Ecological Applications, 22(2), 2012, pp. 597–605

Implementation of a marine reserve has a rapid but short-lived effect on recreational angler use

Marie L. Fujitani,[1,4] Eli P. Fenichel[1,2] Jorge Torre,[3] and Leah R. Gerber[1]

[1]*School of Life Sciences, Arizona State University, Tempe, Arizona 85287-4501 USA*
[2]*EcoSERVICES Group, Arizona State University, Tempe, Arizona 85287-4501 USA*
[3]*Comunidad y Biodiversidad A.C., Boulevard Agua Marina 297, Entre Jaiba y Tiburon, Colonia Delicias, Guaymas, Sonora 85420 Mexico*

Abstract. Changes in human behavior are a precursor to measurable impacts of no-take marine reserves. We investigated changes in recreational fishing site selection in response to the 2005 announcement of enforcement in a marine reserve in the Gulf of California, Mexico. We used a novel data set of daily self-reported boating destinations from emergency rescue logbooks for a recreational angling community from 2000 to 2008. Because the reserve system has no experimental control, we modeled the data two ways to test for robustness to model specification. We tested for changes in human fishing behavior with regression and fit a fleet-level discrete choice model to project a counterfactual scenario. The counterfactual is the statistically constructed ex post expectation of the human behavior we would have observed if the reserve never existed. We included month and year fixed effects in our models to account for seasonal and interannual fluctuations in fishing behavior and catch rates. We detected a decrease in reserve use compared to the counterfactual, indicating that the reserve rapidly experienced a decrease in visitation. However, the reserve's effect to reduce trips diminished with time. These results indicate that the reserve is unlikely to meet its ecological goals without institutional changes that enhance compliance. This illustrates the value of human use data to understanding the processes underlying marine reserve function. We suggest that managers should consider human use with the same frequency, rigor, and tools as they do fishery stocks. Marine reserves directly affect people, and understanding human behavioral responses to marine reserves is an important step in marine reserve management.

Key words: Bahia de Kino, Gulf of California; human behavior; marine protected areas; marine reserves; monitoring; no-take areas; recreational fishing; San Pedro Mártir Island, Mexico.

Introduction

Worldwide, marine reserves are designed and assessed (Wells et al. 2008, Wood et al. 2008) focusing on biological criteria (e.g., Halpern and Warner 2002, Gerber et al. 2003, Roberts et al. 2003, D'Agrosa et al. 2007, Gerber et al. 2007). However, theoretical results and empirical evidence from commercial and for-hire recreational fisheries suggest that in order for marine reserves to meet biological goals, management strategies must take into account how fishers respond to marine reserves and the incentives that drive fishing behavior (Sanchirico and Wilen 2001, Smith and Wilen 2003, Sanchirico et al. 2006, Kellner et al. 2007, Smith et al. 2008, Kaplan et al. 2009). Therefore, it is necessary to monitor and evaluate the effects of a reserve on human use with the same frequency and rigor that are applied to fish populations.

Marine reserves are human institutions and have no direct effects on ecological systems. Instead, reserves regulate people by establishing areas where certain activities are restricted (Lynch 2006). Thus, the success of a marine reserve depends on managers' ability to alter human behavior in ways that support reserve objectives (Hilborn 2007). Human behavioral decisions are multidimensional, and people may respond to policy by adjusting over these multiple dimensions (e.g., gear restrictions may lead people to change where they fish; Wilen et al. 2002). It is often difficult to predict human responses to policy change, and behavioral substitution may result in failure for the reserve (Smith and Wilen 2003, Kellner et al. 2007, Smith et al. 2008, Kaplan et al. 2009). Empirical studies of human responses to terrestrial and marine reserves have demonstrated that human behavior is critical but often unpredictable (Liu et al. 2001, Christie 2004, Guidetti et al. 2008). Despite the importance of human behavior, there have been few empirical studies of spatial reallocation due to marine reserves (Willis et al. 2003, Sale et al. 2005, Lynch 2006, Parnell et al. 2010). Human use monitoring (specifically, the monitoring of fishing-site choice by stakeholders) is commonly only conducted for high-value commercial fisheries in developed countries with commercial vessel observer programs (e.g., NMFS 2007).

Manuscript received 1 April 2011; revised 17 August 2011; accepted 18 August; final version received 21 September 2011. Corresponding Editor (ad hoc): C. H. Ainsworth.

[4] E-mail: marie.fujitani@asu.edu

Studies of marine reserves have focused largely on commercial fisheries (reviewed in Branch et al. 2006) and for-hire recreational fisheries, which function more like commercial fisheries than other recreational fisheries (Smith et al. 2008). Less consideration has been given to non-chartered recreational fisheries in both conservation policy and ecological theory (McCluskey and Lewison 2008, Parnell et al. 2010). Yet, decentralized recreational fisheries are biologically and economically important (McPhee et al. 2002, Schroeder and Love 2002, Coleman et al. 2004), and require explicit research due to their unique incentive structures and behavioral responses (Murray-Jones and Steffe 2000, McCluskey and Lewison 2008). Models of aggregate recreational fishing behavior necessarily differ from models of commercial fisheries because recreational fishers are individuals with heterogeneous utility profiles and skill levels (Murray-Jones and Steffe 2000, Lynch 2006, Johnston et al. 2010). Recreational anglers enjoy an overall experience (Holland and Ditton 1992), and observed fishing-site selection patterns are the result of individual anglers making decisions about where (and whether) to fish. The intended proximate impact of a marine reserve is to alter the trade-off structure that enters the site choice decision process.

