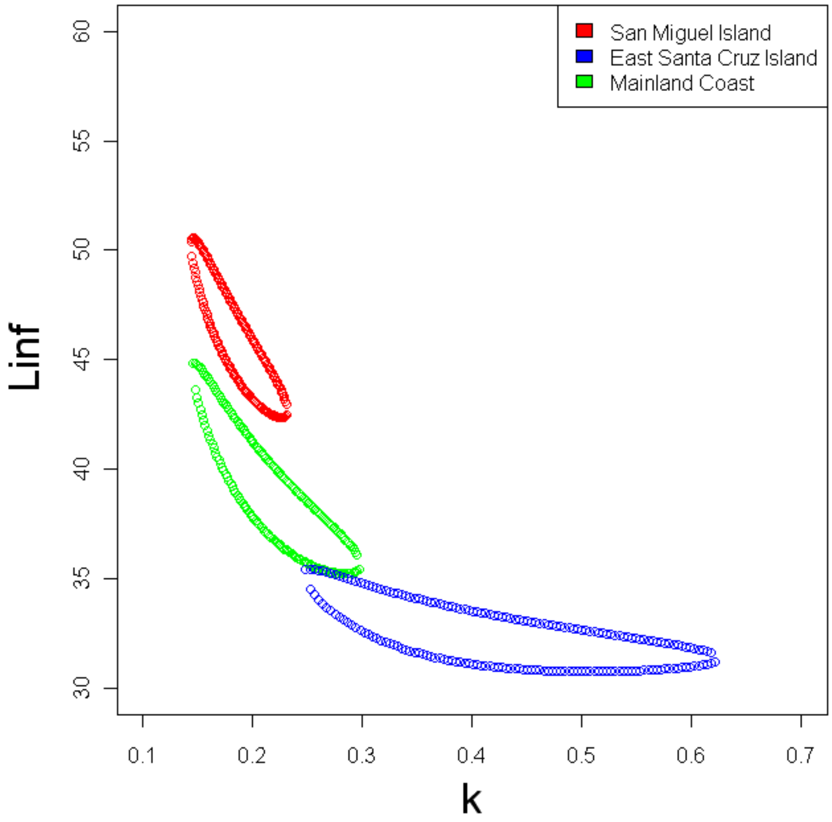


Figure 35. Plots displaying the 95% confidence intervals around the relationship of k and L infinity in the von Bertalanffy growth equation.

MLE

Site	K	Linfinity	To
All Sites Combined	0.199	40.9	0
San Miguel Island	0.181	46.1	0
East Santa Cruz Island	0.374	32.5	0
Mainland Coast	0.130	39.0	0

NLS

Site	K	Linfinity	To
All Sites Combined			0
San Miguel Island	.181	46.1	0
East Santa Cruz Island	.374	32.5	0
Mainland Coast	.208	39.0	0

Table 12. von Bertalanffy growth rate parameters for two estimation methods. MLE = Maximum Likelihood Estimation with a log normal error structure. NLS= Non linear least squares estimation. Parameter estimates assume T0 = 0.

MLE

Site	K	Linfinity	To
All Sites Combined	0.088	49.6	-3.63
San Miguel Island	0.10	50.3	-3.76
East Santa Cruz Island	0.29	33.2	-0.68
Mainland Coast	0.095	47.5	-2.84

NLS

Site	K	Linfinity	To
All Sites Combined	0.088	49.7	-3.67
San Miguel Island	0.095	51.0	-4.15
East Santa Cruz Island	0.344	32.76	-0.26
Mainland Coast	.088	48.75	-3.04

Table 13. Grass rockfish von Bertalanffy growth rate parameters for two estimation methods. MLE = Maximum Likelihood Estimation with a log normal error structure. NLS= Non linear least squares estimation. Parameter estimates assume t0 is a free parameter.

Results from the growth curve/aging analyses reveal strong site differences between three environmentally distinct regions of the channel. We calculated Akaike Information Criterion scores for a single model as well as three separate models for each site. AIC values were lowest when von Bertalanffy growth rates were calculated independently (Figure 34), suggesting a better model fit for three separate growth curves at each site. We also determined 95% confidence intervals around the relationship between k and L infinity (Kimura 1980); two parameters that represent the growth rate and mean asymptotic size, respectively, in the von Bertalanffy growth equation. Confidence intervals were non-overlapping (Figure 135), further supporting site differences in growth rates. Mean asymptotic size is greatest at San Miguel Island, the coldest, most nutrient rich site. Mean asymptotic size is smallest in the east Channel, where water temperatures are coolest.

Differences in growth rates between sites may pose interesting complexities for traditional stock assessment approaches that assume a single homogenous stock. Spatial based management approaches that account for demographic variability may afford more appropriate yields while maintaining sustainability criteria. To further explore the issue of site specific variability we examined differences in the length-weight relationship and differences in reproductive maturity.

FS2 – Length-Weight Relationship for grass rockfish

We measured total length and weight to nearest gram for 171 grass rockfish. Although, there were minor differences between sites, none of the results were statistically different, and therefore we present results of the length-weight relationship as a single regression equation for all sites with the form:

Weight = 0.0199 x Length^2.974

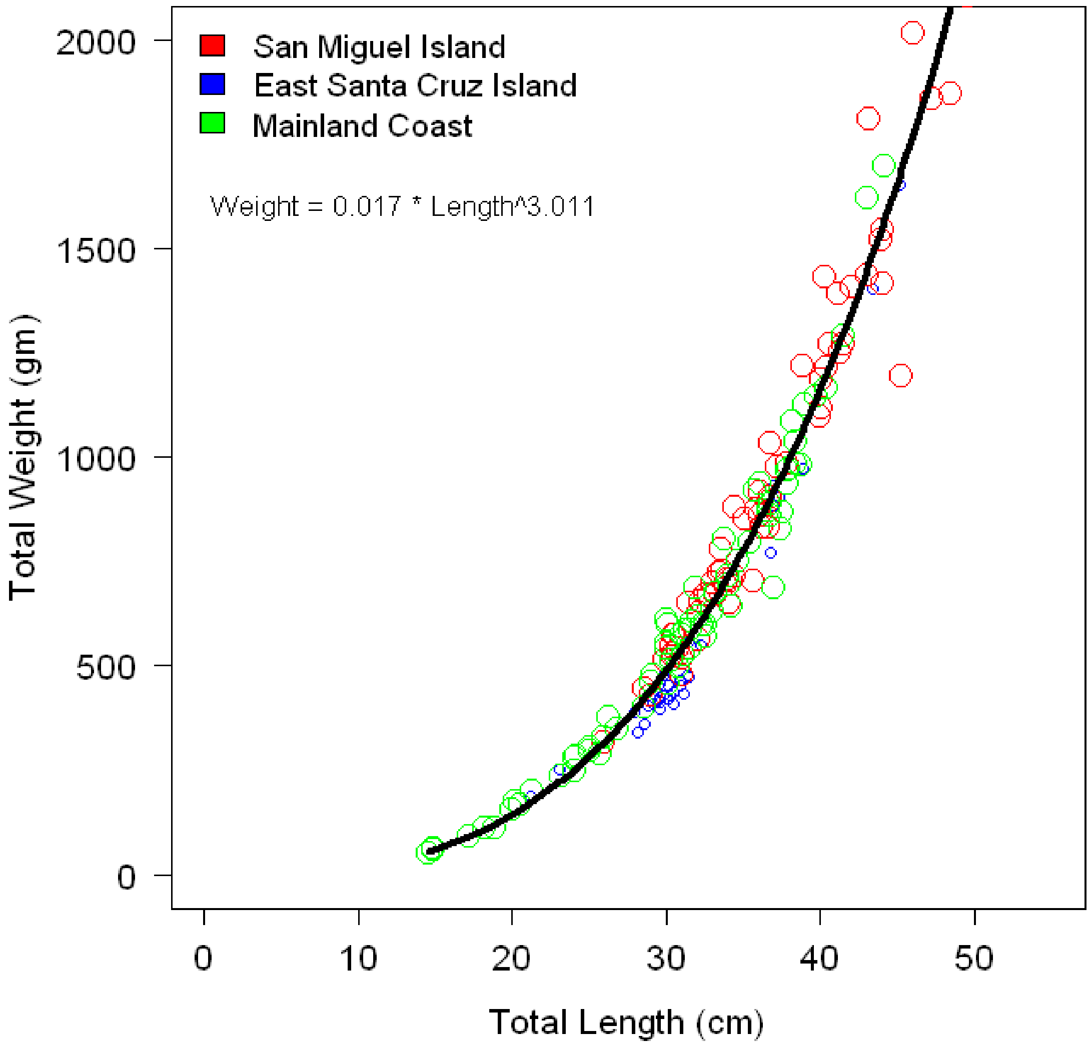


Figure 36. Length-weight data and best fit line for grass rockfish at three sites.

FS2 - Reproductive Maturity_of grass rockfish

We estimated length and age at first maturity by classifying gonads as immature or mature based on criteria given in Westrheim (1975), Gunderson (1977), and Love and Westphal (1981). It is difficult to distinguish between immature and mature resting-stage females during the non reproductive season. Therefore, we only used fish captured during the reproductive season for the length-maturity analyses. We separated fish by site, and fit a logistic model to the data using maximum likelihood techniques.

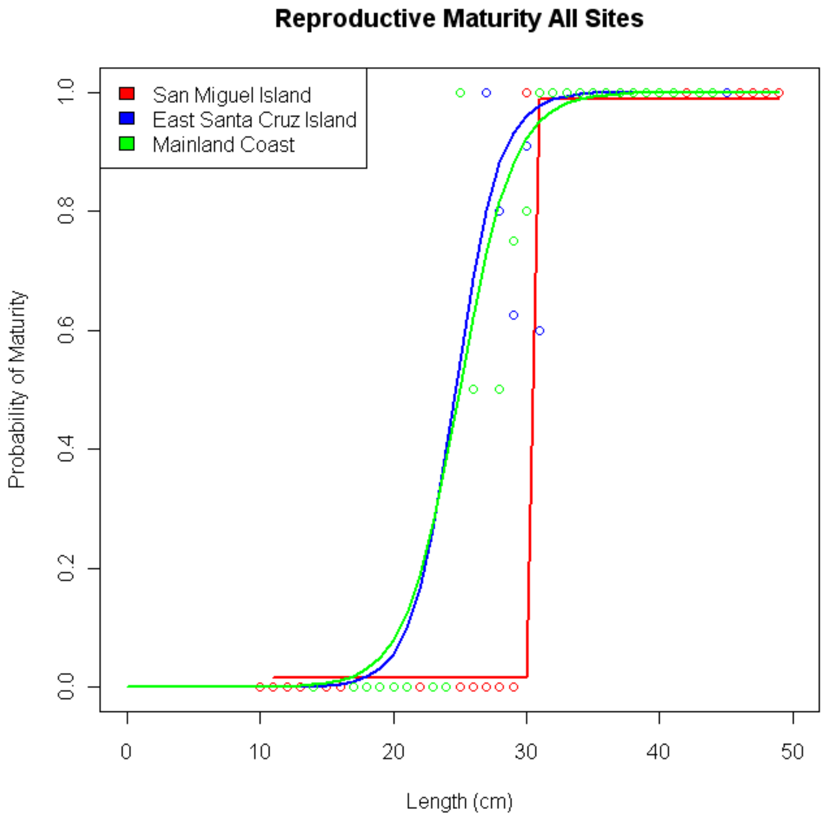


Figure 37. Length at reproductive maturity for grass rockfish.

