Synthesizing ecological and human use information to understand and manage coastal change

Marie L. Fujitani[*,a,b], Eli P. Fenichel[c], Jorge Torre[d], Leah R. Gerber[e]

*** Corresponding author:**

[a]*Leibniz Center for Tropical Marine Research*
Fahrenheitstraße 6, 28359 Bremen
Marie.Fujitani@leibniz-zmt.de
Tel: +49(0)421 238 00-139

[b]*Leibniz-Institute of Freshwater Ecology and Inland Fisheries, Müggelseedamm 310, Berlin 12587, Germany*
Tel: +49(0)30-64181-657

[c]*School of Forestry & Environmental Studies, Yale University, New Haven, CT 06511, USA*
eli.fenichel@yale.edu

[d]*Comunidad y Biodiversidad A.C., Blvd. Agua Marina #297, entre Jaiba y Tiburon, Colonia Delicias, Guaymas, Sonora 85420, Mexico*
jtorre@cobi.or.mx

[e]*School of Life Sciences, Arizona State University, Tempe AZ 85287-4501, USA*
Leah.Gerber@asu.edu

Synthesizing ecological and human use information to understand and manage coastal change

Keywords:

marine reserves; marine protected areas; marine conservation; human dimensions; MPA; monitoring; management; recreational angling, recreational fishing

ABSTRACT

Coastal systems are constantly in flux, and feedback from monitoring is necessary to support decision making for effective sustainable natural resource management. Frequently natural resources are the ultimate target of management actions, but management programs work through the proximate step of regulating human behavior towards those resources. For example, a marine reserve is considered a conservation success when the abundance and diversity of organisms increase within reserve boundaries, all relative to existing trends that would have affected ecological communities in the absence of a reserve. Biological monitoring can assesses whether reserve management achieves these goals. However, when monitoring data are inconclusive or do not match expectations, managers face uncertainty in understanding why particular biological patterns occurred, whether a reserve is a biologically appropriate management strategy for the system, and what steps to take moving forward. Monitoring human behavior can provide information that may alleviate some uncertainty and help explain observed biological patterns. In this study we illustrate the utility of complimenting biological monitoring data with monitoring of human behavior. We used a before-after control-impact analysis to test for effects of a no-take reserve in the Gulf of California, Mexico on the density and biomass of seven fished species. We failed to detect a positive biological effect of the reserve, and found the density of five monitored species had declined. These results indicated that the reserve was not succeeding, but provided no insight into why. Evaluation of recreational angler use of the reserve provided a possible explanation: first, the frequency of angler visits to the study area was increasing over time. Second, the reserve reduced the propensity of anglers to visit the

reserve, but not by enough to offset the overall increasing visitation trend. Biological and human use monitoring results in tandem indicated that a reserve could potentially be an effective conservation tool for the system, and allowed us to suggest modifications that could help the reserve succeed. Our work illustrates the necessity of monitoring human use changes alongside biological responses to a reserve for a holistic portrait of reserve functioning, providing a concrete example of the importance of human behavioral aspects of marine reserve success.

1. Introduction

Marine reserves are a popular strategy for marine conservation (Roberts et al. 2001, Halpern et al. 2004, Guidetti 2007, McClanahan 2010, Devillers et al. 2015). However, as with any management strategy, reserves are subject to implementation error (uncertainty associated with rules and regulations, and how people respond to them) and process error (uncertainty associated with the biophysical production process) that may undermine the ability of a reserve to meet its conservation and fishery enhancement goals (Wilen et al. 2002, Smith et al. 2008). Monitoring and assessment are intended to help managers identify and address these problems. Frequently, the purpose of reserve monitoring is to determine whether fish stocks have increased in size and abundance; reserve evaluation has focused overwhelmingly on biological criteria (Halpern and Warner 2002, Halpern 2003, UNEP-WCMC 2008, Lester et al. 2009, Bonaldo et al. 2017, Woodcock et al. 2017). If such a monitoring program indicates increasing stocks, this provides positive feedback for the current management program. However, some reserves may show no effect on protected species, despite decades of operation and monitoring

3

(Halpern and Warner 2002, Babcock et al. 2010). If a monitoring program fails to provide evidence of stock improvements, then it is tempting to declare the reserve a failure. If the reserve is indeed failing, then a prudent manager might desire information to understand why: perhaps the reserve is too small or in the wrong place (Devillers et al. 2015), or the life history characteristics of the monitored species render them unsuitable for management by the current marine reserve structure (Gerber et al. 2002, Claudet et al. 2010). There may be a lack of stakeholder stewardship or knowledge of the rules coupled with inadequate enforcement to protect the reserve (Guidetti et al. 2008, Kelaher et al. 2015, Watson et al. 2015). Failure to observe increases in fish size and abundance provides no information about the likelihood of any of these alternative hypotheses, and thus provides little guidance to improve future management. Even a result indicating stock improvement is only informative to a point; management still has only incomplete information to help sustain the reserve’s performance or provide lessons learned for applications in other locations. We assess the ultimate and proximate impacts of the creation of a marine reserve in the Gulf of California, Mexico, and demonstrate that in tandem these impacts help us to understand how the reserve is functioning mechanistically. The information garnered from monitoring proximate effects (i.e. changes in human use) provides approaches to help managers meet their goals. Thus, the issues raised by this study are broadly applicable to natural resource management situations where regulatory interventions attempt to affect human behavior.

Marine reserve assessment is frequently viewed as a two-step process: a reserve is created (Fig. 1; I.) and over time the biological effects of the reserve are observed (Fig. 1; IV.). Yet at the core marine reserves, like other policies, adjust the rules and incentives of people; for example through limitations on access or use (Lynch 2006, Fenichel et al. 2013) (Fig. 1; II.). The ultimate success of a reserve requires management provisions and incentives that change human behavior in ways that support reserve goals (Fig 1; III.). Thus, people must first alter their behavior in response to a reserve before any biological changes (Fig. 1; IV.) attributable to the reserve can occur. The response of fish stocks may further alter people's incentives leading to feedbacks between stock changes and behavior (Fig. 1; V.). Mechanistically the biological response following declaration of new regulations lags human behavioral change, and thus logic suggests that human use changes may be observed earlier and possibly with less error than biological responses.

Though it is generally assumed that a marine reserve will discourage use of a marine area by increasing the cost of fishing through fines (Smith et al. 2008), the creation of new rules often result in unintended human behavioral responses and feedbacks (Wilen et al. 2002, Smith and Wilen 2003, Kellner et al. 2007, Smith et al. 2008, Lynch 2014). Behavioral responses that could reduce or eliminate the conservation benefits of a reserve include the intensification of fishing effort (Smith et al. 2008), spatial displacement of fishing to reserve boundaries (Gribble and Robertson 1998, Kellner et al. 2007), and non-compliance with reserve rules (Kritzer 2004, Guidetti et al. 2008). Understanding how a reserve alters human use provides insight into how the reserve is functioning mechanistically (Hilborn 2007). While stock increases or declines may be the ultimate management outcomes of concern, intermediate information tracking human use of a marine reserve is essential to understand and improve management.

We first describe a biological assessment (Fig. 1; IV) of the San Pedro Mártir Island (SPMI) Biosphere Marine Reserve in the Gulf of California (GoC), Mexico (Fig.

2). Revisiting the conceptual system model of the reserve's socio-ecological production processes, we assess an intermediate step using results of human use monitoring for the reserve (Fig. 1; III). We present the results of each assessment, and illustrate how in tandem they present a fuller picture of the reserve. We discuss the management implications of this monitoring regime to the SPMI reserve, and elaborate on how human use monitoring, as with other social science data, are essential for marine reserve monitoring and success.

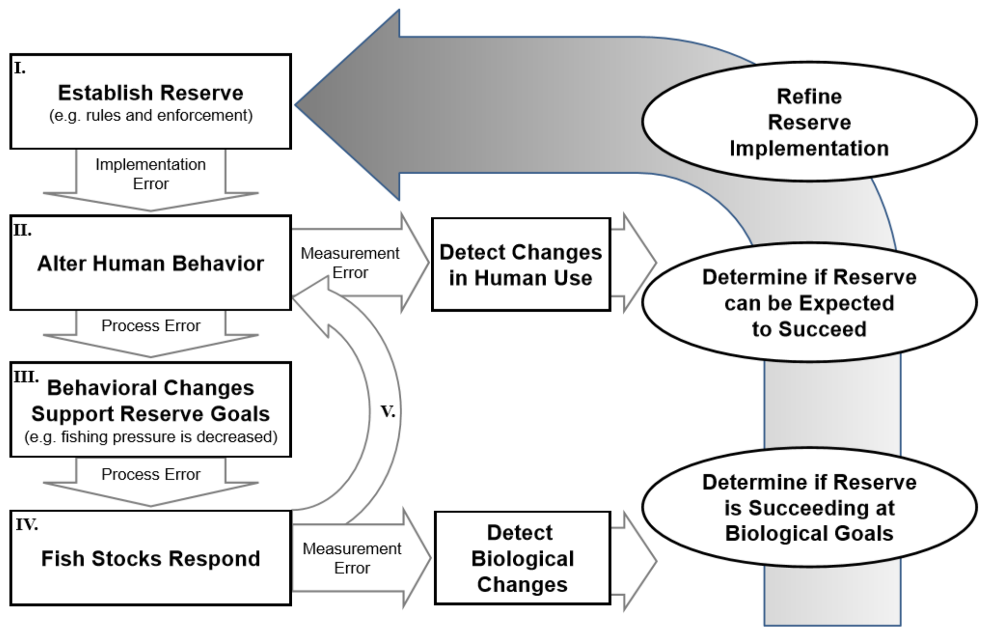

Fig. 1. A conceptual iterative management model of how the effects of a marine reserve are carried to protected fish populations. The stages where human use and biological monitoring occur are illustrated along with sources of error that confound the detection of treatment effects.