We hypothesize two ways that a marine reserve can affect an angler's decision to visit a reserve site. First, recreational anglers could visit a reserve site less frequently than they would have had the area remained open to fishing. Indeed, this is the intended policy outcome of a reserve. The net benefits to an angler to fish at a reserve site are diminished as restrictions remove an attractive fishing area, and anglers would distribute to other sites or leave the fishery (Smith et al. 2010). Second, it is possible that the spatial allocation of fishing effort would not change relative to what would have occurred without the reserve. This is different than failing to detect a change in the absolute amount of fishing, as other factors besides the presence of a reserve can affect trends in fishing over time (Hurlbert 1984, Stewart-Oaten et al. 1986). These hypotheses inform our interpretation of recreational anglers' responses to the creation of a marine reserve.

To examine the support for these hypotheses, we synthesized a data set from the logbooks of a community-based search and rescue service in the Gulf of California, Mexico. Our data include a complete census of daily fishing-site choices for the members of a community of recreational anglers over a nine-year period. In year six of the data series, a marine reserve was implemented at one popular fishing site. We analyzed the data with regression to check for a structural break that would indicate whether visitation to that site changed after the marine reserve was implemented. We then modeled the propensity for a boat from the fleet to visit the reserve site on any given day with a fleet-level discrete choice model. These analyses were conducted to determine whether there is support against the null hypothesis that the reserve had no effect on human behavior despite a lack of evidence for changes in fish stocks (Fujitani 2010). We modeled the data two different ways (regression and discrete choice model) to verify that our qualitative results are robust to model specification.

Methods

Data and study area

The recreational angling community of Bahia de Kino consists of United States and Canadian citizens who are semipermanent residents in Mexico. The Bahia de Kino community is a major angling community in the central Gulf of California. Between 2000 and 2008, the recreational fishing community operated an average of 252 boats. Anglers target both pelagic species such as yellowtail (*Seriola lalandi*) and dorado (*Coryphaena hippurus*), and resident species such as grouper and sea bass (Serranidae), at a variety of sites in the central region of the Gulf of California, including the area surrounding San Pedro Mártir Island (SPMI; Fig. 1).

The recreational angling community of Bahia de Kino maintains a volunteer rescue radio service, known as Rescue One, on a dedicated VHF channel to track boaters in the event a water rescue becomes necessary. Boaters radio their planned destination(s) and their expected return time, and Rescue One volunteers track each boat until it returns. Anglers reported destinations in terms of general regions or landmarks, and have strong incentives for accurate reporting because there are no official water rescue services. Furthermore, variable weather conditions, mechanical uncertainty, and the lack of alterative rescue services lead almost every member of the Bahia de Kino recreational angling community to utilize Rescue One on every trip (J. Jerdee, *personal communication*). Recreational anglers logged 14 894 trips on Rescue One between 2000 and 2008, and 89% of fishing trips were to one of seven sites identified by major landmarks (sites A–F and SPMI; Fig. 1). The remaining sites were outside the immediate area of Bahia de Kino and frequently required multiple days of travel.

SPMI is remote and isolated (Fig. 1; see Plate 1), and recreational anglers are the most numerous fishers in the waters surrounding the island (Meza et al. 2008). SPMI was designated as a biosphere reserve in 2002 by executive order (Poder Ejecutivo Federal 2002) and consists of a 290-km^2 buffer zone and a marine no-take zone of $\sim$9 km^2 (Fig. 1). No extractive activities are permitted in the no-take zone, but small-scale and recreational fishing are allowed in the buffer zone (Poder Ejecutivo Federal 2002). Though the reserve was created in 2002, institutional support and reserve implementation followed in 2005. On 10 April 2005, an advisory 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 would begin in a few days, and that federal