FS2 - Fecundity of grass rockfish

We calculated egg production for 6 reproductively mature females from the Santa Barbara Channel. In order to effectively estimate the number of eggs, we ensured that only fish at the peak of reproduction were included in this analysis. To estimate reproductive status, we followed guidelines developed in Caillet et al (1986) to stage reproductive maturity. Gonads were removed, weighed and preserved in formalin for a period of three months. To count eggs, a subsample of the eggs was removed, formalin was decanted, and eggs were rinsed in fresh water and allowed to dry.

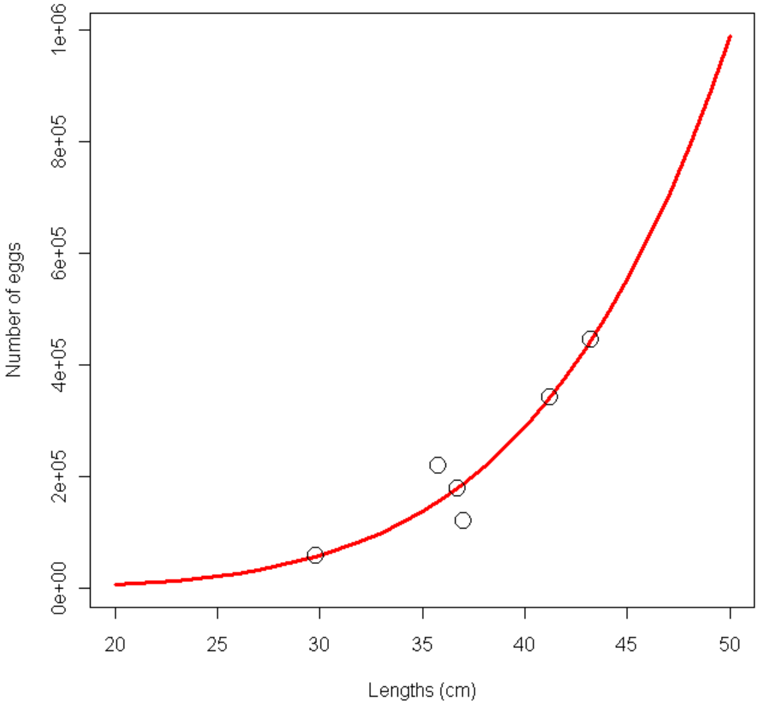


Figure 38. Fecundity of grass rockfish.

Fish Section 3 - Development of an MPA-Based Management Strategy

FS3 - Introduction

Classical approaches to fisheries stock assessment rely on methods that are not conducive to managing data-poor stocks. Moreover, many nearshore rocky reef species exhibit spatial variation in harvest pressure and demographic rates, further limiting traditional stock assessment approaches. Novel management strategies to overcome data limitations and account for spatial variability are needed. With the ever-increasing implementation of no-take marine protected areas (MPAs), there is great potential for improving decision making in management through comparisons of fished populations with populations in MPAs at spatially explicit scales.

We developed a management strategy that uses a combination of data-based indicators sampled inside and outside of MPAs as well as model-based reference points for data-poor, sedentary nearshore species. We performed a management strategy evaluation of this MPA-based decision tree model for a hypothetical population of grass rockfish Sebastes rastrelliger in California. We introduced process, observation, and model uncertainty in numerous scenarios and compared these scenarios with the precautionary approach currently used to manage data-poor species. Our model consistently improved total catches while maintaining the biomass and spawning potential ratio at levels well within acceptable thresholds of management. We suggest further exploration of this MPA-based management approach, and we outline a collaborative research program in the California Channel Islands that may well be suited for testing an experimental management procedure.

FS3 - Materials and Methods

Decision Tree Model

The MPA-based decision tree management strategy we develop here uses a combination of empirically derived CPUE and size-based metrics inside and out of MPAs as well as model based reference points to set sustainable harvest levels. The model also requires basic biological information such as an age-length relationship (e.g. von Bertalanffy), size/age at reproductive maturity, length-fecundity relationship, and an estimate of natural mortality (M). The basis of the management strategy was developed by Froese (2004) who suggested that sustainable management of fisheries resources may be achieved by assuring adequate representation of three size classes in the harvest: recruits, prime, and old individuals. "Recruits" refer to the smallest size bin in the catch representing individuals that have not yet reproduced to those individuals that have been reproductively mature for 1-3 years. The "prime" sized bin represents those individuals in the center of the size distribution (around the mode), while the "old fish" bin represents the oldest individuals, known as mega-spawners.

The decision tree has four successive levels that each compares data-based performance indicators with predetermined reference points. Adjustments to the previous years' TAC are made based on these comparisons (Figure 40). The following sections provide an overview of each of the four levels of the decision tree and a description of the associated equations used for calculating the necessary adjustment to TAC.

Level 1

Level 1 of the decision tree sets an initial TAC using a modified slope to target rule. The slope to target rule is an algorithm that adjusts a current TAC up or down based on the slope between the present measured CPUE of prime sized fish ($\mathrm{CPUE}_{\mathrm{prime}}$) and a desired fraction of $\mathrm{CPUE}_{\mathrm{prime}}$ observed within an MPA, given an acceptable time frame to achieve the desired level. When the present $\mathrm{CPUE}_{\mathrm{prime}}$ is below the desired level, then the subsequent setting of TAC will decrease. When the present $\mathrm{CPUE}_{\mathrm{prime}}$ is above a desired state, then the subsequent TAC will increase.

To account for uncertainty surrounding CPUE estimates, an exponentially weighted 5-year moving average of $\mathrm{CPUE}_{\mathrm{prime}}$ is used in both the fished area (A) and the MPA population (B). To calculate the TAC using the modified slope to target rule, we first determined the optimal target reference point for $\mathrm{CPUE}_{\mathrm{prime}}$ that would achieve $\mathrm{SPR}_{0.4}$ while simultaneously maximizing catch. We calculated the value to be 40% of the $\mathrm{CPUE}_{\mathrm{prime}}$ found inside the MPA (see *Decision Tree Parameter Optimization* for details). The use of this reference level however, is not appropriate until the MPA population reaches an approximation of carrying capacity. Therefore, for the phase in period, we use the following equation to calculate the appropriate slope to target (*Vt*):

(1) $$V_t = (A_t - \Theta_t B_t)/d$$

where d is the time frame to return the stock to the desired level
Θ_t is:

(2) $$\Theta_{t+1} = \begin{cases} \Theta_t - \dfrac{0.6}{MGT} & \textit{for } t = 2 \textit{ to } t_K \\ 0.4 & \textit{for } t > t_K \end{cases}$$

where $\Theta_{t=1}$ is 1, t_K is the time at which our simulated age-structured population reaches 90% of the carrying capacity under no harvest; roughly equal to the mean generation time (MGT) of this hypothetical population (10 years).

We use the slope to target calculation to set the TAC in level 1 with the following equation:

(3) $$\mathrm{TAC}_{t+1} = \mathrm{TAC}_t * [1 + k(V_t)]$$

where k is the responsiveness factor determining how extreme the adjustment of the TAC will be relative to the slope to target value (V_t).
When V_t is positive, this indicates the $\mathrm{CPUE}_{\mathrm{prime}}$ in the fished population is above the target value and therefore the TAC for the following year will increase. When V_t is negative, $\mathrm{CPUE}_{\mathrm{prime}}$ in the fished population is below the target value and the TAC will decrease in the following year.

Level 2

In Level 2, the trend in CPUE of the prime size individuals ($\mathrm{CPUE}_{\mathrm{prime}}$) over a five year period is used to determine whether catches are increasing, stable, or falling. Time-averaged CPUE is used as an initial estimation of whether the population is increasing or decreasing. An exponentially weighted moving average over five years is used again to evaluate whether the change from the previous year to the current year falls outside of the five year average. If the annual change is greater than 5% above the average, then the trend is increasing. If the annual change is below 5% of the average then the trend is decreasing (Table 15), otherwise it is stable.

Level 3

In Level 3, the relationship between the proportion of old fish ($\text{Proportion}_{\text{old}}$) and the CPUE of old fish (CPUE_{old}) in the fished population is compared with the CPUE_{old} and $\text{Proportion}_{\text{old}}$ derived from per recruit models. Level 3 is intended to inform managers if catches of old fish are increasing or decreasing, and determine whether the trend results from a change in the selectivity of older size classes to the gear, or is due to recruitment pulses that altered (i.e., reduced) the true proportion of old fish in the population.

In our simulated case study, our objective was to maintain SSBR levels at 40% of unfished conditions ($\text{SPR}_{0.4}$). Values for CPUE_{old} and Proportion_old that result in $\text{SPR}_{0.4}$ conditions were derived from per recruit modeling which required basic biological information and an estimate of natural mortality. The proportions of old fish in the harvested population as well as in the modeled population were calculated relative to the proportion of the other size classes (recruits and prime) in these respective populations, and were therefore scale less. CPUE on the other hand is an absolute value and therefore the data based estimate of CPUE may not scale with the modeled CPUE value at $\text{SPR}_{0.4}$. To reconcile the scaling problem, a number of options are available to managers including the use of historical fishermen knowledge, and data from inside existing MPAs. We assumed that the maximum attainable CPUE in a real population is equivalent to the maximum attainable CPUE in the modeled population. This assumption made it possible to scale the estimate of CPUE_{old} that results in $\text{SPR}_{0.4}$, for comparison with the data-based estimates.