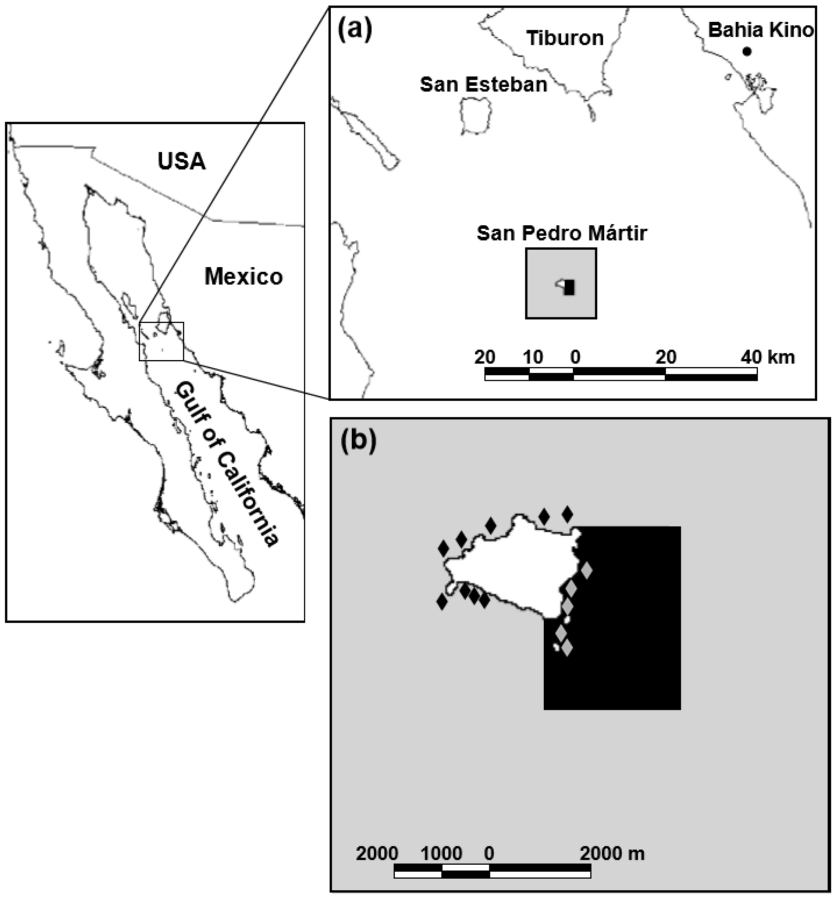

Fig. 2. Map of the Gulf of California, Mexico. (a) Inset map of the Central Gulf of California Midriff Islands region. The square around San Pedro Mártir Island delineates the boundaries of the Biosphere Reserve. The grey area is the buffer zone, and the black area the no-take zone. (b) Close-up of San Pedro Mártir Island showing the no-take zone and part of the buffer zone. Diamonds indicate sites inside (grey) and outside (black) sampled with non-destructive underwater survey techniques between 2002 and 2009.

2. Methods

2.1 Study Site

Coastal ecosystems surrounding SPMI (Fig. 2) are highly productive, but high exploitation rates throughout the GoC have led to catch declines (Sala et al. 2004, Saenz-Arroyo et al. 2005) including around SPMI (CEDICAR and CONANP 2007). A biosphere reserve was declared in June 2002 (Poder Ejecutivo Federal 2002). The SPMI biosphere reserve is managed by the Mexican National Commission on Natural Protected areas (CONANP) with support from the local non-governmental organization Comunidad y Biodiversidad (COBI). The creation of the reserve was a community-based effort, and the reserve was located and zoned with active participation from the local community and small-scale commercial fishers, but not local recreational fishers (Cudney-Bueno et al. 2009).

The aim of the biosphere reserve is to protect terrestrial and marine natural resources, and provide for their sustainable use (Poder Ejecutivo Federal 2002). The reserve is composed of a rectangular no-take zone of approximately 9 km^2 surrounded by a buffer zone of approximately 290 km^2 (Fig. 2). Recreational and small-scale commercial fishing are permitted within the buffer zone, while trawling and other industrial-scale fishing is prohibited. Beyond fishing mortality, trawling impacts the structure of the ecosystem and ecosystem processes, and often negatively affects species diversity and abundance (Thrush and Dayton 2002). Further, Gulf of California shrimp trawl fisheries capture as much as 13 times as much biomass as bycatch than shrimp (Perez-Mellado and Findley 1985). Within the buffer zone is a core “no-take” zone. All extractive activities are prohibited in the no-take zone, which previous to reserve designation was considered a good fishing area by personal accounts from Mexican small-scale (Cudney-Bueno et al. 2009) and local recreational (Fujitani 2010) fishers. For example, the no-take zone encompasses an area of spawning aggregations for *Mycteroperca rosacea*, one of the most important and abundant fished grouper in the Central and Southern GoC (Ramírez and Rodríguez 1990, Sala et al. 2003). *M. rosacea* are popular targets for commercial and recreational anglers who fish near SPMI (CEDICAR and CONANP 2007, Fujitani 2010) and are easily captured during aggregations at spawning sites (Sala et al. 2003), only four of which are protected in the GoC (Sala 2005). No penalties for non-compliance are specified in the reserve declaration (Poder Ejecutivo Federal 2002).

Bahia Kino, the nearest fishing village to SPMI (60 kilometers distant; Fig. 2), hosts a sizable foreign (USA, Canadian, and European) semi-permanent resident community of recreational anglers. Our analysis of human use changes resulting from the reserve focused on this Bahia Kino recreational angling community. This was because recreational anglers fished regularly at SPMI while the waters directly surrounding the island were historically visited less frequently by Mexican small-scale commercial fishers (Cudney-Bueno et al. 2009). Unlike most recreational anglers, small-scale fishers use pangas (small <7 m open skiffs), which make the trips time-consuming and hazardous. The island is fished only by a small subset of small-scale fishers, and even then only irregularly due to the distance and cost of travel (Cudney-Bueno et al. 2009). These qualitative data from oral accounts are supported by human use monitoring data from 2003-2008 (Meza et al. 2008). However, these monitoring visits were not conducted to be representative and the number of visits varied widely between years (from 14 in 2003 to

102 in 2007), and thus must be interpreted with caution. The data may provide information on the relative proportions of visitors to SPMI (no site-specific catch data are available for either group). Recreational angling boats were the most frequently observed group (44%), followed by pangas (36%). Also observed were larger commercial boats such as shrimp trawlers and longliners (4%). The remaining boats were sightseeing or diving tourists (3%), reserve enforcement (9%), and unidentified (4%).

Though the reserve was officially created in 2002 regular enforcement did not begin until 2006, with patrols making eight random visits to the no-take zone per month (two per week). The recreational anglers were not involved in the reserve creation process, and as a group learned of the reserve and its rules in 2005 (unpublished interview data). On April 10th, 2005 an announcement (in English) was broadcast on the VHF radio channel that the recreational angling community relies upon for communication. It stated that enforcement of the SPMI Biosphere Reserve was commencing, and federal officers and the navy would enforce penalties of fines, boat impoundment, and imprisonment.

2.2 Biological Sampling Design

From 2002-2004 and 2007-2009 divers sampled fish populations using underwater visual census (UVC) along sub-tidal transects. Divers recorded the abundance and total length for five species of grouper and sea bass (Serranidae) and two species of snapper (Lutjanidae) (Table 1) considered important to both commercial and recreational fishers (CEDICAR and CONANP 2007; unpublished interview data). Data were collected from 15 fixed sites (Fig. 2), though the number of sites sampled varied from year to year. From 2002-2004 divers surveyed 50 x 5 m transects. In 2007, the Partnership for Interdisciplinary Studies of Coastal Oceans (PISCO) monitoring protocol (Hamilton et al. 2010; www.piscoweb.org) was adapted by the PANGAS initiative (Pesca Artesanal del Norte del Golfo de California; pangas.arizona.edu), and divers employed 30 x 2 m transects. Data were standardized by the transect area. The best available length-weight relationships for the seven species were obtained from FishBase (Froese and Pauly 2016) and used to convert recorded lengths to biomass.