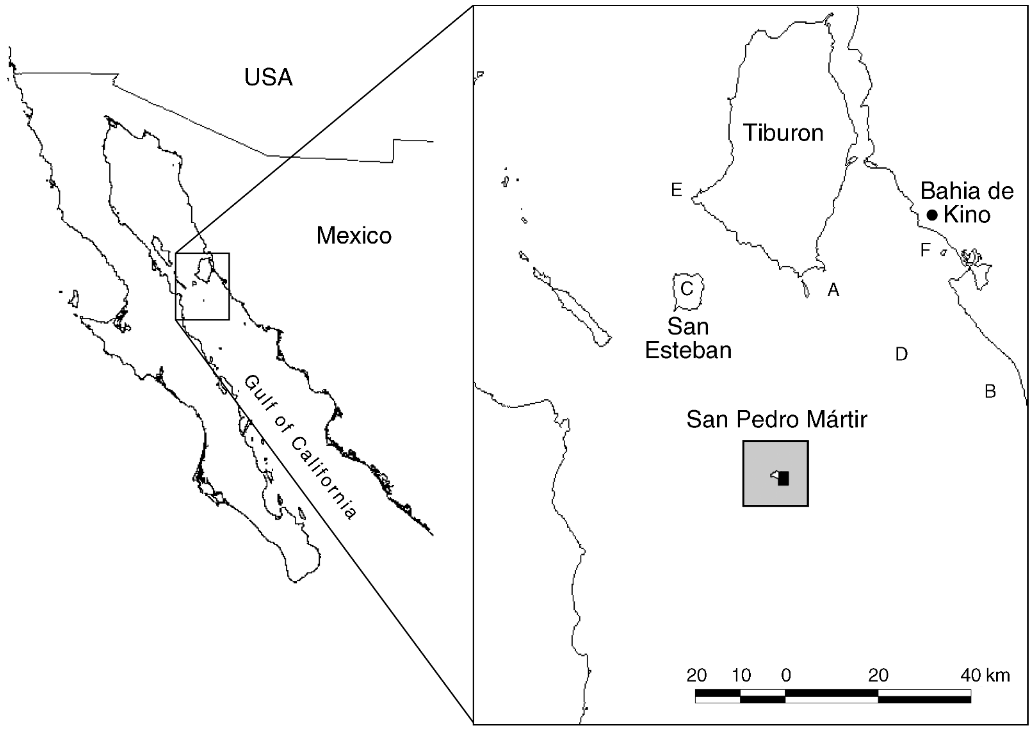

FIG. 1. Map of the central Gulf of California, Mexico, and the location of Bahia de Kino with respect to San Pedro Mártir Island (SPMI) and alternate fishing sites (A–F). The light gray square surrounding San Pedro Mártir Island indicates the 290-km^2 buffer zone of the reserve, with the marine no-take zone (9 km^2) represented by the smaller black rectangle.

officers would be on hand to enforce penalties of fines, boat impoundment, and imprisonment. This marks the definitive date when the SPMI area was reserved from the Bahia de Kino recreational angling community.

Strategies for detecting reserve effects without an environmental control

We pursued two strategies to detect an effect of the reserve on human use given the lack of a suitable control site in the data, and to check model agreement for robustness. In order to determine if there is evidence of a behavioral change, we ideally would replicate the coupled social–ecological system of the reserve area and compare behavior in the system with and without the reserve. However, any kind of site replication is seldom possible in the quantitative evaluation of marine reserves (Ludwig et al. 1993).

A next-best option might appear to be to compare mean human use before and after the reserve was implemented. Quantitatively comparing the means of a site characteristic (e.g., human use) before and after an impact is not particularly meaningful because before–after analyses do not account for other factors driving changes in human use patterns (e.g., weather, the economy) that may occur in concert with reserve establishment (Hurlbert 1984, Stewart-Oaten et al. 1986, Underwood 1994, Willis et al. 2003). An alternative to before–after analysis is to detect a structural change in behavior, termed a "break," before and after the reserve is implemented (Chow 1960). We do this by fitting an unrestricted model over the entire data set, and then fitting separate models of behavior before and after the reserve is created. If more variation in the data is explained by the two separate models, this provides evidence that a qualitative change in behavior occurred between those periods that may be used to begin to build a phenomenological case for the influence of the reserve.

The sociological and economics literature regularly investigates complex non-replicable systems. One method developed to address the problem of replication is to statistically construct a control (Rubin 1974, Winship and Morgan 1999). This control is termed a "counterfactual scenario" because it represents the ex post (statistically constructed from past data) world without the treatment effect. We constructed a counterfactual scenario by projecting over the factors besides the treatment that capture latent trends. The counterfactual scenario is valuable because independent of a treatment, human behavior may change over time. Therefore, a projected counterfactual can be used to reveal the presence of a reserve effect, even if no biological change is ever detected. For example, if we observe no change in fish stocks and doubling fishing pressure over time at a site that has been declared a reserve, we might conclude that the reserve is having no effect. However, if the projected counterfactual scenario showed fishing pressure tripling each year, this would reveal that in actuality the reserve reduced fishing pressure relative to the no-reserve case. The counterfactual scenario allows identification of these hidden changes resulting from the reserve.

Models

First, we used linear regressions to characterize the overall behavior of the recreational angling population over nine years. Preliminary analysis revealed that the total number of annual trips ($R^2 = 0.15$, $F_{1,7} = 1.26$, $P = 0.30$) and number of anglers taking trips annually ($R^2 = 0.18$, $F_{1,7} = 1.59$, $P = 0.25$) remained unchanged. Therefore, we analyzed changes in the fleet's proportion of visits to SPMI over time to investigate how the fishing community allocated trips. We regressed proportions of total embarkations per month (arcsine square-root transformed to satisfy the assumption of normality) to SPMI (y_i) over time (t) with month as a fixed effect to account for seasonal fluctuations in fishing behavior and catch rates, and tested for a structural break (Chow 1960) using the unrestricted model

$$y_i = \text{factor(month)} + tb + \varepsilon$$

and a restricted model that divided the data before the month, April 2005, when enforcement of the reserve was announced to the recreational anglers. The arcsine square-root transformed trips proportion increase per unit time is represented by b. If b is zero, then the allocation of trips to SPMI is not changing over time. The error term is represented by ε. The model does not provide a mechanism for behavioral change, and it is therefore possible that either the intercept or slope term could absorb fleet-level changes. Therefore, we also performed a fixed-intercept test for a difference in the slope of a regression of the arcsine square-root transformed proportion of visits per month over time before and after April 2005

$$y_i = \text{factor(month)} + (1 - D)tb_1 + Dtb_2 + \varepsilon$$

where D is a dummy variable that equals 0 for months before April 2005 and 1 otherwise. Month is again included in the model as a fixed effect. The slope coefficients "before" (b_1) and "after" (b_2) have the same interpretation as b and were compared with a paired t test.