Level 4

Level 4 provides an estimate of whether recruitment overfishing is occurring by assessing whether the CPUE of young fish ($\text{CPUE}_{\text{recruit}}$) are above or below desired reference levels. Depending on the outcome in Levels 2 and 3, the analysis in Level 4 compare the $\text{CPUE}_{\text{recruit}}$ to estimated unfished levels of $\text{CPUE}_{\text{recruit}}$ calculated through per recruit modeling, or, alternatively whether the pattern of $\text{CPUE}_{\text{recruit}}$ over the previous few years has been rising, stable, or falling. In the former scenario we determine whether $\text{CPUE}_{\text{recruit}}$ is significantly below unfished conditions by setting a threshold at 80% of unfished levels. In the latter scenario we determine whether the trend is rising, stable, or falling based on whether the annual change in $\text{CPUE}_{\text{recruit}}$ was greater or less than 10% of the 5 year moving average.

Management Strategy Evaluation

To conduct the MSE, we first built an age structured population dynamics model specific to grass rockfish based on published data (Love and Johnson 1999). The population was then “sampled” via a simulated collaborative data collection program, and associated performance indicators were calculated. We used these performance indicators in the decision tree model to calculate the appropriate Total Allowable Catch (TAC). The TAC is then harvested from the simulated population in the following year and the population is updated via a series of dynamic equations (Figure 39). This cycle is repeated for 30 years, and uncertainty is introduced into the model via process, observation, and sampling error using Monte Carlo simulation (Cooke 1999; Smith et al. 1999). We developed an MSE specific to the decision tree that addresses four objectives of fisheries management. First, we wanted to calculate the probability that the model can maintain biomass and SPR above the limit reference levels of 10% and 20% of virgin levels under multiple uncertainty. Second, we wanted to test whether yield could be increased relative to yield under the present management strategy while maintaining biomass and SPR at acceptable levels of sustainability. Third, we wanted to test whether the model could maintain biomass and SPR and allow increased yield while also reducing year to year variability in catch. Finally, we wanted to determine if the cost of management could be reduced by comparing model outcomes conducted annually versus once every three years.

Operating Model

We built two age-structured population models specific to grass rockfish with 19 age classes and a plus group (Punt and Hilborn 1997; Appendix A), representing two distinct populations with similar life history characteristics and environmental pressures. The models were parameterized such that they could be subjected to process and observation error. Life history information such as growth rates, maturity ogives, and fecundity ogives were based on empirical data (Love and Johnson 1999). Natural mortality (m) was assumed to be 0.2, typical of most west coast rockfish stock assessments (pers. comm. Alec MacCall). Selectivity of fishing gear took on a logistic form with knife edged selectivity occurring at the minimum size limit, similar to other species in the nearshore finfish complex (Alonzo 2004; Key et al. 2005). Recruitment was modeled using a Beverton Holt stock recruitment function, with steepness (h) of 0.75 and subject to year to year recruitment variation (σ^w) that was the same for both populations. The representative equations for the population dynamics model are listed in Appendix A and the associated parameter values are provided in Table 14.

Temporal patterns in the operating models were chosen to reflect the conditions observed in the live fish fishery at the northern Channel Islands in the Santa Barbara Channel, CA, USA, from 1984 to the present day, encapsulating the growth, peak, and decline of the commercial fishery. Significant management measures were incorporated into the model, including a minimum size limit regulation enacted in 1999, and the establishment of a network of MPAs in 2003. For the first fifteen years of the simulated fishery, we set harvest pressure equal to $3*F_{max}$ in both populations. After this period, we "instituted" a minimum size limit, and reduced harvest pressure to 50% of historically stable levels, similar to that which occurred in the nearshore commercial finfish fishery during this time. In 2003, we removed harvest on one population to resemble the initiation of an MPA. At this point we began making harvest adjustments on the population outside the reserve using the decision tree model. All other dynamics remained the same with the exception that the harvested population received a maximum of 5% of the available recruiting age 1 individuals from the MPA population via larval spillover.

The equations for catch and CPUE and the resulting size structures from these metrics were assumed to be taken from a standardized catch and release sampling regime inside and outside of MPAs in collaboration with commercial fishermen. The selectivity of the gear, and thus the resulting metrics are similar to that which occurs in the commercial fishery.

Decision Tree Parameter Optimization

We optimized four decision tree parameters that had significant influence on the adjustment of TAC from one year to the next. Each of the four decision tree levels was parameterized such that the setting of TAC would result in an SPR of 0.4 under limited uncertainty. These four parameters are identified in Table 15 by an asterisk, and include: the number of years over which the slope of $\text{CPUE}_{\text{prime}}$ is calculated in the level 1 slope to target algorithm, the target value for the $\text{CPUE}_{\text{prime}}$ found inside MPAs for level 1, the responsiveness factor (k) in level 1, and the reduction factor for level 4 (Table 15). To optimize these parameters, we explored all possible combinations using 1000 Monte-Carlo simulations, each executed over a 30 year time period with minimal uncertainty. To compare the decision tree model with the status quo precautionary approach, in which the TAC is set at a fraction of historically stable catches, we also calculated SPR values and total catch biomass over a wide range of possible fractions of historically stable catches from 10% to 100%. These simulations were also executed over 30 year periods. We used the 5 years of catch prior to establishment of MPAs for the historically stable period, as this stability held true in all iterations. We plotted the Pareto frontier between the realized SPR and total catch biomass from each of these combinations of parameters for the decision tree model and the precautionary approach (Figure 41) at year 30. The combination of parameters that resulted in the desired levels of $\text{SPR}_{0.4}$, while maximizing total catch biomass at year 30 were chosen for future MSE tests (Table 15, denoted by asterisk) with increased uncertainty.

Management Strategy Evaluation Scenarios

We ran six decision tree scenarios incorporating process and observation error as well as sampling variability and compared them to two scenarios in which TACs were set at 50% of historically stable levels to reflect the current management approach. The eight scenarios (Table 16) examined combined the following conditions: year to year recruitment variation, observation error surrounding CPUE estimates, hyperstability and hyperdepletion relationships between CPUE and abundance, and situations in which fishermen target juveniles disproportionately to their abundance (effort creep).

The uncertainties and error structures covered a broad but not comprehensive range of possible scenarios. The first four scenarios simulated extreme levels of uncertainty for recruitment and CPUE, as well as hyperdepleted conditions of the harvested population (Table 16). In scenario 1 (baseline), we allowed sampling from the population and TAC decisions to be made every year. Scenario 2 (10%) had a 10% limit on the annual allowable decrease in TAC levels as well as a 25% maximum annual allowable increase in TAC. In scenario 3 (3 years), we allowed sampling, TAC decisions and adjustments to be made every third year. Scenario 4 is the reference case in which no decisions were made and a constant precautionary TAC was applied, set at 50% of the average catch levels in the five years prior to MPA establishment. Scenarios 5-8 simulated extreme levels of uncertainty in CPUE and recruitment variability, a hyperstable relationship between CPUE and abundance, and effort creep on recruit sized fish (Table 16). Effort creep on juvenile fishes is modeled into scenarios by placing a 50% effort increase on *recruits* while reducing effort by 50% on *prime* sized and *old* fishes. Scenarios 5-7 were subjected to the same sampling conditions and harvest rules as scenarios 1-3. Scenario 8 is the reference case similar to scenario 4.

Performance Measures

Our model was programmed to maintain SPR as close to 0.4 as possible. Maintenance of $SPR_{0.4}$ may be an appropriate risk averse level for grass rockfish which appear to be shorter lived and more resilient to overfishing than deeper dwelling, long lived west coast rockfish (Parker et al. 2000). To test whether the decision tree model is robust to uncertainty in our MSE scenarios, we calculated the probability that SPR and total biomass dropped below the limit reference points of 10% and 20% of virgin levels during a 30 year period. We also calculated the average SPR and the total catch biomass over the same time period. Total catch biomass was represented as a percent change in total catch relative to the reference scenarios in which the precautionary approach is used to set TACs. We chose these metrics as they cover a range of potential user group objectives.

FS3 - Results

We used the set of parameter combinations (Table 15) that maximized the Pareto efficiency between $SPR_{0.4}$ and total catch biomass at the end of a 30 year time period under minimal uncertainty (Figure 41, denoted by asterisk) for all future MSE tests under various levels of uncertainty. We also examined the tradeoff between $SPR_{0.4}$ and total catch biomass for a range of precautionary harvest levels set between 10% and 100% of historically stable catch levels, and applied annually for a 30 year time period. The precautionary approach never yielded higher SPR and catch than the MPA-based decision tree approach (Figure 41). We then compared the decision tree model under multiple uncertainty scenarios, fishermen behavior, and management options to the precautionary approach for grass rockfish in which the TAC is set at 50% of historically stable catch.

The first three scenarios, consisting of hyperdepletion, recruitment variability, and error in CPUE estimates, resulted in substantially higher total catch biomass after a 30-year period than the precautionary approach. Scenario 1, in which decisions were made annually, resulted in a 91% increase in catch relative to the precautionary method. Scenario 2 resulted in a 147% increase, and catch in Scenario 3 increased by 100% (Table 17). All scenarios maintained SPR and total relative biomass (B/B_o) at levels close to or above the target ($SPR_{0.4}$), except for Scenario 2, in which harvesting was never allowed to increase more than 25% nor decrease more than 10% (Figure 42). In the first three scenarios, biomass did not drop below $0.2B_o$ more than 2.3% of the time. Biomass never dropped below $0.1B_o$ under any scenario, including the precautionary approach (Scenario 4).

In Scenarios 5 through 7, we incorporated hyperstability, effort creep on juvenile fish, recruitment variation, and error around CPUE estimates (Table 16). This extreme variability still managed to substantially increase catches while maintaining total biomass and SPR conditions near target reference levels (Figure 42). When decisions were made annually (Scenario 5), catch increased relative to precautionary levels by 39%. Scenario 6 resulted in a 69% increase, and Scenario 7 resulted in a 32% increase. Scenarios five and seven were both conservative, yielding lower catches but high values for SPR and total relative biomass. Scenario 6 was right on target at $SPR_{0.4}$ at 30 years. For all the scenarios, SPR values and total relative biomass (B/B_o) never dropped below 20% of virgin more than 3% of the time.

Table 17 presents the percent increase in total catch biomass relative to the precautionary approach, the average relative total biomass, and the probabilities that SPR and total relative biomass (B/B_o) drop below 10% and 20% of virgin levels at any time during the 30 year analysis. All eight scenarios revealed that catch levels increased and SPR and biomass levels remained above threshold values whether or not decisions were made every year, every three years, or when a limit on the allowable annual change in TAC was implemented. We plotted the trajectory of SPR over a 30-year time period for all modeled scenarios (Figure 42). As noted above, when hyperstability and effort creep on recruits is modeled into the scenarios, SPR is maintained at high levels and catch decreases. The opposite is true for hyperdepletion in which catches increase and SPR remains between 0.25 and 0.4.