To understand recreational angling behavior, we used a dataset that includes information on daily angling destinations from 2000 to 2008 compiled by a volunteer rescue radio channel servicing the recreational angling community of Bahia Kino. No official water rescue services exist in the region, so the community relies upon the volunteer service for boat tracking, search, and rescue. The community maintains a dedicated VHF channel for this purpose. As each boat departs, a volunteer records the planned destination(s) and the expected return time, and the volunteer checks boaters' status until they return. There is a strong incentive for accurate trip reporting to guide potential rescue efforts, as mechanical failures and sudden changes in weather are common (J. Jerdee, personal communication). Almost every member of the recreational angling community checks out to a volunteer on every trip (J. Jerdee, personal communication). Semi-structured interviews with recreational anglers were conducted in June 2009. A convenience sample of anglers was solicited from Bahia Kino's single boat ramp as anglers returned from fishing. Interviews were given at a later date, and lasted about an hour. Topics covered in each interview included targeted species, fishing sites, attitudes towards the SPMI reserve, and attitudes towards the environment. Twenty interviews were taken representing 8% of the boats that were recorded to have taken a fishing trip that year.

Table 1. Species that were quantified in the monitoring program from 2002-2009.

Scientific Name	Common Name	Count Over Study Period
Mycteroperca rosacea	leopard grouper	1087
Lutjanus argentiventris	yellow snapper	153
Cephalopholis panamensis	Panama graysby	106
Epinephelus labriformis	flag cabrilla	87
Mycteroperca jordani	Gulf grouper	60
Mycteroperca prionura	sawtail grouper	42
Hoplopagrus guentherii	barred pargo	16

2.3 Data Analysis

With our UVC transects we compared the abundance and biomass of seven economically important species in the no-take and the buffer zones between 2002 and 2009 to evaluate the impact of the no-take zone with Before-After Control-Impact (BACI) methodology (Green 1979), widely used for impact assessment in the ecological (Emslie et al. 2015, Kelaher et al. 2015) and social science literatures (Orley and Card 1985, Greenstone and Gayer 2009). Because reserves are often sited non-randomly due to characteristics of outstanding conservation importance (UNEP-WCMC 2008), ecological criteria (Roberts et al. 2003), or opportunity, cost, and political considerations (Roberts 2000, Hansen et al. 2011), analyses to detect the effect of a marine reserve is a 'quasi-experiment,' not a true randomized experiment (Rubin 1974). BACI analyses have advantages over Before-After or Control-Impact (e.g., Pollnac et al. 2001, Halpern 2003, Pollnac et al. 2010) comparisons, by helping account for unobserved covariates (e.g., if in the absence of a reserve, a reserve site would still have higher biological productivity than an outside reference site) (Stewart-Oaten et al. 1986).

For our BACI analysis we set the implementation of the SPMI biosphere reserve as the impact. Though the reserve was designated in 2002, full implementation did not occur until 2005, and enforcement was stepped up in 2006 (J. Torre, personal communication). The first year UVC data were collected, following the full implementation of the reserve, was 2007. Data collected between 2007 and 2009 were used as the after period.

We used a linear mixed model to detect an effect of reserve implementation on abundance and biomass of fish species in the no-take zone. This approach allowed the inclusion of differences in underlying distributions by setting group-level model intercepts and variance while providing parameter estimates for fixed effects (McDonald et al. 2000, Gelman and Hill 2007). Mixed-effects models are useful to analyze hierarchically grouped data as they partition group level variation from the fixed effects of interest that characterize all individuals in the model (Fox 2002, Gelman and Hill 2007). Mixed models are commonly used in reserve evaluation to account for group-level variation in, for example, site (Bonaldo et al. 2017), country (Geldmann et al. 2015), or observer (White et al. 2015). Linear mixed models were fit by maximum likelihood in the statistical programming environment R (v. 3.2.5; R Development Core Team 2016) using the package *nlme* (Pinheiro et al. 2016) and checked for improvement compared to the null model as restricted maximum likelihood cannot be used to compare models with different fixed effect structures (Zuur et al. 2009). Final models were fit by restricted maximum likelihood with Satterthwaite approximations to degrees of freedom for t-tests on model fixed effects (package lmerTest; Kuznetsova 2016).

$$\mathbf{y}_{ij} = \mathbf{X}_i\boldsymbol{\beta} + \mathbf{b}_j + \boldsymbol{\varepsilon}_{ij} \quad \mathbf{b}_j \sim \mathbf{N}_{\mathrm{j}}\,(0, \boldsymbol{\Psi}) \qquad \boldsymbol{\varepsilon}_{ij} \sim \mathbf{N}_{ij}\,(0, \sigma^2)$$

$\mathbf{y}_{ij}$ was the response vector for the i^{th} observation in the j^{th} group (survey location) for each model. $\mathbf{X}_i$ was the design matrix, and $\boldsymbol{\beta}$ was the vector of coefficients for fixed effects ‘before or after’ (i.e. 2002-2004 or 2007-2009; before = 0, after = 1), ‘control or impact’ (i.e. buffer zone or no-take zone; buffer zone = 0, no-take zone = 1), and the interaction between the two (the treatment effect BAxCI). $\mathbf{b}_j$ was the vector of random effects for each of the 15 surveyed sites. The sites were similar in bathymetry and other physical characteristics, but they differed in fishery uses (COBI, unpublished data), so ‘site’ was included as a random effect to account for the unobserved variation and panel nature of these grouped transects. As a random effect, each site was given its own variance and co-variance matrix (**Ψ**) parsing it from the residual global variance ($\boldsymbol{\varepsilon}_{ij}$). Two models for each of the seven species were run, first using biomass (g/m^2) and then density (m^{-2}) as the dependent variable. Log transformations were employed on biomass data and square-root transformations were employed on abundance ratios to satisfy normality assumptions; models were checked for deviations from homoscedasticity assumptions and none were found.

Our ability to detect an effect of the reserve may be confounded by edge effects, since the control and no-take sites abut and may directly influence each other (Stewart-Oaten et al. 1986). We ran additional models for the biomass and density of each species with a covariate for distance from the no-take zone border included in the design matrix $\mathbf{X}_i$. This covariate, 'distance from border' is zero before the reserve is established, and post-reserve is the positive distance from the nearest border for sites within the no-take zone, and the negative distance from the nearest border for sites outside the no-take zone. These models were run both with and without site-specific random effects. To account for edge effects in another way, we dropped data from sites within 500 meters of the no-take zone border, and re-ran mixed model analyses for all species.

The reserve could also have affected different size classes of fish in different ways. We tested this by splitting our observations of *M. rosacea* into two size classes: spawning size and juvenile. Juveniles were defined as individuals 30 cm or less total length, as fish below this size have not been found in leopard grouper spawning aggregations (Diaz-Uribe et al. 2001). As mentioned above, the no-take zone protects a spawning aggregation of leopard grouper, where spawners could otherwise be easily exploited. Juveniles are overall too small to be preferentially targeted. Thus, the intended direct effect of the reserve is expected to be observed on the spawning age class. If the reserve increases the number of spawners, an indirect effect of the reserve on recruits could be observed with a lag. Even if the reserve protects the spawning class just long enough to increase the number of recruits, we may expect to see an increase in juvenile recruits that is not confined to the no-take zone. We also analyzed mixed models for juvenile and spawning age class leopard grouper with distance from the no-take zone border as a covariate.

From the recreational angler trip data we analyzed the 14,894 trips the anglers logged to the volunteer rescue service between 2000 and 2008. Because the logged trips needed to be sufficiently accurate to guide rescue efforts we have high confidence in the accuracy of destination sites reported by anglers. Our data did not allow us to discriminate the precise locations visited by anglers around SPMI on each reported trip, and interviews indicated that anglers frequently visit multiple fishing sites during a visit to SPMI. However, even if all anglers only fished in the buffer zone after reserve declaration (contrary to qualitative interviews and supplementary monitoring data), our analysis would present a conservative upper bound on fishing pressing in the no-take zone. We tested for a structural change in recreational angling use with a model of the proportion of fishing trips taken to SPMI before and after the announcement in 2005 (Chow 1960, Fujitani et al. 2012). A structural change is a discrete break in the best fitting parameters for the system concomitant with a mechanism for such change (e.g., new regulations). To analyze the nature of the behavioral change, we employed a binomial logit model to explore how the reserve affected the propensity of at least one angling boat from the community to take a trip to SPMI on a day that any angler chose to take a trip.

$$\Pr(z = 1) = \Lambda(\mathbf{X}\boldsymbol{\beta})$$

z is a fleet-level indicator of whether at least one boat took a trip to SPMI on a given day; this is because interview data indicated that anglers share boats and fish in groups for safety, particularly to distant fishing sites such as SPMI. The predictors used to test the hypothesis that reserve announcement affected trip behavior in the design matrix **X** included days since the reserve announcement and days since the announcement squared to look at non-linear behavior over time. Predictors in **X** accounting for other factors besides the reserve that affected the propensity of anglers to travel to SPMI were month and weekday versus weekend as fixed effects, and year and daily maximum wind speed as slope variables. These captured the behavioral responses of anglers to variation in weather, safety of open water trips, seasonal non-fishing opportunities, seasonal variation in the composition and abundance of targeted species, and gas prices (Fujitani et al. 2012). The vector $\boldsymbol{\beta}$ contained the parameter estimates from the predictors, and Λ is the logistic cumulative distribution function.