Second, to test for robustness to model specification, we modeled the propensity of at least one boat to choose SPMI as a fishing destination on a given day that any boat traveled to any destination. Members of the recreational angling community often share boats and fish in groups, particularly to distant locations such as SPMI (M. Fujitani, *unpublished data*). Thus, modeling the propensity of one boat to travel to SPMI on a fishable day is a good indicator of the behavior of the Bahia de Kino community on that day. To conduct this analysis we constructed a counterfactual scenario by fitting a binomial logit model using the statistical package R (v. 2.10; R Development Core Team 2010), as follows:

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

The vector $\mathbf{z}$ holds the binary responses that at least one boat leaving port took a trip to SPMI on a day any boat took a trip from Kino (i.e., a day with "fishable" conditions), conditional on $\mathbf{x}$, a matrix of predictors influencing the decision to travel to SPMI.

The vertical vector $\boldsymbol{\beta}$ contains estimated parameters that account for variation in the choice to visit SPMI; Λ is the logistic cumulative distribution function. The variables in $\mathbf{x}$ used to estimate the reserve effect were the days since the reserve announcement and days since the announcement squared (to capture any nonlinear behavior). The coefficient associated with the squared term can be interpreted as half the second derivative of the response with respect to time (ignoring higher order effects). The positive (negative) sign indicates that the behavioral response was convex (concave) in time. These curvature properties help describe behavioral adjustments following reserve implementation. Variables accounting for other sources of variation in trip behavior were year and daily maximum wind speed (measured in miles per hour; *available online*)[5] as slope variables. Wind speed directly affects the safety of open water travel. Year proxies temporal trends that may include gas prices and regional-level fish population changes or changes in catch rates. Month of the year and whether a particular day was a weekend were included as fixed effects. Month of the year captures seasonal non-fishing opportunities as well as seasonal variation in catch rates and the composition and abundance of fished species. Also, within a given week anglers are expected to recreate more on weekends. Stock abundances were not explicitly included in the model. Limited data for stock at SPMI were available, but no clear biological trends have been observed and no effects of the reserve have been detected (Fujitani 2010). In addition, no data exist for the set of other sites available to the fleet. Fisheries data available are for commercial fisheries, but are only at port-level resolution (Cinti et al. 2009). We accounted for seasonal and interannual changes in stock abundances and fishing conditions with year and month fixed effects.

Results

Regression analyses indicated that the number of boats in the recreational angling fleet declined between 2000 and 2008 ($R^2 = 0.58$, $F_{1,7} = 9.97$, $P = 0.02$), but the total number of annual trips ($R^2 = 0.15$, $F_{1,7} = 1.26$, $P = 0.30$) and number of anglers taking trips annually ($R^2 = 0.18$, $F_{1,7} = 1.59$, $P = 0.25$) remained unchanged. Because the population of fishers and the number of trips stayed constant over the nine-year period, any variation would

[5] http://www.wunderground.com/history/airport/MMGM/2000/1/1/CustomHistory.html

TABLE 1. R^2 values, regression coefficients, and significance levels for regressions of the proportional number of visits by recreational anglers to San Pedro Mártir Island (SPMI) and six other important fishing sites in the Gulf of California, Mexico.

Location	R^2	Intercept	Slope	F	P	Proportion of total visits
San Pedro Mártir	0.58	−26.81	0.01	10.07	0.016	0.11
Site A	0.64	43.65	−0.02	12.61	0.009	0.43
Site B	0.33	13.91	−0.01	3.53	0.109	0.10
Site C	0.75	−16.57	0.01	21.31	0.002	0.05
Site D	0.01	−0.82	0.00	0.09	0.77	0.04
Site E	0.15	−10.78	0.00	1.22	0.30	0.08
Site F	0.63	-19.65×10^3	9.87	12.52	0.009	0.07
Other	0.68	10.58	−0.01	16.95	0.006	0.11

Notes: Sites are shown in Fig. 1. The category of "other" sites is composed of non-fishing sites and fishing sites comprising <2% of total sites visited in each year. The proportional visits to each site are based on the total number of sites visited from 2000 to 2008.

be in the proportional allocation of trips to sites and not total trips. The proportion of trips to SPMI increased significantly over time ($R^2 = 0.76$, $F_{1,7} = 9.31$, $P = 0.02$; Table 1, Fig. 2). Regression results for the proportion of visits to popular alternate fishing sites over time indicate that visitation trends to alternate sites also changed (Table 1).