FS4 - Discussion

Our results reveal that data-based management strategies incorporating MPAs provide a powerful tool in helping to set sustainable harvest levels for nearshore sedentary marine species. We found that over a 30 year time period, the decision tree model maintained biomass and SPR levels close to target reference levels in nearly all cases, with little probability of dropping below limit reference points. Catch biomass consistently increased relative to the precautionary approach in which suboptimal harvesting occurred. Although the scenarios examined in this paper do not cover the entire range of possible forms of uncertainty and stock dynamics that influence spatially structured nearshore stocks, the scenarios we used tested the ability of the model to maintain SPR at sustainable levels while also producing high levels of catch.

Important outcomes of this modeling exercise were the gains in efficiency from scenarios in which analyses were performed every three years and TAC levels were constrained to an allowable annual increase of 25% and a decrease of 10%. This is encouraging because the costs of implementing a model such as this will be significantly reduced if sampling and analyses can be undertaken every three years. Moreover, if fishermen can reasonably expect to maintain stable annual catches, they may be more inclined to share the costs of management.

Non-linear relationships between CPUE and abundance posed significant difficulties for maintaining target SPR levels, especially hyperdepletion. In these cases, TAC was often set too high. When applied to real world cases, issues such as non-linear CPUE estimates should be thoroughly vetted with stakeholders to determine the strength of these interactions. By taking advantage of well-designed, objective-driven monitoring programs for nearshore rocky reef species currently conducted in California, estimates of CPUE may be approximately linearly related to abundance. The sampling methodology should always be standardized to reduce uncertainty in comparisons. Although we used CPUE as the level one metric to compare inside and outside of MPAs, it is perfectly reasonable to test the ability of fisheries independent sampling such as diver transect surveys to set the initial TAC. This may further reduce nonlinearities in CPUE and abundance relationships. In fact, we recommend that a thorough examination of all possible data sources should be subjected to MSE if and when a method such as this is formally accepted for design and use in a fishery.

The use of MPAs in this model contributes significantly to the success of this management strategy, but there are a number of potential concerns with using MPAs as proxies for an unfished population. These include (but are not limited to), the relationship between adult movement and size of the MPA and density dependent changes in growth and survivorship of species within MPAs. Indeed, the use of MPAs in this management strategy is successful for those species that have small home ranges relative to the size of the reserve, such that little to no migratory spillover occurs. For many of California's nearshore rocky reef species such as sea urchin, abalone, nearshore fish, crab, and lobster, this assumption may be valid. Density dependent changes in growth and mortality may be more difficult to account for. There is still very little empirical evidence validating changes in these ecological dynamics inside MPAs. We recommend that future use of the decision tree model incorporate ecological dynamics as a means of learning.

We assumed in our simulation tests that the MPA and the fished area were separate, self-recruiting populations, save for the 5% larval spillover out of the reserve into the fished area. It is clear that increased rates of larval spillover significantly decrease the potential for dropping below threshold values of SPR and total relative biomass while allowing for increased catches. A full examination of larval connectivity scenarios between the reserve and the fished population is beyond the scope of this paper, but should be considered when determining the appropriate spatial scale at which to apply a method such as this.

We did not include the aggregate contribution of individuals inside the MPA to our calculations of total biomass and SPR. Therefore our calculations of the probability of dropping below critical values of biomass and SPR are extremely conservative. In real world applications, the size and spacing of MPAs relative to the harvested area will play a major role in determining the true probability of a population dropping below threshold values. A full examination of size and spacing of MPAs is beyond the scope of this paper. Nevertheless, we recommend that a rigorous evaluation of these issues, based on the best available information, be conducted by stakeholder groups engaged in designing a decision tree process.

The MSE we performed assumed that life history information such as growth and natural mortality were known without error in the equilibrium models (Levels 3 and 4), thereby biasing our results. This assumption causes the population to stabilize at a level above or below the target reference point indefinitely (Campbell et al. 2007). The propensity of the decision tree model to stabilize population indicators under uncertainty around life history data is superior to traditional stock assessments in which misinformation may result in stock decline or even collapse. Nevertheless, consideration of the potential problems associated with errors around basic life history information is warranted and basic biological research to gather needed data is advised. If there is valid concern about dropping below $SPR_{0.4}$ due to uncertainty around life history information, or other forms of process and observation uncertainty, the best solution would be to set reference SPR levels greater than necessary thereby increasing precaution.

Our scenarios represent relatively simple cases that do not fully illustrate the flexibility of the decision tree process, especially in its capacity to use various forms of information and generate different outputs. For example, fisheries independent estimates of density such as diver transect surveys could be used in level 1. Instead of generating a TAC, which may not be the appropriate regulatory metric, effort allocation (number of traps or days) outputs can be generated. Many different adjustments to the model are possible, and should be thoroughly considered before full MSE and implementation. As in any management strategy, management objectives should be thoroughly discussed among stakeholders, and when possible, formal evaluation of empirical data should be used in simulation models prior to proceeding with any strategy. In our case study with grass rockfish, we chose the decision tree parameters that maximized catch while maintaining $SPR_{0.4}$. However, a well-organized stakeholder process should examine these target reference points and objective functions to design a strategy that best suits the needs of the fishery. The decision tree process provides the opportunity for stakeholders to proactively manage the fishery in a transparent procedural framework, rather than through a reactionary approach (Campbell et al. 2007).

We suggest that efficient gains in management can be achieved by adopting use of the decision tree in a localized, collaborative framework. The appropriate spatial scale of management units should consider the spatial variability in demographic rates, the geographic placement of MPAs, and the ability to organize stakeholders at ports of landing. This method has potential to fulfill the goals of the California Marine Life Management Act (MLMA, 1998) and lead to effective community based management for a number of reasons: 1) The fisheries dependent nature of the data inputs required in the model present a tremendous opportunity to include fishermen in collaborative research and management, 2) The spatial scale with which MPAs are being implemented will allow for socially and biologically appropriate regulations reflecting variability in harvest pressure, demographics and social organization in local ports, 3) The method is transparent, user friendly, and generally understood by fishermen and community stakeholders at large, 4) The use of MPAs in this process supports the stated goals of the CDFG who advocate MPAs as tools in fisheries management (CDFG 2002).

Research programs that foster community involvement in the data collection and management of nearshore finfish and other species (e.g., www.calobster.org) provide a foundation to develop and implement collaborative management programs like the decision-tree process. The authors are currently engaged in a research program that fosters community involvement in the data collection and management of nearshore finfish and other species at the northern Channel Islands off the coast of Santa Barbara, CA. We are gathering spatially explicit life history information, size structure and CPUE data on grass rockfish, cabezon (*Scorpaenichthys marmoratus*), California sheephead (*Semicossyphus pulcher*) and other nearshore finfish harvested in the live fish fishery in California. There is growing interest among the involved stakeholders to explore management options, including the establishment of an experimental program centered on using the decision tree framework to manage nearshore finfish at the Channel Islands.

Implementing novel assessment techniques for data poor stocks in California and elsewhere will first require adaptive approaches at local scales. The success of such programs will rely heavily on the involvement of local communities, the flexibility of the management authority and the scientific rigor of the decision making strategy. As such, we are continually refining the evaluation process as stakeholder objectives become clear and more complex issues such as spatial connectivity of populations and dedicated access agreements are considered. We encourage further discussion of this approach from the stakeholder communities at large in order to stimulate reform in California's nearshore fisheries management.

Table 14. Parameter set of the operating model.

Parameter	Value	Source	Definition
# of age classes	19 +	Love and Johnson 1998	19 age classes and a plus group
s	0.8	Pers. comm. Alec MacCall	1 - M
u (years 1-30)	0.51	-	$3 * F_{max}$
z	0.75	Pers. comm. Alec MacCall	Steepness of B-H stock-recruit function
$L\infty$	51.3	Love and Johnson 1998	Asymptotic von Bertalanffy length
k	0.11	Love and Johnson 1998	Von Bertalanffy growth parameter
t_o	-2.41	Love and Johnson 1998	Theoretical age at length 0
$\alpha1$	0.045	Love and Johnson 1998	Coefficient of the length-weight relationship
$\beta1$	2.77	Love and Johnson 1998	Coefficient of the length-weight relationship
$\alpha2$	0.12	Love and Johnson 1998	Coefficient of the length-fecundity ogive
$\beta2$	4.09	Love and Johnson 1998	Coefficient of the length-fecundity ogive
$\alpha3$	-0.73	Love and Johnson 1998	Coefficient of the length-maturity relationship
$\beta3$	17.49	Love and Johnson 1998	Coefficient of the length-maturity relationship
hyperdepletion	0.5	Hilborn and Walters (1992)	Non linear relationship b/w CPUE and abundance
hyperstability	1.5	Hilborn and Walters (1992)	Non linear relationship b/w CPUE and abundance

Table 15. Parameters used in the decision tree model at each level of inquiry. Asterisks indicate the four parameters that were optimized using formal techniques. All other parameter values were taken from previous work (Campbell et al. 2007) and discussions with fishery scientists.

Decision Level	Parameter	Value
Level 1	- Number of years over which the slope of $CPUE_{prime}$ is calculated (slope to target)	10 years*
	- Target value for $CPUE_{prime}$	0.4 of $CPUE_{prime}$ inside MPA*
	- Feedback gain/responsiveness factor, *k*	0.9*
	- time till MPA achieves carrying capacity	10 years
Level 2	- Bound on the percentage annual change in $CPUE_{prime}$ to define stability in this indicator (Note: change is relative to the mean value of $CPUE_{prime}$ over the previous 5 years)	5% per year
	- Number of years mean $CPUE_{prime}$ is calculated over	5 years (weighted moving average)
Level 3	- Target value for $CPUE_{Old}$	*SPR* = 0.4
	- Target value for *Proportion-Old*	*SPR* = 0.4
Level 4	- Value of $CPUE_{Recruits}$ to define high recruitment	80% *CPUEo*
	- Decrease in $CPUE_{Recruits}$ to define declining recruitment	10% per year
	- Reduction factor on *TAC*	10%*
	- Number of years mean $CPUE_{Recruits}$ is calculated over	5 years (weighted moving average)

Table 16. The 8 scenarios modeled in this case study. Columns 2 and 3 represent life history information, columns 4-7 represent various uncertainties in the model, and the final column depicts how decisions were made using the decision tree. See text for details.