3. Results

Scatter plots of mean (± SE) density (Fig. 3) and biomass (Fig. S1) for the seven species showed little difference between control (white circles) and impact (filled squares) sites, particularly following reserve implementation. Regression analysis indicated that five of seven species (*M. rosacea*, *Cephalopholis panamensis*, *Lutjanus argentiventris*, *Hoplopagrus guentherii*, and *Mycteroperca prionura*) decreased significantly in both abundance and biomass in the time period following reserve implementation (Table 2), though no difference was seen between the control and impact areas.

Significant interaction effects (BA×CI; Table 2) were observed for the biomass and abundance of several species; however these indicated a negative effect of the reserve and/or were an artifact of their rarity in the sample. *Epinephelus labriformis* were more abundant within the no-take zone than outside, but declined in abundance faster within the no-take zone compared to the buffer zone. The remaining two species with significant interaction effects (*M. prionura* and *H. guenterii*) had very low observed counts due to their rarity in the system (Fig. 3). Only three of 43 *M. prionura* were observed in the no-take zone during the survey, and none were observed in the no-take zone following reserve implementation. Only sixteen *H. guenterii* were recorded in the time series, and only three were counted after the reserve was implemented.

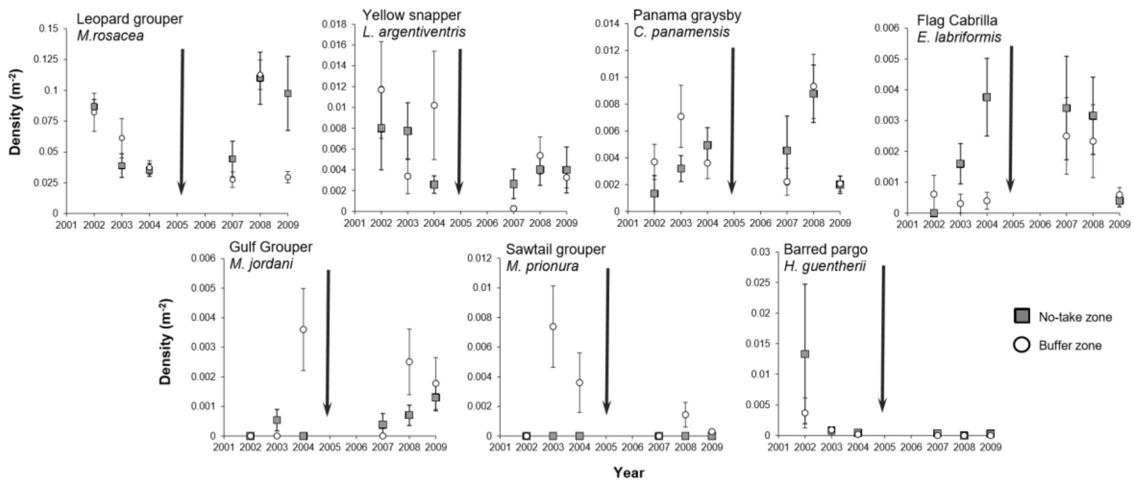

Fig. 3. Scatter plots of mean (±SE) density of seven commercially important species in the no-take zone and the buffer zone between 2002 and 2009 show no clear effect of the reserve on species preferentially targeted by fishing. The arrows indicate the announcement of the reserve.

Table 2. Parameter values and standard errors from linear mixed before-after control-impact models to detect the effects of the SPMI reserve on the square-root transformed density and log-normal transformed biomass of seven species targeted by fishing.

Density

	M. rosacea			*C. panamensis*			*E. labriformis*			*L. argentiventris*		
Parameter	β	SE	p-value	β	SE	p-value	β	SE	p-value	β	SE	p-value
Intercept	0.248	0.036	<0.001	0.052	0.010	<0.001	0.007	0.006	0.243	0.054	0.012	<0.001
BA	-0.105	0.035	0.003	-0.032	0.010	0.002	0.001	0.006	0.893	-0.026	0.011	0.015
CI	-0.041	0.057	0.484	-0.012	0.016	0.470	0.022	0.009	0.026	-0.005	0.019	0.804
BA*CI	0.075	0.060	0.215	0.015	0.017	0.395	-0.020	0.009	0.036	-0.011	0.019	0.576

	H. guentherii			*M. jordani*			*M. prionura*		
Parameter	β	SE	p-value	β	SE	p-value	β	SE	p-value
Intercept	0.031	0.014	0.033	0.013	0.006	0.027	0.028	0.009	0.002
BA	-0.008	0.003	0.003	-0.002	0.006	0.703	-0.016	0.004	<0.001
CI	0.000	0.023	0.984	-0.011	0.009	0.243	-0.028	0.014	0.074
BA*CI	-0.047	0.006	<0.001	0.008	0.010	0.453	0.016	0.008	0.046

Biomass

	M. rosacea			*C. panamensis*			*E. labriformis*			*L. argentiventris*		
Parameter	β	SE	p-value	β	SE	p-value	β	SE	p-value	β	SE	p-value
Intercept	3.696	0.305	<0.001	0.356	0.065	<0.001	0.115	0.078	0.143	0.860	0.151	<0.001
BA	-2.364	0.294	<0.001	-0.240	0.066	<0.001	-0.021	0.083	0.805	-0.474	0.132	<0.001
CI	-0.225	0.488	0.652	-0.089	0.102	0.398	0.361	0.120	0.010	-0.101	0.245	0.686
BA*CI	0.491	0.511	0.337	0.086	0.109	0.429	-0.322	0.128	0.012	-0.191	0.242	0.430

	H. guentherii			*M. jordani*			*M. prionura*		
Parameter	β	SE	p-value	β	SE	p-value	β	SE	p-value
Intercept	0.879	0.374	0.019	0.731	0.232	0.002	0.654	0.224	0.004
BA	-0.304	0.068	<0.001	-0.355	0.217	0.103	-0.456	0.077	<0.001
CI	0.034	0.592	0.955	-0.647	0.373	0.107	-0.654	0.358	0.091
BA*CI	-1.403	0.140	<0.001	0.545	0.385	0.158	0.456	0.156	0.004

Our tests to account for edge effects between the no-take and control zones did not qualitatively change results. For example, for the most common species, *M. rosacea*, dropping sites less than 500 meters from a no-take zone border (Table S1; I.), or including distance from the nearest border as a covariate (Table S1. II.) yielded similar results as the original mixed model of no significant effect of the reserve. Models accounting for size-class similarly did not show a positive effect from the reserve (Table S1; III.), and we did not find an increase in juveniles over time that could have indicated an indirect effect of the reserve through recruitment (Table S1; IV.).

We found a significant structural change in the proportion of recreational angling trips to SPMI before and after the reserve was announced, indicating the reserve had an effect on recreational angling use of the island. The nature of this effect was explained in the binomial logit model, showing that the reserve had the effect of reducing the propensity of the fleet to visit SPMI (days since impact -1.61×10^{-3}, p-value = 0.003). Though the model indicated the reserve had a negative first-order effect on the propensity to travel to SPMI, this effect grew weaker with time (days since impact squared 1.34×10^{-6}, p-value = 0.001). Independent of any reserve effect, results suggest that the propensity of anglers to travel to SPMI was increasing over time and overall use of the reserve is projected to increase (year coefficient 0.18, p-value < 0.001). These results are summarized in Figure 4, which uses parameters estimated from the model to project the statistical expectation of what would have been observed had the reserve not been established, and compare it to the reserve's effect on recreational angling use (also see Fujitani et al. 2012). Increasing use of the reserve area over time can be seen in the open squares of Figure 4 (open squares connected by a dotted line post-2005 represent the projected counterfactual scenario) (Heckman 2010). The open circles show the reserve effect on the propensity to visit SPMI. The initial dip in angler propensity reflects the negative first-order effect: the reserve initially reduced the propensity of anglers to visit. However, the positive second-order effect indicates the conservation effect of the reserve was diminishing over time, and thus the reserve and no-reserve scenarios converge over time. In summary, though anglers initially decreased visits to SPMI, in a few years their visitation behavior was the same as if the reserve had never existed.