The proportion of trips to SPMI per month, controlled for month effects, showed trips to SPMI increasing over time (adjusted $R^2 = 0.41$, $F_{12,95} = 7.30$, $P < 0.001$). Adjusted R^2 can be interpreted as a goodness of fit that imposes a penalty for additional parameters (Zar 1984). The Chow test indicated a significant difference in the trend of visits to SPMI before and after the reserve was announced ($F_{13,82} = 2.25$, $P = 0.014$). The fixed-intercept model showed that the slope coefficients were significantly different before and after the reserve was announced if the baseline is held constant ($t_{106} = 3.53$, $P < 0.001$). Fig. 2 illustrates a simplified version of this model, as month fixed effects cannot be represented in a two-dimensional graph.

We performed a post hoc survey with each month serving as a break point for a Chow test with year and month as fixed effects (Fig. 3). We found that 2005 showed the strongest signal of a structural change. The two peaks in 2005 and 2006 are during the summer months, when the most trips are taken to SPMI; thus, we may expect that the strongest behavioral change would be observed during those months.

The parameters estimated from the binomial logit model provide information on the magnitude and direction of changes in human use due to the implementation of the reserve. The model shows the influence of various predictors on the propensity of at least one member of the recreational angling community to travel to SPMI on a particular day (Table 2). More trips to SPMI are expected in the spring and early summer months, over time, and on weekends, and fewer trips are expected with increased wind speed. The coefficients associated with these variables were all significantly different from zero and had the expected signs (Table 2). Without site-specific data we cannot mechanistically account for structural changes in catch rates; however, the fixed effects do control for changing catch rates. The coefficient characterizing changes in behavior concurrent with the reserve announcement, days since announcement, was significant and negative, indicating a decline in the propensity of anglers to visit the area after the announcement. The second-order term, days since the announcement squared, was significant and positive indicating that the negative first-order effect of the reserve on behavior diminished with time.

Fig. 4 is a simplified two-dimensional illustration of the fitted logit model and a projected counterfactual with bootstrapped 95% projection intervals. Because we

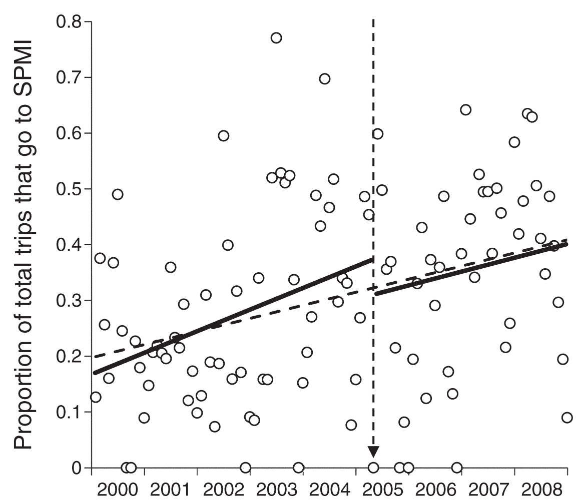

FIG. 2. A scatterplot of the arcsine square-root transformed proportion of trips by recreational anglers to SPMI out of the total number of trips taken each month from 2000 to 2008. The dashed arrow indicates the announcement of the reserve, and the black dashed line is the least-squares regression over the entire time series. The solid lines are fixed-intercept regressions for the patterns of visitation before and after the reserve was announced. This is a simplified visual representation of the fitted model, which included month of the year as a fixed effect. The estimated slope coefficients from that model are significantly different at $P < 0.001$, indicating there was a structural break in behavior concurrent with the announcement of the reserve.

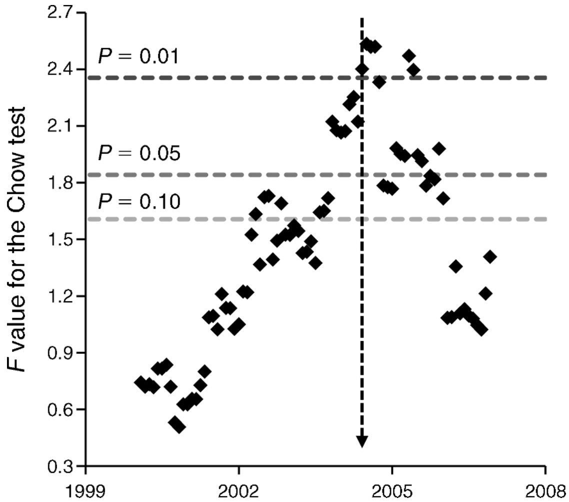

FIG. 3. Plot of F values from Chow tests performed with month and year fixed effects for a break at each month in the time series. The arrow indicates the announcement of the reserve. Horizontal lines are thresholds for P values, with 13 and 82 degrees of freedom.

have a complete census of the population, these intervals do not reflect uncertainty in the sampling method, but instead a fleet by site interaction that captures unexplained fleet-level decisions. We used the coefficients estimated from our model and average values (e.g., for wind) to project the statistically expected propensity of at least one angler to visit SPMI on a day any angler chose to take a trip for a weekday in June (Fig. 4). June was chosen because it is the month with the highest proportion of angler use of SPMI. The greatest projected difference between the fitted logit model and the projected counterfactual scenario was observed in 2006, the year after the reserve was implemented. These results indicate that, in June 2006, the reserve reduced the propensity of anglers to travel to SPMI by an average of 10.05%, and by potentially as much as 19.41%.