Scenario	m	Steepness	Sigma R	Sigma CPUE	Effort Creep	Hyper stability/ depletion	Decision making
1	0.2	0.75	0.6	0.5	0	hyperdepletion	baseline
2	0.2	0.75	0.6	0.5	0	hyperdepletion	10%
3	0.2	0.75	0.6	0.5	0	hyperdepletion	3 years
4	0.2	0.75	0.6	0.5	0	hyperdepletion	no dec.
5	0.2	0.75	0.6	0.5	-0.5	hyperstability	baseline
6	0.2	0.75	0.6	0.5	-0.5	hyperstability	10%
7	0.2	0.75	0.6	0.5	-0.5	hyperstability	3 years
8	0.2	0.75	0.6	0.5	-0.5	hyperstability	no dec.

Table 17. Outputs from decision tree management strategy evaluation. Percent Catch Change relates the percentage increase or decrease in catch relative to the baseline precautionary approach (Scenarios 4 and 8). Columns 2 and 3 depict the probability that the Spawning Potential Ratio (SPR) will drop below critical thresholds of 0.10 and 0.20 of unfished levels in 1000 Monte-Carlo simulations. Columns 4 and 5 depict the probability that the total biomass will drop below critical thresholds of 0.10 and 0.20 in 1000 Monte-Carlo simulations. Average SPR is the average Spawning Potential Ratio for the 30 year time period.

Scenario	% Catch Change	<0.10 SPR	<0.20 SPR	<0.10 Biomass	<0.20 Biomass	Avg. SPR
1	+91	1.42	22.84	0.00	1.79	0.32
2	+147	1.82	28.35	0.00	2.42	0.30
3	+100	1.70	25.40	0.00	2.29	0.32
4	0	0.43	10.19	0.00	0.93	0.49
5	+39	0.99	17.09	0.00	1.25	0.43
6	+69	1.85	24.04	0.00	2.42	0.35
7	+32	1.67	22.81	0.00	2.25	0.38
8	0	0.47	11.17	0.00	0.98	0.46

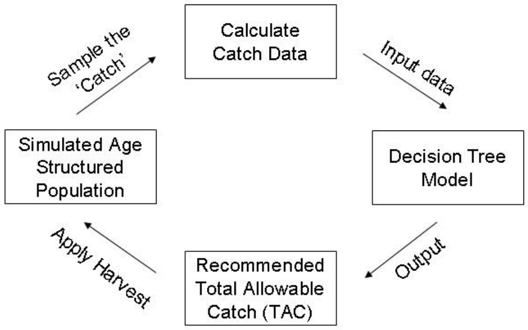
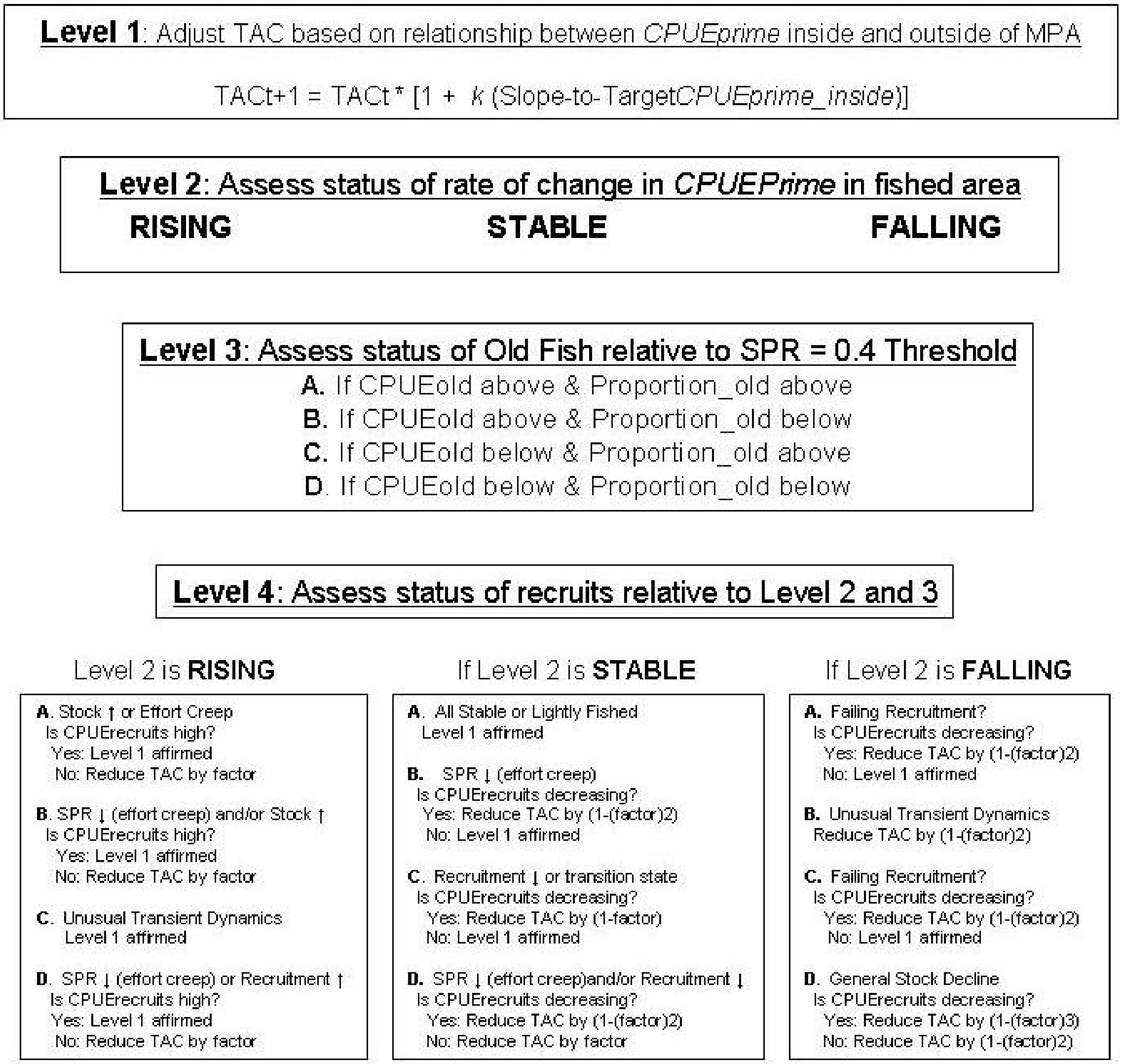


Figure 39. Flow chart of the Management Strategy Evaluation (MSE) Process.



Level 1: Adjust TAC based on relationship between *CPUEprime* inside and outside of MPA

TACt+1 = TACt * [1 + *k* (Slope-to-Target*CPUEprime_inside*)]

Level 2: Assess status of rate of change in *CPUEPrime* in fished area

RISING **STABLE** **FALLING**

Level 3: Assess status of Old Fish relative to SPR = 0.4 Threshold

A. If CPUEold above & Proportion_old above
B. If CPUEold above & Proportion_old below
C. If CPUEold below & Proportion_old above
D. If CPUEold below & Proportion_old below

Level 4: Assess status of recruits relative to Level 2 and 3

Level 2 is **RISING**

A. Stock ↑ or Effort Creep
Is CPUErecruits high?
Yes: Level 1 affirmed
No: Reduce TAC by factor

B. SPR ↓ (effort creep) and/or Stock ↑
Is CPUErecruits high?
Yes: Level 1 affirmed
No: Reduce TAC by factor

C. Unusual Transient Dynamics
Level 1 affirmed

D. SPR ↓ (effort creep) or Recruitment ↑
Is CPUErecruits high?
Yes: Level 1 affirmed
No: Reduce TAC by factor

If Level 2 is **STABLE**

A. All Stable or Lightly Fished
Level 1 affirmed

B. SPR ↓ (effort creep)
Is CPUErecruits decreasing?
Yes: Reduce TAC by (1-(factor)2)
No: Level 1 affirmed

C. Recruitment ↓ or transition state
Is CPUErecruits decreasing?
Yes: Reduce TAC by (1-factor)
No: Level 1 affirmed

D. SPR ↓ (effort creep)and/or Recruitment ↓
Is CPUErecruits decreasing?
Yes: Reduce TAC by (1-(factor)2)
No: Reduce TAC by factor

If Level 2 is **FALLING**

A. Failing Recruitment?
Is CPUErecruits decreasing?
Yes: Reduce TAC by (1-(factor)2)
No: Level 1 affirmed

B. Unusual Transient Dynamics
Reduce TAC by (1-(factor)2)

C. Failing Recruitment?
Is CPUErecruits decreasing?
Yes: Reduce TAC by (1-(factor)2)
No: Level 1 affirmed

D. General Stock Decline
Is CPUErecruits decreasing?
Yes: Reduce TAC by (1-(factor)3)
No: Reduce TAC by (1-(factor)2)

Figure 40. Schematic of the 4 levels of the MPA-Based Decision Tree Model.

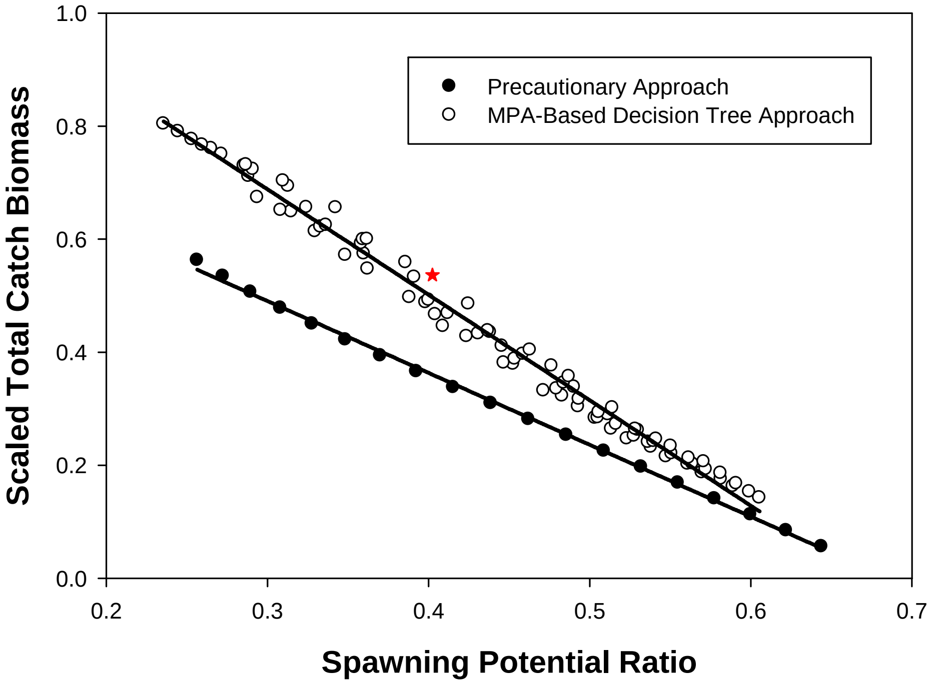


Figure 41. Schematic depicting the tradeoffs between Spawning Potential Ratio (SPR) and total catch biomass. The open circles represent the combinations of the four most critical decision tree parameter that we searched over to find the optimal parameterization. The closed circles represent a range of TAC levels between 10% and 100% of historically stable catch levels in our simulated population reflecting the precautionary approach to management. The red star indicates the chosen combination of parameter values for future Management Strategy Evaluation (MSE).