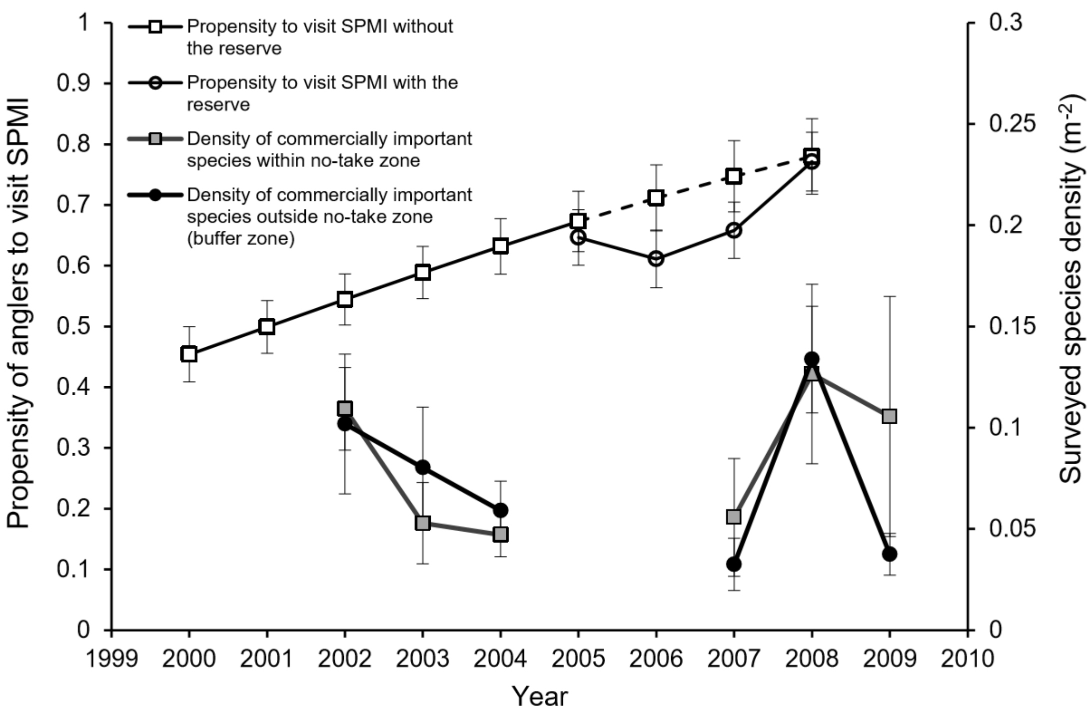

Fig. 4. Trends in projected propensities for anglers to travel to SPMI (left axis) and density of surveyed species inside and outside of the SPMI no-take zone (right axis). Travel projections (with 95% projection intervals) were estimated from a logistic regression model using average visitation for a weekday in June, the most popular month for trips to SPMI. Open squares indicate the propensity to visit SPMI without a marine reserve; the dotted line shows post-2004 values that are the statistically expected propensity to visit SPMI with reserve effects set to zero. Open circles are the propensity to visit SPMI with the reserve effects estimated by the logistic regression model. The right-hand axis corresponds to the total density of the surveyed species inside and outside of SPMI's no-take zone, with 95% confidence intervals.

4. Discussion

We did not find evidence that the SPMI no-take zone met stated reserve goals of protecting and enhancing biological resources by increasing fish stocks, though marine reserves elsewhere have demonstrated to be effective for species sharing the exploitation status and life history characteristics of the seven monitored species (Micheli et al. 2004, Claudet et al. 2010). Further, reserves have demonstrated significant stock increases in a shorter period following the initiation of protection than the monitored period for the SPMI reserve (Halpern and Warner 2002, but see Edgar et al. 2014). *M. prionura* and *E. labriformis* appeared to decline faster within the no-take zone than in the control. Five other species showed their stocks deteriorating over time, with no significant protective effect of the reserve (BAxCI; Table 2). This may be explained by the large number of zeroes in transects overall for some species— though the biological monitoring program was intensive and costly, uncommon species were infrequently observed, making it difficult to discern trends in their stocks (Table 1). Further, we were unable to detect an effect of the no-take zone for leopard grouper despite the rich dataset available (Table 1). Indeed, mean leopard grouper abundance and biomass in the period before the reserve was implemented was higher than after protection went into effect (Table 2; S1) despite the expected benefit of protecting a spawning aggregation site (Sala et al. 2003, Russel and Sadory 2005). Other studies of marine protected areas in the GoC have shown similar results, with no-take zones showing no effect or even significant declines compared to control sites (Cudney-Bueno et al. 2009, Stamieszkin et al. 2009). These biological results, taken alone, could be interpreted as evidence that marine reserves are not a viable management tool in the GoC. However these data alone provide no information to address alternate hypotheses and provide advice to management.

Conversely, our analysis of recreational angling showed that the SPMI reserve had a significant negative effect on fishing trips to the island relative to the projected level of trips without the reserve, and thus that a reserve could potentially work to reduce fishing pressure in this system relative to the case of no reserve. The reserve reduced the number of visits by members of an important stakeholder group relative to the statistical expectation of fishing effort had the reserve not been established (Fig. 4; open circles for 2006 and 2007). Interviews with anglers supported the theory that the reserve had increased the implicit cost and decreased the incentive to travel to SPMI; anglers expressed worries about fines and boat seizures.

Our analysis provided a possible explanation as to why we did not observe increased fish stocks despite the *relative* reduction in fishing effort – the reserve did not decrease the *absolute* level of fishing effort. Our model showed that independent of the establishment of the no-take zone, trips to SPMI had been increasing over time (Fig. 4; open squares). Interviews with recreational anglers indicated that this trend is due to the perception that SPMI has some of the best fishing in the region for the highest valued targets. Anglers cited advances in boat safety and technology (especially increases in the sizes of boats and motors) as a reason why more boats are making the remote, exposed, and treacherous crossing to SPMI. Given this overall trend of increasing use of SPMI, the reserve only dampened this progression (Fig. 4). Though the reserve reduced the otherwise expected increase in recreational angling use of the island over two years, it appears any alleviated fishing pressure was insufficient to reverse stock declines. The effect of the reserve to reduce fishing diminished over time, so that by 2008 realized angler behavior was indistinguishable from the level of use that would have been expected had the reserve never existed (Fig. 4).

Our results raise two important questions about the SPMI reserve that are likely applicable to other management proposals. First, marine reserves and other natural resource management strategies are intended as a press perturbation management intervention, but SPMI seems to have acted as a pulse, only reducing relative angler visitation (compared to the counterfactual) for a short period. Could pulse effects be worthwhile investments? Drury and Lodge (2009) suggest that they might be if the ecosystem displays multiple stable states. However, the human behavioral response may counter such pulse interventions (Horan et al. 2011), requiring enduring institutional changes. Second, what should be considered a biological success? Often reserves are considered successful if they stop or reverse population declines. Is slowing downward trends sufficient to justify a reserve? While stocks did not increase following the establishment of SPMI, it is possible that they declined less rapidly than they would have without the reserve. Depending on management interpretations, this may meet the stated SPMI reserve goal of protecting biological resources.

Interviews with anglers and COBI staff provided insight into why the reserve acted as a pulse instead of press disturbance to angling behavior. Anglers learned that the reserve was not being enforced. During interviews conducted in 2009 anglers still expressed confusion and wariness about the rules of the reserve, but all subjects knew that the rules of the reserve had not been enforced to date. Many expressed a desire for reserve rules to be enforced as long as it was done fairly among all stakeholders.

CONANP/COBI's human use monitoring found a small number of commercial and recreational anglers in the no-take zone every year until monitoring ended in early 2008 (Meza et al. 2008). All boats discovered in the no-take zone were warned and asked to leave, after being asked if they knew they were in a no-take zone, to which 71% responded 'yes' (Meza et al. 2008).

Enforcement (Kritzer 2004, Guidetti et al. 2008, Kelaher et al. 2015), as well as compliance (Pollnac et al. 2010, Dalton et al. 2015) is critical to marine reserve success and meta-analyses that found significant benefits to marine reserves had as a selection criterion the reasonably successful exclusion of fishing (Cote et al. 2001, Halpern 2003). *Ex ante*, this may be what one would expect, but human responses to new institutions can be difficult to predict. Our detailed recreational angling dataset allowed us to empirically document angler response to weak enforcement. The SPMI reserve, like others in the GoC, is limited in management resources and faces challenges mobilizing across the multiple government bureaus that have specific jurisdictions, who need to coordinate for effective outreach and law enforcement (Cudney-Bueno et al. 2009). To date the rules of the SPMI no-take zone have not been enforced beyond warnings (J. Torre, pers. comm.). Though reserves have been suggested as fishery management strategies for developing countries (Polunin 2002) and regions lacking strong institutional control (Agardy 1997, National Research Council 2001), institutions are critical for management and law enforcement. Community-based management is a popular conservation paradigm in the absence of top-down administration, but these initiatives also require strong community institutions to be successful (Barrett et al. 2001, Cinner et al. 2012). These results are broadly relevant to the global management of marine reserves, which like other spatial rights-based management approaches (e.g. exclusive zones, TURFS) require oversight, enforcement, and stakeholder cooperation (Claudet and Guidetti 2010).