DISCUSSION

We detected a change in site visitation patterns following the announcement of enforcement in the SPMI marine reserve. This result is robust to model specification, as tests for a structural change and the logit model provide evidence that the announcement of the reserve affected fleet-level behavior. This finding is particularly noteworthy given the failure to detect an effect of the reserve through biological monitoring (Fujitani 2010). Human use monitoring provided evidence of a reserve effect on human behavior in the absence of an ecological effect.

The counterfactual scenario provided key information on latent temporal trends in behavior, as well as changes attributable to the marine reserve. The projected counterfactual showed a latent trend of increasing travel to SPMI (circles, Fig. 4). Data on fishing-site visitation showed that the proportion of fishing visits to remote islands including SPMI and site C (Table 1) were increasing over time. Interviews with members of the recreational angling community suggested this trend reflected both the perception of a decline in the quality of fishing close to Bahia de Kino and improvements in boat technology that allow anglers to safely pursue more distant fishing grounds (M. Fujitani, *unpublished data*). The announcement of the reserve reduced visits to SPMI (days since announcement, Table 2), and visitation to the reserve area decreased relative to the counterfactual scenario (squares, Fig. 4). This suggests that without the reserve the level of visitation to SPMI likely would have been greater than the realized visitation. By dampening

TABLE 2. Parameter estimates, standard errors, and significance levels from a binomial logit model of the choice of a trip to San Pedro Mártir Island.

Parameter	Estimate	SE	Z	P
Days since announcement	-1.61×10^{-3}	5.51×10^{-4}	−2.93	0.003
[Days since announcement]^2	1.34×10^{-6}	4.08×10^{-7}	3.27	0.001
Intercept	−364.70	75.36	−4.84	$<$0.001
February	1.07	0.24	4.50	$<$0.001
March	1.47	0.23	6.38	$<$0.001
April	0.65	0.24	2.67	0.001
May	1.72	0.23	7.48	$<$0.001
June	2.23	0.24	9.41	$<$0.001
July	0.97	0.25	3.88	$<$0.001
August	0.15	0.29	0.53	0.59
September	0.09	0.30	0.06	0.95
October	0.22	0.26	0.85	0.40
November	−0.80	0.30	−2.69	0.007
December	−0.56	0.30	−1.86	0.06
Year	0.18	0.04	4.82	$<$0.001
Weekend	0.29	0.10	2.90	0.004
Wind	−0.06	0.02	−3.13	0.002

Note: Changes in travel behavior due to the announcement of the reserve are seen in the parameters "Days since announcement" (defined as the number of days since the reserve was announced) and "[Days since announcement]^2."

PLATE 1. San Pedro Mártir Island in the Gulf of California. Photo credit: M. L. Fujitani.

an increasing trend, the marine reserve may have protected stocks from further deterioration. Nevertheless, the marine reserve did not afford long-term protection necessary for stock recovery.

Our data set was uniquely rich, and allowed us not only to detect the initial effect of the SPMI reserve, but also changes in human use over time following reserve implementation. The positive and significant squared term from the logit model ([days since announcement]^2, Table 2) indicates that the strength of the reserve to reduce trips to SPMI diminished with time relative to the counterfactual scenario. Anglers responded rapidly to the reserve announcement by reducing their propensity to visit SPMI (days since announcement, Table 2). This is evidence that a reserve could work in this system, as threats of fines and boat impoundment appears to have influenced human behavior. However, the relative decrease in trips did not last, and we observed the reserve and projected no-reserve scenarios to converge (Fig. 4). This suggests that the institutions that govern the reserve provided anglers with inadequate disincentives for visitation, as anglers may have learned that the rules of the reserve were not being enforced. In the case of SPMI, anglers have been observed fishing in the no-take zone (Meza et al. 2008), although no penalties have been enforced to date (M. Fujitani, *personal observation*). This is a common outcome for marine reserves in the Gulf of California (Cudney-Bueno et al. 2009, Stamieszkin et al. 2009) and reserves worldwide (Liu et al. 2001, Guidetti et al. 2008). Our results provide a detailed empirical example of human responses to "paper reserves," and support the notion that environmental "failures" are often actually institutional failures (Dietz et al. 2003).

The SPMI reserve decreased human use of the reserve area, though the effect was insufficient to offset the

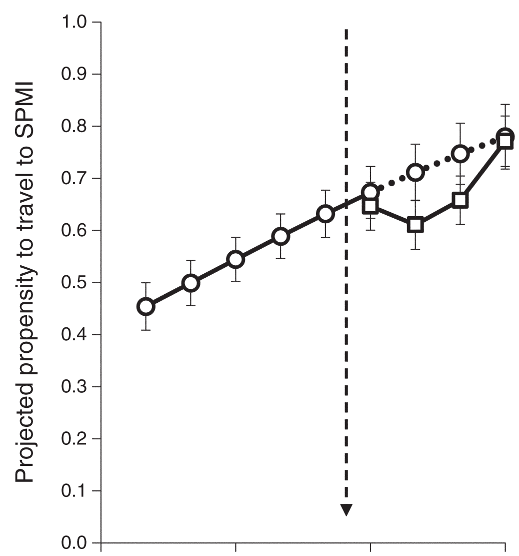

FIG. 4. Plots of the propensity of at least one boat to travel to SPMI on a given weekday in June that any boat took a trip, with projected parameters from the logit model, average values, and bootstrapped 95% projection intervals. Circles indicate the propensity to visit SPMI without a marine reserve; post-2004 values (dotted line) are projections of the counterfactual scenario with reserve effects set to zero. Squares are the propensity to visit SPMI with the reserve effects estimated by the logit model.