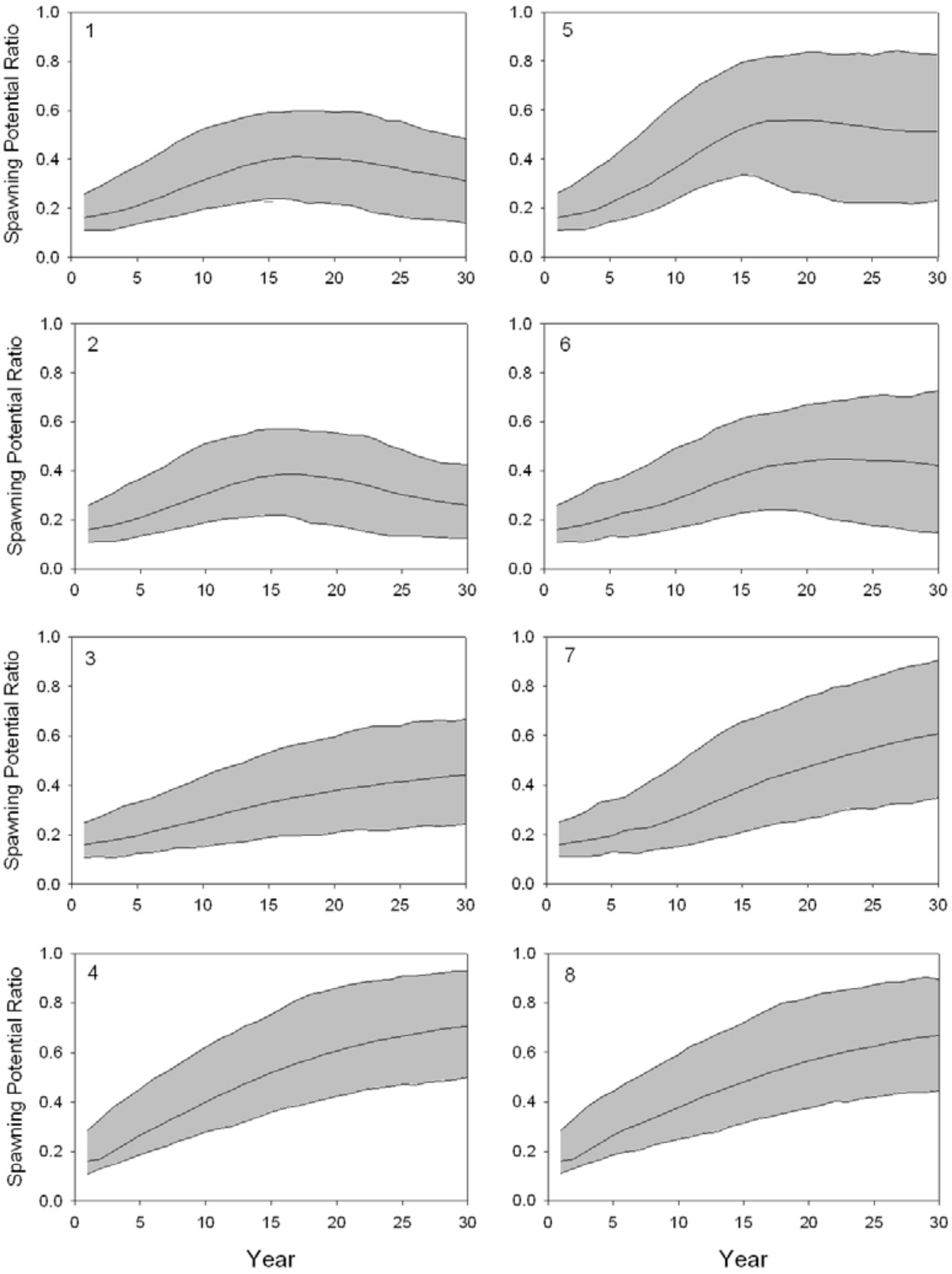


Figure 42. Results of eight scenarios using Management Strategy Evaluation (MSE) for a 30 year time period. The solid black line depicts the median Spawning Potential Ratio (SPR) over a 30 year time period using the MPA-based decision tree. The grey shaded area represents the range of the 10th and 90th percentile region of SPR. The hashed line represents the target value of $SPR_{0.4}$. The numbered inset relates to the scenario modeled (1-8). Scenarios four and eight are the precautionary scenarios in which harvest was constant at 50% of historically stable catch levels.

Literature Cited

Abesamis, R., and G. Russ. 2005. Density-dependent spillover from a marine reserve: long-term evidence. Ecological Applications **15**:1798-1812.

Abesamis, R., G. Russ, and A. Alcala. 2006. Gradients of abundance of fish across no-take marine reserve boundaries: evidence from Philippine coral reefs. Aquatic Conservation: Marine and Freshwater Ecosystems **16**:349-371.

Acosta, C. 2001. Assessment of the functional effects of a harvest refuge on spiny lobster and queen conch populations at Glover's Reef, Belize. Proceedings of the Gulf and Caribbean Research Institute **52**:212-221.

Acosta, C., and D. Robertson. 2003. Comparative spatial ecology of fished spiny lobsters *Panulirus argus* and an unfished congener *P. guttatus* in an isolated marine reserve at Glover's Reef atoll, Belize. Coral Reefs **22**:1-9.

Alonzo SH, Key M, Ish T, MacCall A (2004) Status of the California sheephead (*Semicossyphus pulcher*) stock (2004). California Department of Fish and Game. 146 pp.

Babcock, R., J. Phillips, M. Lourey, and G. Clapin. 2007. Increased density, biomass and egg production in an unfished population of western rock lobster (Panulirus cygnus) at Rottnest Island, Western Australia. Marine and Freshwater Research **58**:286-292.

Babcock, R., N. Shears, A. Alcala, N. Barrett, G. Edgar, K. Lafferty, T. McClanahan, and G. Russ. 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. Proceedings of the National Academy of Sciences.

Barrett, N., C. Buxton, and C. Gardner. 2009. Rock lobster movement patterns and population structure within a Tasmanian marine protected area inform fishery and conservation management. Marine and Freshwater Research **60**:417-425.

Beamish**, R.J. and D.A.**Fournier 1981. A Method for Comparing the Precision of a Set of Age Determinations**.** Canadian Journal of Fisheries and Aquatic Sciences, 1981, 38:(8) 982-983, 10.1139/f81-132.

Bertelsen, R., and T. Matthews. 2001. Fecundity dynamics of female spiny lobster (Panulirus argus) in a south Florida fishery and Dry Tortugas National Park lobster sanctuary. Marine and Freshwater Research **52**:1559-1569.

Beverton, R., and S. Holt. 1957. On the Dynamics of Exploited Fish Populations. Chapman and Hall, London.

Board, O. S. 2000. Improving the Collection, Management, and Use of Marine Fisheries Data. National Research Council, Washington DC.

Booth, J. 2001. Habitat preferences and behavior of newly settled Jasus edwarsii (Palinuridae). Marine and Freshwater Research **52**:1055-1065.

Botsford, L., D. Brumbaugh, C. Grimes, J. Kellner, J. Largier, M. O'Farrell, S. Ralston, E. Soulanille, and V. Wespestad. 2009. Connectivity, sustainability, and yield: bridging the gap between conventional fisheries management and marine protected areas. Reviews in Fish Biology and Fisheries **19**:69-95.

Campbell, R., and coauthors. 2007. Development and preliminary testing of the harvest strategy framework for the Eastern and Western tuna and billfish fisheries

Final Report to the Australian Fisheries Management Authority. Published by CSIRO Marine and Atmospherics Research, Castray Esplanade, Hobart, Tasmania, Australia.70 pp.

CDFG. 2008. Channel Islands Marine Protected Areas: First 5 Years of Monitoring: 2003-2008.

Cailliet, GM, MS Love, and AW Ebeling. 1986. Fishes: a field and laboratory manual on their structure, identification, and natural history. Wadsworth Publishing, Belmont, California.

Cole, R., T. Ayling, and R. Creese. 1990. Effects of marine reserve protection at Goat Island, northern New Zealand. New Zealand Journal of Marine and Freshwater Research **24**:197-210.

Cooke, J.G. 1999. Improvement of fishery management advice through simulation testing of harvest algorithms. *ICES J. Mar. Sci.* (1999) 56 (6): 797-810

Conway, F., and C. Pomeroy. 2006. Evaluating the human - as well as the biological - objectives of cooperative fisheries research. Fisheries **31**:447-454.

Cote, I., I. Mosqueira, and J. Reynolds. 2001. Effects of marine reserve characteristics on the protection of fish populations: a meta analysis. Journal of Fish Biology **59 (Supplement A)**:178-189.

Council), N. N. R. 2004. Cooperative Research in the National Marine Fisheries Service. National Academies Press, Washington, DC.

Cox, C., and J. Hunt. 2005. Change in size and abundance of caribbean spiny lobsters Panulirus argus in a marine reserve in the Florida Keys National Marine Sanctuary, USA. Marine Ecology Progress Series **294**:227-239.

Davidson, R., E. Villouta, R. Cole, and R. Barrier. 2002. Effects of marine reserve protection on spiny lobster (Jasus edwarsii) abundance and size at Tonga Island Marine Reserve, New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems **12**:213-227.

Davis, G., and J. Doldrill. 1980. Marine parks and senctuaries for spiny lobster fisheries management. Proceedings of the Gulf and Caribbean Research Institute **32**:194-207.

DeMartini, E. 1993. Modeling the potential of fishery reserves for managing Pacific coral reef fishes. Fishery Bulletin **91**:414-427.

Edgar, G., and N. Barrett. 1997. Short term monitoring of biotic change in Tasmanian marine reserves. Journal of Experimental Marine Biology and Ecology **213**:261-279.

Edgar, G., and N. Barrett. 1999. Effects of the declaration of marine reserves on Tasmanian reef fishes, invertebrates and plants. Journal of Experimental Marine Biology and Ecology **242**:107-144.