Reserves are human institutions, and policies cascade through changes in human use to affect fish stocks. Monitoring intermediate points along this cascade (Fig. 1) provides information on how policy is functioning (Hilborn 2007). Insufficient time for a biological response, as well as process and measurement error (Fig. 1) can obfuscate the effect of a reserve on stocks. As human use changes occur higher up on the cascade of socio-ecological reserve processes (Fig. 1; II.), human use monitoring can meaningfully be conducted sooner, potentially immediately after the reserve is established. The way that people respond to management is subject to implementation error (Fig. 1; II.), but may be spared the process error and response lag of biological systems (Fig. 1; IV.). Thus, monitoring the type and quantity of human use may provide an earlier signal of reserve processes with less error. Further, unlike traditional biological monitoring (e.g., stock size metrics) that contributes limited information about mechanisms that could be prohibiting recovery, monitoring human activity provides information about a potential target for improving management. Without positive signals from a biological monitoring program, managers might conclude that monitoring should continue until a reserve effect is detected. However, there is no consensus on how long one should continue to monitor a reserve (Sainsbury 1991, Gerber et al. 2005). In deciding the type and length of monitoring, it is important to consider the cost of monitoring, the value of data, and the implications of the decisions the monitoring data will inform including potential stock losses (Dayton 1998, Field et al. 2004, Gerber et al. 2007, Hansen and Jones 2008, Fenichel and Hansen 2010). Human use monitoring provides information on how institutions are functioning. If no effect of a management program on human use of an area is detected, managers can immediately use this cue to both change tactics (e.g., outreach, enforcement) and gather more specific information as to why.

Besides the data-driven decision support reasons to monitor human use data in tandem with biological data, positive social and economic outcomes are explicitly or implicitly a broader target of natural resource management programs (Christie et al. 2003, Hicks et al. 2016) as well as essential to the success of management programs (Brechin et al. 2002, Cinner et al. 2012). Ecological, social, and economic sustainability goals can be synergistic, as marine resources provide coastal communities with subsistence, livelihoods, and other social and cultural ecosystem services (Daw et al. 2015). Social and economic characteristics and outcomes for marine reserve stakeholders are rightfully given increased attention and importance in reserve evaluation (Pollnac et al. 2010, Dalton et al. 2015, Gurney et al. 2015), but monitoring direct human behavioral responses to reserves remains rare. We demonstrate how monitoring data on human activity is also an essential part of fisheries management not only during the selection and implementation of a management strategy (Smith and Wilen 2002, Christie et al. 2003, Mascia et al. 2003, Hilborn 2007, Ban et al. 2009) but also, as we have demonstrated, in the evaluation of its effects.

Monitoring biological and human use data in tandem provide information on underlying processes and can uncover latent trends that enhance our understanding of a system's behavior. Thus, this monitoring strategy is valuable even if a reserve is found to be meeting stock enhancement goals, because it can explain and sustain that success, and transfer lessons learned to other systems. By understanding the nature and direction of human use changes and how they influence biological populations, managers can make informed decisions that increase the likelihood that management programs are successful.

Acknowledgements

We thank Dave White, Jamie Jerdee, Ana Luisa Figueroa, Jennifer Duberstein, the monitoring team of divers, and the Prescott College Kino Bay Field Station. MF was supported by Comunidad y Biodiversidad, A.C. LRG was supported as a Sabbatical Fellow at the National Center for Ecological Analysis and Synthesis, a Center funded by NSF (Grant #EF-0553768), the University of California, Santa Barbara, and the State of California. MF and EF were partially supported by NOAA-Saltonstall Kennedy grant NA09NMF4270098.

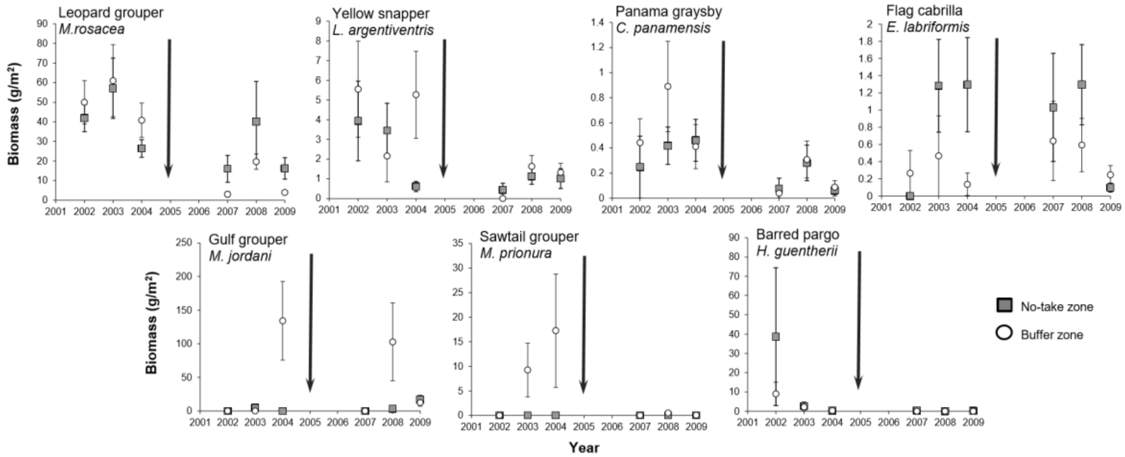

Fig. S1. Scatter plots of mean (±SE) biomass of seven commercially important species in the no-take zone and the buffer zone between 2002 and 2009 show no clear effect of the reserve on species preferentially targeted by fishing. The arrows indicate the announcement of the reserve.

Table S1. Parameter values and standard errors from alternate models to detect an effect of the no-take zone on the density and biomass of *M. rosacea*.

I. Drop sites < 500 meters from the no-take boundary, site random effects						
	Density			*Biomass*		
Parameter	β	SE	p-value	β	SE	p-value
Intercept	0.25	0.038	< 0.001	3.707	0.329	< 0.001
BA	-0.113	0.036	0.002	-2.43	0.294	< 0.001
CI	0.003	0.096	0.978	-0.067	0.808	0.936
BACI	0.046	0.083	0.581	0.403	0.669	0.547

II. Distance from border as covariate						
	Density			*Biomass*		
Parameter	β	SE	p-value	β	SE	p-value
Intercept	0.218	0.03	< 0.001	3.429	0.243	< 0.001
BA	-0.003	0.039	0.947	-1.555	0.323	< 0.001
CI	-0.029	0.045	0.518	-0.092	0.369	0.804
BACI	-0.032	0.058	0.58	-0.341	0.479	0.477
Distance from Border	0.042	0.017	0.012	0.297	0.137	0.031

III. Spawning class (> 30cm total length)						
	With site random effects			*With distance from border as covariate*		
Parameter	β	SE	p-value	β	SE	p-value
Intercept	0.159	0.02	< 0.001	0.154	0.019	< 0.001
BA	-0.121	0.021	< 0.001	-0.107	0.025	< 0.001
CI	-0.007	0.031	0.833	-0.009	0.028	0.738
BACI	0.033	0.033	0.315	0.025	0.037	0.496
Distance from Border				0.004	0.011	0.72

IV. Juveniles (< 30cm total length)						
	With site random effects			*With distance from border as covariate*		
Parameter	β	SE	p-value	β	SE	p-value
Intercept	0.17	0.03	< 0.001	0.146	0.025	< 0.001
BA	-0.044	0.03	0.143	0.048	0.033	0.149
CI	-0.053	0.048	0.29	-0.04	0.038	0.298
BACI	0.073	0.051	0.151	-0.031	0.05	0.538
Distance from Border				0.042	0.014	0.003

References

Centro de Investigación y Capacitación Rural, AC (CEDICAR), Comisión Nacional de Áreas Naturales Protegidas (CONANP). Indicadores de efectividad de la gestión de la Reserva de la Biósfera Isla San Pedro Mártir. Guaymas, Mexico; 2007.

Agardy, M. T. 1997. Marine Protected Areas and Ocean Conservation. Academic Press, San Diego.

Babcock, R. C., N. T. Shears, A. C. Alcala, N. S. Barrett, G. J. Edgar, K. D. Lafferty, T. R. McClanahan, and G. R. Russ. 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. PNAS **107**:18256-18261.

Ban, N. C., G. J. A. Hansen, M. Jones, and A. C. J. Vincent. 2009. Systematic marine conservation planning in data-poor regions: Socioeconomic data is essential. Marine Policy **33**:794-800.

Barrett, C. B., K. Brandon, C. Gibson, and H. Gjertsen. 2001. Conserving tropical biodiversity amid weak institutions. Bioscience **51**:497-502.

Bonaldo, R. M., M. M. Pires, P. R. J. Guimaraes, A. S. Hoey, and M. E. Hay. 2017. Small Marine Protected Areas in Fiji Provide Refuge for Reef Fish Assemblages, Feeding Groups, and Corals. PLoS One **12**:e0170638.

Brechin, S. R., P. R. Wilshusen, C. L. Fortwang, and P. C. West. 2002. Beyond the square wheel: toward a more comprehensive understanding of biodiversity conservation as a social and political process. Society and Natural Resources **15**:41-64.

Chow, G. C. 1960. Tests of equality between sets of coefficients in two linear regressions. Econometrica **28**:591-605.

Christie, P., B.J. McCay, M.L. Miller, C. Lowe, A.T. White, R. Stoffle, D.L. Fluharty, L.T. McManus, R. Chuenpagdee, C. Pomeroy, D.O. Suman, B.G. Blount, D. Huppert, R.L.V. Eisma, E. Oracion, K. Lowry, and R.B. Pollnac. 2003. Toward developing a complete understanding: a social science research agenda for marine protected areas. Fisheries **28**:22-26.