latent trend of increasing exploitation (Fig. 4). Therefore, we would not expect to observe increasing biological stocks around SPMI, and data suggest that stocks have not increased (Fujitani 2010). This result begs the question: Is slowing exploitation a sufficient justification for the reserve? Indeed, a decreased rate of decline may be considered success in some cases. However, our analysis indicates that visitation effects can be complicated, because reductions in use that follow the establishment of a reserve may not be sustained without institutional support. In the SPMI case, the effect of the reserve on this stakeholder group did not last, and we may expect angler use of SPMI to return to levels predicted by the counterfactual scenario. In effect, the reserve acted as a one-time shock, and such shocks are likely to be insufficient for most systems in need of conservation. This reserve is likely to fail to meet its ecological objectives if the reserve's governing institutions are not modified. Reserves are top-down implements, and immediately bring to mind top-down instruments such as policing. Enforcing reserve laws through policing is effective (Guidetti et al. 2008), but in some cases may be more costly, incite more conflict, and be more difficult to maintain long term than other institutional mechanisms (Brechin et al. 2002, Christie et al. 2003, Christie 2004). Self-enforcing contracts (Ostrom 1990), shifting social and cultural norms (McKenzie-Mohr 2000), and the incorporation of human dimensions from the reserve planning stage (Christie et al. 2003, Mascia et al. 2003) are other institutional arrangements that may help perpetuate successful reserves.

As this case illustrates, human use monitoring can provide a leading indication that the institutions governing a reserve are or are not effective, and potentially can do so faster than biological effects can be detected. Furthermore, information on how management has impacted human use provides actionable intelligence on the only portion of the coupled human–ecological reserve system that management directly affects: incentives affecting human decisions. In this case we learn that institutional changes are required, along with stronger disincentives for visitation if the goal of the reserve is to reduce recreational fishing trips to the reserve area. Marine reserves regulate people. Therefore, understanding how people respond to a reserve should be a central part of reserve creation and is essential to provide feedback for reserve rules and management, which directly affects people, but only indirectly affects fish.

Acknowledgments

We thank Dave White, Jamie Jerdee, Ana Luisa Figueroa, Kirsten Rowell, Jennifer Duberstein, the Organization for the Isla San Pedro Mártir Biosphere Reserve, and the Prescott College Kino Bay station. We thank two anonymous reviewers for helpful comments and suggestions. M. L. Fujitani was supported by Comunidad y Biodiversidad, A.C. L. R. Gerber was supported as a Sabbatical Fellow at the National Center for Ecological Analysis and Synthesis, a Center funded by NSF (Grant number EF-0553768), the University of California–Santa Barbara, and the State of California. M. L. Fujitani and E. P. Fenichel were partially supported by NOAA-Saltonstall Kennedy grant NA09NMF4270098.

Literature Cited

Branch, T. A., R. Hilborn, A. C. Haynie, G. Fay, L. Flynn, J. Griffiths, K. N. Marshall, J. K. Randall, J. M. Scheuerell, E. J. Ward, and M. Young. 2006. Fleet dynamics and fishermen behavior: lessons for fisheries managers. Canadian Journal of Fisheries and Aquatic Sciences 63:1647–1668.

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. 2004. Marine protected areas as biological successes and social failures in Southeast Asia. American Fisheries Society Symposium 42:155–164.

Christie, P., et al. 2003. Toward developing a complete understanding: a social science research agenda for marine protected areas. Fisheries 28:22–26.

Cinti, A., W. Shaw, R. Cudney-Bueno, and M. Rojo. 2009. The unintended consequences of formal fisheries policies: Social disparities and resource overuse in a major fishing community in the Gulf of California, Mexico. Marine Policy 34:328–339.

Coleman, F. C., W. F. Figueira, J. S. Ueland, and L. B. Crowder. 2004. The impact of United States recreational fisheries on marine fish populations. Science 305:1958–1960.

Cudney-Bueno, R., L. Bourillo, A. Saenz-Arroyo, J. Torre-Cosio, P. Turk-Boyer, and W. W. Shaw. 2009. Governance and effects of marine reserves in the Gulf of California, Mexico. Ocean and Coastal Management 52:207–218.

D'Agrosa, C., L. R. Gerber, E. Sala, J. Wielgus, and F. Ballantyne IV. 2007. Navigating uncertain seas: Adaptive monitoring and management of marine protected areas. Arizona State University, Tempe, Arizona, USA, and Scripps Institution of Oceanography, University of Californiua, San Diego, California, USA. http://gerberlab.faculty.asu.edu/MPAs_LOWRES.pdf

Dietz, T., E. Ostrom, and P. Stern. 2003. The struggle to govern the commons. Science 302:1907–1912.