Eggleston, D., and C. Dahlgren. 2001. Distribution and abundance of Caribbean spiny lobsters in the Key West National Wildlife Refuge: relationship to habitat features and impact of an intensive recreational fishery. Marine and Freshwater Research **52**:1567-1576.

FAO. 2007. Report and Documentation of the Expert Workshop on Marine Protected Areas and Fisheries Management: Review of Issues and Considerations. *FAO Fisheries Report.* No. 825. FAO, Rome.

Follesca, M., D. Cuccu, R. Cannas, A. Sabatini, and A. Cau. 2008. Emigration and retention of Palinurus elephas (Fabricius, 1787) in a central western Mediterranean marine protected area. Scientia Marina **71**:279-285.

Forcada, A., C. Valle, P. Bonhomme, G. Criquet, G. Cadiou, P. Lenfant, and J. Sanchez-Lizaso. 2009. Effects of habitat on spillover from marine protected areas to artisinal fisheries. Marine Ecology Progress Series **379**:197-211.

Goni, R., S. Adlerstein, D. Alvarez-Berastegui, A. Forcada, O. Renones, G. Criquet, S. Polti, G. Cadiou, C. Valle, P. Lenfant, P. Bonhomme, A. Perez-Ruzafa, J. Sanchez-Lizaso, J. Garcia-Charton, G. Bernard, V. Stelzenmuller, and S. Planes. 2008. Spillover from six western Mediterranean marine protected areas: evidence from artisinal fisheries. Marine Ecology Progress Series **366**:159-174.

Goni, R., R. Hilborn, D. Diaz, S. Mallol, and S. Alderstein. 2010. Net contribution of spillover from a marine reserve to fishery catches. Marine Ecology Progress Series **400**:233-243.

Goni, R., A. Quetglas, and O. Renones. 2006. Spillover of spiny lobsters Palinuris elephas from a marine reserve to an adjoining fishery. Marine Ecology Progress Series **308**:207-219.

Goni, R., O. Renones, and A. Quetglas. 2001. Dynamics of a protected Western Mediterranean population of the European spiny lobster Palinurus elephas (Fabricius, 1787) assessed by trap surveys. Marine and Freshwater Research **52**:1577-1587.

Guenette, S., T. Lauck, and C. Clark. 1998. Marine reserves: from Beverton Holt to the present. Reviews in Fish Biology and Fisheries **8**:251-272.

Halpern, B. 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecological Applications **13 Supplement**:S117-S137.

Halpern, B., and R. Warner. 2002. Marine reserves have rapid and lasting effects. Ecology Letters **5**:361-366.

Hamilton, L., and O. Otterstad. 1998. Demographic change and fisheries dependence in the Northern Atlantic. Human Ecology Review **5**:16-22.

Hampton, J., J. Sibert, P. Kleiber, M. Maunder, and S. Harley. 2005. Decline of Pacific tuna populations exagerrated? Nature **434**:E1.

Harmelin-Vivien, M., L. Le Direach, J. Bayle-Sempere, E. Charbonnel, J. Garcia-Charton, D. Ody, A. Perez-Ruzafa, O. Renones, P. Sanchez-Jerez, and C. Valle. 2008. Gradients of abundance and biomass across reserve boundaries in six Mediterranean marine protected areas: Evidence of fish spillover? Biological Conservation **141**:1829-1839.

Hartley, T., and R. Robertson. 2009. Stakeholder collaboration in fisheries research: Integrating knowledge among fishing leaders and science partners in northern New England. Society & Natural Resources **22**:42-55.

Hilborn, R. 2006. Faith-based fisheries. Fisheries **31**:554-555.

Hilborn, R., F. Micheli, and G. De Leo. 2006. Integrating marine protected areas with catch regulation. Canadian Journal of Fisheries and Aquatic Sciences **63**:642-649.

Hilborn, R., and C. Walters. 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Chapman & Hall, New York.

Hunt, J., T. Matthews, D. Forcucci, B. Hedin, and R. Bertelsen. 1991. Management implications of trends in the population dynamics of the Caribbean spiny lobster, Panulirus argus, at Looe Key National Marine Sanctuary. Final Report to NOAA. Florida Marine Research Institute, Marathon, FL.

Iacchei, M., P. Robinson, and K. Miller. 2005. Direct impacts of commercial and recreational fishing on spiny lobster, Panulirus interruptus, populations at Santa Catalina Island, California, United States. New Zealand Journal of Marine and Freshwater Research **39**:1201-1214.

Jackson, J., M. Kirby, W. Berger, K. Bjorndal, L. Botsford, B. Bourque, R. Bradbury, R. Cooke, J. Erlandson, J. Estes, T. Hughes, S. Kidwell, C. Lange, H. Lenihan, J. Pandolfi, C. Peterson, R. Steneck, M. Tegner, and R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science **293**:629-637.

Karp, W., C. Rose, J. Gauvin, S. Gaichas, M. Dorn, and G. Stauffer. 2001. Government-industry cooperative fisheries research in the north Pacific under the MSFCMA. Marine Fisheries Review **63**:40-46.

Kaundra-Arara, B., and G. Rose. 2004. Effects of marine reef National Parks on fishery CPUE in Coastal Kenya. Biological Conservation **118**:1-13.

Kay, M., H. Lenihan, C. Miller, and K. Barsky. 2008. Numbers, body sizes and movement of lobster. Pages 8-9 *in* S. Airame and J. Ugoretz, editors. Channel Islands Marine Protected Areas: First 5 Years of Monitoring. California Department of Fish and Game, Santa Barbara.

Kay, M., H. Lenihan, J. Wilson, and C. Miller. *In Press*. The cost of vessel insurance in collaborative fisheries research: Strategies and perspectives from a program in California, USA. California Department of Fish and Game Scientific Journal.

Kellner, J., I. Tetreault, S. Gaines, and R. Nisbet. 2007. Fishing the line near marine reserves in single and multispecies fisheries. Ecological Applications **17**:1039-1054.

Kelly, S., and A. MacDiarmid. 2003. Movement patterns of mature spiny lobsters, Jasus edwarsii, from a marine reserve. New Zealand Journal of Marine and Freshwater Research **37**:149-158.

Kelly, S., D. Scott, and A. MacDiarmid. 2002. The value of a spillover fishery for spiny lobsters around a marine reserve in Northern New Zealand. Coastal Management **30**:153-166.

Kelly, S., D. Scott, A. MacDiarmid, and R. Babcock. 2000. Spiny lobster, Jasus edwarsii, recovery in New Zealand marine reserves. Biological Conservation **92**:359-369.

Key, M., A.D. MacCall, J.Field, D.Aseltine_Neilson. 2008. The 2007 assessment of blue rockfish (*Sebastes mystinus*) in California. Pacific Fishery Management Council, Portland, Oregon.

Kimura, D.K 1980. Likelihood methods for the von Bertalanffy growth curve. Fish Bull (Wash DC) 77:765–774.

Lester, S., B. Halpern, K. Grorud-Colvert, J. Lubchenco, B. Ruttenberg, S. Gaines, S. Airame, and R. Warner. 2009. Biological effects within no-take marine reserves: a global synthesis. Marine Ecology Progress Series **384**:33-46.

Lipcius, R., W. Stockhausen, and D. Eggleston. 2001. Marine reserves for Caribbean spiny lobster: empirical evaluation and theoretical metapopulation recuitment dynamics. Marine and Freshwater Research **52**:1589-1598.

Lotze, H., H. Lenihan, B. Bourque, R. Bradbury, R. Cooke, M. Kay, S. Kidwell, M. Kirby, C. Peterson, and J. Jackson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science **312**:1806-1809.

Love, M. S., and K. Johnson. 1999. Aspects of the life histories of grass rockfish, *Sebastes rastrelliger*, and brown rockfish, *S. auriculatus*, from southern California. Fishery Bulletin. 97(1): 100-109.

Lubchenco, J., S. Palumbi, S. Gaines, and S. Andelman. 2003. Plugging a hole in the ocean: the emerging science of marine reserves. Ecological Applications **13 Supplement**:S3-S7.

MLPA - California Marine Life Protection Act. 1999. FGC Section 2850-2863. Sacramento, CA. California Department of Fish and Game, Marine Region.

MacDiarmid, A. 1991. Seasonal changes in depth distribution, sex ratio and size frequency of spiny lobster *Jasus edwarsii* on a coastal reef in northern New Zealand. Marine Ecology Progress Series **70**:129-141.

MacDiarmid, A., and P. Breen. 1993. Spiny lobster population change in a marine reserve. Pages 47-56 *in* C. Battershill, editor. Proceedings of the Second International Temperate Reef Symposium, 7-10 January 1992, Auckland, New Zealand. NIWA Marine, Wellington.

Mari, M., D. Diaz, P. Abello, and M. Zabala. 2002. Avaluacio temporal de les poblacions de llagosta (*P. elephas*) en les Illes Medes. Pages 62-69 *in* M. Zabala, editor. Evolucio del patrimoni natural de les Illes Medes. Technical Report: Direccio General de Pesca Maritima, Generalitat de Catalunya.

Mayfield, S., G. Branch, and A. Cockroft. 2005. Role and efficacy of marine protected areas for the South African rock lobster, *Jasus lalandii*. Marine and Freshwater Research **56**:913-924.

McCay, J., and S. Jentoft. 1996. From the bottom up: Participatory issues in fisheries management. Society & Natural Resources **9**:237-250.

McClanahan, T., and S. Mangi. 2000. Spillover of exploitable fishes from a marine park and its effects on the adjacent fishery. Ecological Applications **10**:1792-1805.

Milich, L. 1999. Resource mismanagement versus sustainable livelihoods: the collapse of the Newfoundland cod fishery. Society & Natural Resources **12**:625-642.

Moffitt, E., L. Botsford, D. Kaplan, and M. O'Farrell. 2009. Marine reserve networks for species that move within a home range. Ecological Applications:1835-1847.

Morgan, L., L. Botsford, S. Wing, and B. Smith. 2000. Spatial variability in growth and mortality of the red sea urchin, *Strongylocentrotus franciscanus*, in northern California. Canadian Journal of Fisheries and Aquatic Sciences **57**:980-992.

Murawski, S., R. Methot, and G. Tromble. 2007. Biodiversity loss in the ocean: how bad is it? (Letter to the Editor). Science **316**:1281.

Murawski, S., S. Wigley, M. Fogarty, P. Rago, and D. Mountain. 2005. Effort distribution and catch patterns adjacent to temperate MPAs. ICES Journal of Marine Science **62**:1150-1167.