Cinner, J. E., T. R. McClanahan, M. A. MacNeil, N. A. Graham, T. M. Daw, A. Mukminin, D. A. Feary, A. L. Rabearisoa, A. Wamukota, N. Jiddawi, S. J. Campbell, A. H. Baird, F. A. Januchowski-Hartley, S. Hamed, R. Lahari, T. Morove, and J. Kuange. 2012. Comanagement of coral reef social-ecological systems. Proc Natl Acad Sci U S A **109**:5219-5222.

Claudet, J., and P. Guidetti. 2010. Improving assessments of marine protected areas. Aquatic Conservation: Marine and Freshwater Ecosystems **20**:239-242.

Claudet, J., C. W. Osenberg, P. Domenici, F. Badalamenti, M. Milazzo, J. M. Falcón, I. Bertocci, L. Benedetti-Cecchi, J.-A. García-Charton, R. Goñi, J. A. Borg, A. Forcada, G. A. d. Lucia, Á. Pérez-Ruzafa, P. Afonso, A. Brito, I. Guala, L. L. Diréach, P. Sanchez-Jerez, P. J. Somerfield, and S. Planes. 2010. Marine reserves: fish life history and ecological traits matter. Ecological Applications **20**:830-839.

Cote, I. M., I. Mosqueira, and J. D. Reynolds. 2001. Effects of marine reserve characteristics on the protection of fish populations: a meta-analysis. Journal of Fish Biology **59**:179-189.

Cudney-Bueno, R., L. Bourillón, A. Sáenz-Arroyo, J. Torre-Cosío, P. Turk-Boyer, and W. W. Shaw. 2009. Governance and effects of marine reserves in the Gulf of California, Mexico. Ocean & Coastal Management **52**:207-218.

Dalton, T., R. Pollnac, and G. Forrester. 2015. Investigating Causal Pathways Linking Site-Level Characteristics, Compliance, and Ecological Performance in Caribbean MPAs. Coastal Management **43**:329-341.

Daw, T. M., S. Coulthard, W. W. Cheung, K. Brown, C. Abunge, D. Galafassi, G. D. Peterson, T. R. McClanahan, J. O. Omukoto, and L. Munyi. 2015. Evaluating taboo trade-offs in ecosystems services and human well-being. Proc Natl Acad Sci U S A **112**:6949-6954.

Dayton, P. K. 1998. Reversal of the burden of proof in fisheries management. Science **279**:821-822.

Devillers, R., R. L. Pressey, A. Grech, J. N. Kittinger, G. J. Edgar, T. Ward, and R. Watson. 2015. Reinventing residual reserves in the sea: are we favouring ease of establishment over need for protection? Aquatic Conservation: Marine and Freshwater Ecosystems **25**:480-504.

Diaz-Uribe, J. G., J. F. Elorduy-Garay, and M. T. Gonzalez-Valdovinos. 2001. Age and growth of the Leopard Grouper, Mycteroperca rosacea, in the Southern Gulf of California, Mexico Pacific Science **52**:207-218.

Drury, K. L., and D. M. Lodge. 2009. Using mean first passage times to quantify equilibrium resilience in perturbed intraguild predation systems. Theoretical Ecology **2**:41-51.

Edgar, G. J., R. D. Stuart-Smith, T. J. Willis, S. Kininmonth, B. S. C., B. S., N. S. Barrett, M. A. Becerro, A. T. F. Bernard, J. Berkhout, C. D. Buxton, S. J. Campbell, A. T. Cooper, M. Davey, S. C. Edgar, G. Forsterra, D. E. Galvan, A. J. Irigoyen, D. J. Kushner, R. Moura, P. E. Parnell, N. T. Shears, G. Soler, E. M. A. Strain, and R. J. Thomson. 2014. Global conservation outcomes depend on marine protected areas with five key features. Nature **506**:216–220.

Emslie, M. J., M. Logan, D. H. Williamson, A. M. Ayling, M. A. MacNeil, D. Ceccarelli, A. J. Cheal, R. D. Evans, K. A. Johns, M. J. Jonker, I. R. Miller, K. Osborne, G. R. Russ, and H. P. Sweatman. 2015. Expectations and Outcomes of Reserve Network Performance following Re-zoning of the Great Barrier Reef Marine Park. Current Biology **25**:983-992.

Fenichel, E. P., J. K. Abbott, and B. Huang. 2013. Modelling angler behaviour as a part of the management system: Synthesizing a multi-disciplinary literature. Fish and Fisheries **14**:137-157.

Fenichel, E. P., and G. J. A. Hansen. 2010. The opportunity cost of information: an economic framework for understanding the balance between assessment and control in sea lamprey (Petromyzon marinus) management. Canadian Journal of Fisheries and Aquatic Sciences **67**:209-216.

Field, S. A., A. J. Tyre, N. Jonzen, J. R. Rhodes, and H. P. Possingham. 2004. Minimizing the cost of environmental management decisions by optimizing statistical thresholds. Ecology Letters **7**:669-675.

Fox, J. 2002. “Linear Mixed Models,” Appendix to An R and S-PLUS Companion to Applied Regression.

Froese, R., and D. Pauly. 2016. FishBase. World Wide Web electronic publication. www.fishbase.org.

Fujitani, M. L. 2010. The rapid assessment of a new marine reserve in the Gulf of California, Mexico. Arizona State University.

Fujitani, M. L., E. P. Fenichel, J. Torre, and L. R. Gerber. 2012. Implementation of a marine reserve has a rapid but short-lived effect on recreational angler use. Ecological Applications **22**:597-605.

Geldmann, J., L. Coad, M. Barnes, I. D. Craigie, M. Hockings, K. Knights, F. Leverington, I. C. Cuadros, C. Zamora, S. Woodley, and N. D. Burgess. 2015. Changes in protected area management effectiveness over time: A global analysis. Biological Conservation **191**:692-699.

Gelman, A., and J. Hill. 2007. Data Analysis Using Regression and Multilevel / Hierarchical Models. Cambridge University Press, New York, NY.

Gerber, L. R., M. Berger, M. A. McCarthy, and H. P. Possingham. 2005. A theory for optimal monitoring of marine reserves. Ecology Letters **8**:829-837.

Gerber, L. R., P.M. Kareiva, and J. Bascompte. 2002. The influence of life history attributes and fishing pressure on the efficacy of marine reserves. Biological Conservation **106**:11-18.

Gerber, L. R., J. Wielgus, and E. Sala. 2007. A decision framework for the adaptive management of an exploited species with implications for marine reserves. Conservation Biology **21**:1594-1602.

Green, R. H. 1979. Sampling Design and Statistical Methods for Environmental Biologists. Page 257 pp. Wiley, Chichester.

Greenstone, M., and T. Gayer. 2009. Quasi-experimental and experimental approaches to environmental economics. Journal of Environmental Economics and Management **57**:21-44.

Gribble, N. A., and J. W. Robertson. 1998. Fishing effort in the far northern section cross shelf closure area of the Great Barrier Reef Marine Park: the effectiveness of area-closures. Journal of Environmental Management **52**:53-67.

Guidetti, P. 2007. Potential of marine reserves to cause community-wide changes beyond their boundaries. Conservation Biology **21**:540-545.

Guidetti, P., M. Milazzo, S. Bussotti, A. Molinari, M. Murenu, A. Pais, N. Spano, R. Balzano, T. Agardy, F. Boero, G. Carrada, R. Cattaneo-Vietti, A. Cau, R. Chemello, S. Greco, A. Manganaro, G. N. d. Sciara, G. F. Russo, and L. Tunesi. 2008. Italian marine reserve effectiveness: Does enforcement matter? Biological Conservation **141**:699-709.

Gurney, G. G., R. L. Pressey, J. E. Cinner, R. Pollnac, and S. J. Campbell. 2015. Integrated conservation and development: evaluating a community-based marine protected area project for equality of socioeconomic impacts. Philos Trans R Soc Lond B Biol Sci **370**.

Halpern, B. S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecological Applications **31**:S117-S137.

Halpern, B. S., S. D. Gaines, and R. R. Warner. 2004. Confounding effects of the export of production and the displacement of fishing effort from marine reserves. Ecological Applications **14**:1248-1256.

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

Hamilton, S. L., J. E. Caselle, D. P. Malone, and M. H. Carr. 2010. Incorporating biogeography into evaluations of the Channel Islands marine reserve network. PNAS **107**:18272-18277.

Hansen, G. J. A., N. C. Ban, M. L. Jones, L. Kaufman, H. M. Panes, M. Yasué, and A. C. J. Vincent. 2011. Hindsight in marine protected area selection: A comparison of ecological representation arising from opportunistic and systematic approaches. Biological Conservation **144**:1866-1875.

Hansen, G. J. A., and M. L. Jones. 2008. The value of information in fisheries management. Fisheries **33**:340-348.

Heckman, J. J. 2010. Building bridges between structural and program evaluation approaches to evaluating policy. Journal of Economic Literature **48**:356-398.

Hicks, C. C., A. Levine, A. Agrawal, X. Basurto, S. J. Breslow, C. Carothers, S. Charnley, S. Coulthard, N. Dolsak, J. Donatuto, and C. Garcia-Quijano. 2016. Engage key social concepts for sustainability. Science **352**:38-40.