Fujitani, M. L. 2010. The rapid assessment of a new marine reserve in the Gulf of California, Mexico. Thesis. Arizona State University, Tempe, Arizona, USA. http://gerberlab.faculty.asu.edu/docs/Fujitani_2010_MScThesis.pdf

Gerber, L. R., L. W. Botsford, A. Hastings, H. P. Possingham, S. D. Gaines, S. R. Palumbi, and S. Andelman. 2003. Population models for marine reserve design: a retrospective and prospective synthesis. Ecological Applications 13:S47–S64.

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.

Guidetti, P., et al., editors. 2008. Italian marine reserve effectiveness: Does enforcement matter? Biological Conservation 141:699–709.

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

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

Holland, S. M., and R. B. Ditton. 1992. Fishing trip satisfaction: a typology of anglers. North American Journal of Fisheries Management 12:28–33.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54:187–211.

Johnston, F. D., R. Arlinghaus, and U. Dieckmann. 2010. Diversity and complexity of angler behaviour drive socially optimal input and output regulations in a bioeconomic recreational-fisheries model. Canadian Journal of Fisheries and Aquatic Sciences 67:1507–1531.

Kaplan, D. M., L. W. Botsford, M. R. O'Farrell, S. D. Gaines, and S. Jorgensen. 2009. Model-based assessment of persistence in proposed marine protected area designs. Ecological Applications 19:433–448.

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.

Liu, J., M. Linderman, Z. Ouyang, L. An, J. Yang, and H. Zhang. 2001. Ecological degradation in protected areas: the case of Wolong nature reserve for giant pandas. Science 292:98–101.

Ludwig, D., R. Hilborn, and C. Walters. 1993. Uncertainty, resource exploitation, and conservation: lessons from history. Ecological Applications 3:548–549.

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

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.

McCluskey, S. M., and R. L. Lewison. 2008. Quantifying fishing effort: a synthesis of current methods and their applications. Fish and Fisheries 9:188–200.

McKenzie-Mohr, D. 2000. Promoting sustainable behavior: An introduction to community-based social marketing. Journal of Social Issues 56:543–554.

McPhee, D. P., D. Leadbitter, and G. A. Skilleter. 2002. Swallowing the bait: is recreational fishing in Australia ecologically sustainable? Pacific Conservation Biology 8:40–51.

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, Guaymas, Mexico. www.cobi.org.mx

Murray-Jones, S., and A. S. Steffe. 2000. A comparison between the commercial and recreational fisheries of the surf clam, *Donax deltoides*. Fisheries Research 44:219–233.

NMFS [National Marine Fisheries Service]. 2007. NMFS Strategic Plan for Fisheries Research. U.S. Department of Commerce, NOAA Technical Memo F/SPO-79. NMFS, Silver Spring, Maryland, USA.

Ostrom, E. 1990. Governing the commons: The evolution of institutions for collective action. Cambridge University Press, Cambridge, UK.

Parnell, P. E., P. K. Dayton, R. A. Fisher, C. C. Loarie, and R. D. Darrow. 2010. Spatial patterns of fishing effort off San Diego: implications for zonal management and ecosystem function. Ecological Applications 20:2203–2222.

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.

R Development Core Team. 2010. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org

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

Rubin, D. B. 1974. Estimating the causal effects of treatment in randomized and nonrandomized studies. Journal of Educational Psychology 66:688–701.

Sale, P. F., R. K. Cowen, B. S. Danilowicz, G. P. Jones, J. P. Kritzer, K. C. Lindeman, S. Planes, N. V. C. Polunin, G. R. Russ, Y. J. Sadovy, and R. S. Steneck. 2005. Critical science gaps impede use of no-take fishery reserves. Trends in Ecology and Evolution 20:74–80.

Sanchirico, J. N., U. Malvadkar, A. Hastings, and J. E. Wilen. 2006. When are no-take zones an economically optimal fishery management strategy? Ecological Applications 16:1643–1659.

Sanchirico, J. N., and J. E. Wilen. 2001. A bioeconomic model of marine reserve creation. Journal of Environmental Economics and Management 42:257–276.

Schroeder, D. M., and M. S. Love. 2002. Recreational fishing and marine fish populations in California. California Cooperative Oceanic Fisheries Reports 43:182–190.

Smith, M. D., J. Lynham, J. N. Sanchirico, and J. A. Wilson. 2010. Political economy of marine reserves: Understanding the role of opportunity costs. Proceedings of the National Academy of Sciences USA 107:18300–18305.

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. Zhanga, 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. Weilgus, 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 and Coastal Management 52:449–458.

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

Underwood, A. J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. Ecological Applications 4:3–15.

Wells, S., V. Sheppard, H. van Lavieren, N. Barnard, F. Kershaw, C. Corrigan, K. Teleki, and P. Stock. 2008. National and regional networks of marine protected areas: a review of progress. Master Evaluation for the UN Effort. World Conservation Monitoring Centre, Cambridge, UK.

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.

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

Winship, C., and S. L. Morgan. 1999. The estimation of causal effects from observational data. Annual Review of Sociology 25:659–707.

Wood, L. J., L. Fish, J. Laughren, and D. Pauly. 2008. Assessing progress towards global marine protection targets: shortfalls in information and action. Oryx 42:340–351.

Zar, J. H. 1984. Biostatistical analysis. Second edition. Prentice-Hall, Englewood Cliffs, New Jersey, USA.