Myers, R., and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. Nature **423**:280-283.

Pande, A., A. MacDiarmid, P. Smith, R. Davidson, R. Cole, D. Freeman, S. Kelly, and J. Gardner. 2008. Marine reserves increase the abundance and size of blue cod and rock lobster. Marine Ecology Progress Series **366**:147-158.

Parnell, P., C. Lennert-Cody, L. Geelen, S. LD, and P. Dayton. 2005. Effectiveness of a small marine reserve in southern California. Marine Ecology Progress Series **296**:39-52.

Pauly, D., J. Alder, E. Bennett, V. Christensen, P. Tyedmers, and R. Watson. 2002. The future for fisheries. Science **302**:1359-1361.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres. 1998. Fishing down marine food webs. Science **279**:860-863.

Pauly, D., V. Christensen, S. Guenette, T. Pitcher, U. Rashid Sumaila, C. Walters, R. Watson, and D. Zeller. 2003. Towards sustainability in world fisheries. Nature **418**.

Pelc, R., M. Baskett, T. Tanci, S. Gaines, and R. Warner. 2009. Quantifying larval export form South African marine reserves. Marine Ecology Progress Series **394**:65-78.

Pew. 2003. Our Living Oceans: Charting a Course for Sea Change. Pew Oceans Commission, Washington, D.C.

Polacheck, T. 1990. Year around closed areas as a mangement tool. Natural Resource Modeling **4**:327-354.

Policy, U. C. o. O. 2004. An Ocean Blueprint for the 21st Century. Final Report. US Commission on Ocean Policy Washington, DC.

Punt, A.E. and R. Hilborn. 1997. Fisheries stock assessment: the Bayesian approach. Reviews in Fish Biology and Fisheries. 7, 35-63.

Quinn, T., and R. Derizo. 1999. Quantitative Fish Dynamics. Oxford University Press, New York.

Rowe, S. 2002. Populaiton parameters of American lobster inside and outside no-take reserves in Bonavista Bay, Newfoundland. Fisheries Research **56**:167-175.

Russ, G., A. Alcala, A. Maypa, H. Calumpong, and A. White. 2004. Marine reserve benefits local fisheries. Ecological Applications **14**:597-606.

Sanchez-Lizaso, J., R. Goni, O. Renones, J. Garcia-Charton, R. Galzin, J. Bayle, P. Sanchez-Jerez, A. Ruzafa, and A. Ramos. 2000. Density dependence in marine protected populations: a review. Environmental Conservation **27**:144-158.

Sethi, S., T. Branch, and R. Watson. 2010. Global fishery development patterns are driven by profit but not trophic level. Proceedings of the National Academy of Sciences **107**:12163-12167.

Shears, N., R. Grace, N. Usmar, V. Kerr, and R. Babcock. 2006. Long-term trends in lobster populations in a partially protected vs. no-take marine park. Biological Conservation **132**:222-231.

Smith, A. D. M., K. J. Sainsbury, and R. A. Stevens. 1999. Implementing effective fisheries-management systems - management strategy evaluation and the Australian partnership approach. ICES Journal of Marine Science. 56(6): 967-979.

Sibert, J., J. Hampton, P. Kleiber, and M. Maunder. 2006. Biomass, size, and trophic status of top predators in the Pacific Ocean. Science **314**:1773-1776.

Stobart, B., R. Warwick, C. Gonzalez, S. Mallol, D. Diaz, O. Renones, and R. Goni. 2009. Long term and spillover effects of a marine protected area on an expolited fish community. Marine Ecology Progress Series **384**:47-60.

Taylor, B., and J. McIlwain. 2010. Beyond abundance and biomass, effects of marine protected areas on the demography of a highly exploited reef fish. Marine Ecology Progress Series **411**:243-258.

Tetreault, I., and R. Ambrose. 2007. Temperate marine reserves enhance targeted but not untargeted fishes in multiple no-take MPAs. Ecological Applications **17**:2251-2267.

UNEP-WCMC. 2008. National and Regional Networks of Marine Protected Areas: A Review of Progress. United Nations Environment Programme World Conservation Monitoring Centre, Cambridge.

Walters, C. 2003. Folly and fantasy in the analysis of spatial catch rate data. Journal of Canadian Fisheries and Aquatic Sciences **60**:1433-1436.

Walters, C., R. Hilborn, and R. Parrish. 2007. An equilibrium model for predicting the efficacy of marine protected areas in coastal environments. Canadian Journal of Fisheries and Aquatic Sciences 64:1009-1018.

Westrheim, S.J. 1975 Reproduction, Maturation, and Identification of Larvae of some *Sebastes* (Scorpaenidae) Species in the Northeast Pacific Ocean. Journal of the Fisheries Research Board of Canada, 32:(12) 2399-2411.

Willis, T., and R. Millar. 2005. Using marine reserves to estimate fishing mortality. Ecology Letters 8:47-52.

Willis, T., R. Millar, R. Babcock, and N. Tolimieri. 2003. Burdens of evidence and the benefits of marine reserves: putting Descartes before des horse? Environmental Conservation **30**:97-103.

Wilson, J., J. Prince, and H. Lenihan. 2010. Setting harvest guidelines for sedentary nearshore species using marine protected areas as a reference. Marine and Coastal Fisheries: Dynamics, Management and Ecosystem Science **2**:14-27.

Worm, B., E. B. Barbier, N. Beaumont, J. E. Duffy, C. Folke, B. S. Halpern, J. B. C. Jackson, H. K. Lotze, F. Micheli, S. R. Palumbi, E. Sala, K. A. Selkoe, J. J. Stachowicz, and R. Watson. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science **314**:787-790.

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Publications

Caselle, J.E., D. Malone, M.H. Carr, J.R. Wilson, and D.E. Wendt. 2010 Can we predict interannual and regional variation in delivery of pelagic juveniles to nearshore populations of rockfishes (genus *Sebastes*) using simple proxies of ocean conditions? CalCOFI Reports vol. **51**: 91-105.

Fujita, R.M., K.T. Honey, A. Morris, J.R. Wilson, and H. Russell. 2010. Cooperative strategies in fisheries management: integrating across scales. Bulletin of Marine Science **86**(2): 251–271.

Hamilton, S.L., J.R. Wilson, T. Ben-Horin, and J.E. Caselle. 2011. Utilizing spatial variation in demography and life histories to improve the management of a temperate sex-changing reef fish. PLoS One **6**(9), e24580. doi:10.1371/journal.pone.0024580.

Kay, M.C., H.S. Lenihan, C.J. Miller, and K. Barsky. 2008. Numbers, body sizes and movement of lobster. *In*: Airame S, Ugoretz J (Eds.) Channel Islands Marine Protected Areas: First 5 Years of Monitoring. Santa Barbara: California Department of Fish and Game. p. 8-9.

Kay, M.C., H.S. Lenihan, J.W. Wilson, and C.J. Miller. 2010. The cost of vessel insurance in collaborative fisheries research: Strategies and perspectives from a program in California, USA. California Fish and Game Scientific Journal **96**:33-49.

Kay, M.C., H.S. Lenihan, C.M. Guenther, J.R. Wilson, S.W. Shrout, and C.J. Miller. *In press.* Collaborative assessment of California spiny lobster (*Panulirus interruptus*) population and fishery responses to a marine reserve network. Ecological Applications.

Wilson, J.R., J.D. Prince, and H.S. Lenihan. 2010. A management strategy for nearshore, sedentary species that uses marine protected areas (MPAs) as a reference. Marine and Coastal Fisheries: Dynamics, Management and Ecosystem Science **2**:14-27.

Publications in review

Kay, M.C., and J.R. Wilson. *In review*. Spatially explicit mortality of California spiny lobster (*Panulirus interruptus*) across a marine reserve network. Environmental Conservation.

Kay, M.C., H.S. Lenihan, M.J. Kotchen, and C.J. Miller. *In review*. Habitat and distance from reserve borders influence California spiny lobster (*Panulirus interruptus*) trap performance at the Santa Barbara Channel Islands. Marine Ecology Progress Series.

Wilson, J.R., M.C. Kay, J. Colgate, R. Qi, and H.S. Lenihan. In review. Identifying spatial management options for grass rockfish (*Sebastes rastrelliger*) through collaborative fisheries research. Marine Ecology Progress Series.

Publications in review (cont…)

Wilson, J.R., and S.R. Valencia. In review. Bounded catch curves to estimate natural and fishing mortality using information from marine reserves. Canadian Journal of Fisheries and Aquatic Sciences.

Wilson, J.R., M.C. Kay, and H.S. Lenihan. In review. Marine Reserves Provide Win-Win Solutions for Fishermen and Conservation. Ecological Applications.

Presentations

Kay MC, Lenihan HS, Kotchen M, and Miller CJ. 2010. Fishing, habitat, and distance from marine borders influence spiny lobster abundance on reefs at the Santa Barbara Channel Islands. Western Society of Naturalists. San Diego, California. November 2010.

Kay MC and Lenihan HS. 2009 Spiny lobster research at the northern Santa Barbara Channel Islands. California Department of Fish and Game spiny lobster stock assessment workshop. Santa Barbara, California. December 2009.

Kay MC, Lenihan HS, Miller CJ and Barsky K. 2008. Collaboration and conservation: Monitoring impacts of Channel Islands MPA's on CA spiny lobster. Society of Conservation Biology Annual Meeting. Chattanooga, Tennessee. July, 2008.

Kay MC, Lenihan HS, Miller CJ and Barsky K. 2008. Influence of Channel Islands Marine Reserves on Spiny Lobster. Channel Islands Symposium, Special Session: The First Five Years of Monitoring the Channel Islands Marine Protected Area Network. Oxnard, California. February 2008.

Kay MC and Lotze HK. 2007. Reconstructing human-induced changes in marine ecosystems. Ecological Society of America (ESA) annual meeting, San Jose, California August 2007.

Wilson, JR. 2010 An MPA based method to set sustainable catch levels. CA Department of Fish and Game Commission, Santa Barbara, CA

Wilson JR and MC Kay. 2010. An MPA based method to set sustainable catch levels. Western Society of Naturalists, San Diego, CA

Wilson, JR. 2009 Optimal size limits of California sheephead (Semicossyphus pulcher).CA Department of Fish and Game, Sheephead Workshop, Santa Barbara, CA.

Wilson JR. 2009. An MPA based method to set sustainable catch levels. Bren School Graduate Student Symposium, UCSB.

Wilson JR and JD Prince. 2008. The tragedy of scale in California fisheries.
Mote International Fisheries Symposium, Mote Marine Lab, Sarasota, FL.