Hilborn, R. 2007. Managing fisheries is managing people: what has been learned? Fish and Fisheries **8**:285-296.

Horan, R. D., E. P. Fenichel, K. L. Drury, and D. M. Lodge. 2011. Managing ecological thresholds in coupled environmental–human systems. Proceedings of the National Academy of Sciences **108**:7333-7338.

Kelaher, B. P., A. Page, M. Dasey, D. Maguire, A. Read, A. Jordan, and M. A. Coleman. 2015. Strengthened enforcement enhances marine sanctuary performance. Global Ecology and Conservation **3**:503-510.

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

Kritzer, J. P. 2004. Effects of noncompliance on the success of alternative designs of marine protected-area networks for conservation and fisheries management. Conservation Biology **18**:1021-1031.

Kuznetsova, A., Brockhoff, P.B. & Christensen, R.H.B. . 2016. lmerTest: tests for random and fixed effects for linear mixed effect models. http://CRAN.R-project.org/package=lmerTest.

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

Lynch, T. P. 2006. Incorporation of recreational fishing effort into design of marine protected areas. Conservation Biology **20**:1466-1476.

Lynch, T. P. 2014. A decadal time-series of recreational fishing effort collected during and after implementation of a multiple use marine park shows high inter-annual but low spatial variability. Fisheries Research **151**:85-90.

Mascia, M. B., P. J. Brosius, J. P. Dobson, B. C. Forbes, L. Horowitz, M. A. McKean, and N. J. Turner. 2003. Conservation and the social sciences. Conservation Biology **17**:649-650.

McClanahan, T. R. 2010. Effects of fisheries closures and gear restrictions on fishing income in a Kenyan coral reef. Conservation Biology **24**:1519-1528.

McDonald, T. L., W.P. Erickson, and L. L. McDonald. 2000. Analysis of count data from before-after control-impact studies. Journal of Agricultural, Biological, and Environmental Statistics **5**:262-279.

Meza, A., C. Moreno, J. Torre, and M. Rojo. 2008. Usos Humanos en la Reserva de la Biosfera Isla San Pedro Mártir. Internal document. Comunidad y Biodiversidad, A.C. (COBI) Guaymas, Mexico www.cobi.org.mx.

Micheli, F., B. S. Halpern, L. W. Botsford, and R. R. Warner. 2004. Trajectories and correlates of community change in no-take marine reserves. Ecological Applications **14**:1709-1723.

National Research Council. 2001. Marine protected areas: tools for sustaining ocean ecosystems. National Academy Press, Washington, D.C.

Orley, A., and D. Card. 1985. Using the longitudinal structure of earnings to estimate the effect of training programs. The Review of Economics and Statistics **67**:648-660.

Perez-Mellado, J. L., and L. T. Findley. 1985. Evaluacion de la Ictiofauna del camarón capturado en las costas de Sonora y norte de Sinaloa, México. Pages 201-254 *in* A. Yañez-Arancibia, editor. Recursos Pesqueros Potenciales de México: La pesca acompañante del camarón. Programa Universitario de Alimentos/Instituto de Ciencias del Mar y Limnologia, México.

Pinheiro, J., D. Bates, S. DebRoy, D. Sarkar, and R Core Team. 2016. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-128, http://CRAN.R-project.org/package=nlme.

Poder Ejecutivo Federal. 2002. DECRETO por el que se declara área natural protegida con la categoría de reserva de la biosfera, la región denominada Isla San Pedro Mártir, ubicada en el Golfo de California, frente a las costas del Municipio de Hermosillo, Estado de Sonora, con una superficie total de 30,165-23-76.165 hectáreas. Diario Oficial de la Federación, June 13, 2002:6-14.

Pollnac, R., P. Christie, J. E. Cinner, T. Dalton, T. M. Daw, G. E. Forrester, N. A. J. Graham, and T. R. McClanahan. 2010. Marine reserves as linked social–ecological systems. Proceedings of the National Academy of Sciences **107**:18262-18265.

Pollnac, R., B. R. Crawford, and M. L. G. Gorospe. 2001. Discovering factors that influence the success of community based marine protected areas in the Visayas, Philippines. Ocean and Coastal Management **44**:683-710.

Polunin, N. V. C., editor. 2002. Marine protected areas, fish and fisheries. Blackwell Science, New Jersey, USA.

Ramírez, R. M., and C. M. Rodríguez. 1990. Specific composition of the small scale fishery capture at Isla Cerralvo, BCS, México. Investigaciones Marinas CICIMAR **5**:137–141.

Roberts, C., G. Branch, R. Bustamante, J.C. Castilla, J. Dugan, B. Halpern, H. Leslie, K. Lafferty, J. Lubchenco, D. McArdle, M. Ruckleshaus, and R. Warner. 2003. Application of ecological criteria in selecting marine reserves and developing reserve networks. Ecological Applications **13**:S215-S228.

Roberts, C. M. 2000. Selecting marine reserve locations: optimality versus opportunism. Bulletin of Marine Science **66**:581-592.

Roberts, C. M., J. A. Bohnsack, F. Gell, J. P. Hawkins, and R. Goodridge. 2001. Effects of marine reserves on adjacent fisheries. Science **294**:1920-1923.

Rubin, D. B. 1974. Estimating causal effects of treatments in randomized and nonrandomized studies. Journal of Educational Psychology **66**:688-701.

Russel, M., and Y. Sadory. 2005. "Eastern Pacific – Sea of Cortez." Society For the Conservation of Reef Fish Aggregations Newlsletter May 2005, Number 7.

Saenz-Arroyo, A., J. Torre, L. Bourillon, and M. Kleiberg. 2005. A community-based marine reserve network in North-western Mexico. 2005 in Proceedings of the Symposium and Workshop of the North American Marine Protected Areas Network. Full documents available in Internet www.cec.org. Loreto, Baja California Sur, México. March 1 - 3. North American Commission for Environmental Cooperation. 19 pp.

Sainsbury, K. J. 1991. Application of an experimental approach to management of a tropical multispecies fishery with highly uncertain dynamics. . ICES March Science Symposium **193**:301-320.

Sala, E. 2005. Eastern Pacific – Sea of Cortez. Society For the Conservation of Reef Fish Aggregations Newsletter **7**:5.

Sala, E., O. Aburto-Oropeza, G. Paredes, and G. Thompson. 2003. Spawning aggregations and reproductive behaviour of reef fishes in the Gulf of California. Bulletin of Marine Science **72**:103–121.

Sala, E., O. Aburto-Oropeza, M. Reza, G. Paredes, and L. G. López-Lemu. 2004. Fishing down coastal food webs in the Gulf of California. Fisheries **29**:19–25.

Smith, M. D., and J. E. Wilen. 2002. The marine environment: fencing the last frontier. Review of Agricultural Economics **24**:31-42.

Smith, M. D., and J. E. Wilen. 2003. Economic impacts of marine reserves: the importance of spatial behavior. Journal of Environmental Economics and Management **46**:183-206.

Smith, M. D., J. Zhang, and F. C. Coleman. 2008. Econometric modeling of fisheries with complex life histories: Avoiding biological management failures. Journal of Environmental Economics and Management **55**:265-280.

Stamieszkin, K., J. Wielgus, and L. R. Gerber. 2009. Management of a marine protected area for sustainability and conflict resolution: Lessons from Loreto Bay National Park (Baja California Sur, Mexico). Ocean & Coastal Management **52**:449-458.

Stewart-Oaten, A., W. W. Murdoch, and K. R. Parker. 1986. Environmental impact assessment: "pseudoreplication" in time? Ecology **67**:929-940.

Thrush, S. F., and P. K. Dayton. 2002. Disturbance to Marine Benthic Habitats by Trawling and Dredging: Implications for Marine Biodiversity. Annual Review of Ecology and Systematics **33**:449-473.

UNEP-WCMC. 2008. National and Regional Networks of Marine Protected Areas: A Review of Progress. Cambridge, UNEP-WCMC.

Watson, G. J., J. M. Murray, M. Schaefer, and A. Bonner. 2015. Successful local marine conservation requires appropriate educational methods and adequate enforcement. Marine Policy **52**:59-67.

White, E. R., M. C. Myers, J. M. Flemming, and J. K. Baum. 2015. Shifting elasmobranch community assemblage at Cocos Island--an isolated marine protected area. Conservation Biology **29**:1186-1197.

Wilen, J. E., M. D. Smith, D. Lockwood, and L. W. Botsford. 2002. Avoiding surprises: incorporating fisherman behavior into management models. Bulletin of Marine Science **70**:553-575.

Woodcock, P., B. C. O'Leary, M. J. Kaiser, and A. S. Pullin. 2017. Your evidence or mine? Systematic evaluation of reviews of marine protected area effectiveness. Fish and Fisheries **18**:668-681.

Zuur, A. F., E. N. Ieno, N. J. Walker, A. A. Saveliev, and G. M. Smith. 2009. Mixed Effects Models and Extensions in Ecology with R. Springer, New York.