Effectiveness of community-based marine reserves in small-scale fisheries

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ABSTRACT

Coastal marine ecosystems provide livelihoods for small-scale fishers and coastal communities around the world. Artisanal fisheries face great challenges since they are difficult to monitor, enforce, and manage. Combining territorial user rights for fisheries (TURF) with no-take marine reserves to create TURF-reserves is believed to improve the performance of small-scale fisheries by buffering fisheries from environmental variability and management errors, while ensuring that fishers reap the benefits of conservation investments. In the last six years, and following a 2012 regulation, 18 TURF-reserves have been implemented in Mexico; their effectiveness has not been formally evaluated. We combine causal inference techniques and a social-ecological systems framework to provide a holistic evaluation of community-based TURF reserves in three coastal communities in Mexico. We find that while reserves have not yet achieved their stated goal of increasing lobster densities, they continue to receive significant support from the fishing communities. A lack of ecological and socioeconomic effects likely results from a combination of factors. First, the lobster fisheries are already well managed, and it is unlikely that reserves might have a detectable effect. Second, some of the reserves are not large enough to protect lobsters' home ranges. Third, some of these reserves might be too young for the effects to show. However, these reserves have shaped small-scale fishers' way of thinking about marine conservation, which can provide a foundation for establishing additional, larger marine reserves needed to effectively conserve mobile species.

Keywords: TURF-reserves, Causal Inference, Social-Ecological Systems, Marine Protected Areas, Marine Conservation, Small-Scale Fisheries

1 INTRODUCTION

Marine ecosystems around the world sustain significant impacts due to overfishing and unsustainable fishing practices (Halpern et al., 2008; Worm et al., 2006; Pauly et al., 2005). In particular, artisanal fisheries face great challenges since they tend to be hard to monitor and enforce (Costello et al., 2012). Recent research shows that combining Territorial Use Rights for Fisheries (TURFs) with no-take marine reserves (MR) can greatly improve the performance of coastal fisheries and the health of the local resouces (Costello and Kaffine, 2010; Lester et al., 2017). Commonly known as TURF-Reserves, these systems increase the benefits of spatial access rights allowing the maintainance of healthy resources (Afflerbach et al., 2014; Lester et al., 2017). Although in theory these systems are successful (Costello and Kaffine, 2010), there is little empirical evidence of their effectiveness and the drivers of their success (Afflerbach et al., 2014; Lester et al., 2017; Smallhorn-West et al., 2018).

The performance of these systems depends on how environmental and social factors combine and interact. The science of marine reserves has largely focused on understanding the ecological effects of these areas, which include increased biomass, species richness, and densities of organisms within the protected regions, climate change mitigation, and protection from environmental variability (Lester et al., 2009; Giakoumi et al., 2017; Sala and Giakoumi, 2017; Roberts et al., 2017; Micheli et al., 2012). Modelling studies show that fishery benefits of marine reserves depend on initial stock status and the management under which the fishery operates, as well as reserve size and the amount of larvae exported from these (Hilborn et al., 2006; Krueck et al., 2017; De Leo and Micheli, 2015). Other research has focused on the relationship between socioeconomic and governance structures and reserve effectiveness (Halpern et al., 2013; López-Angarita et al., 2014; Mascia et al., 2017). However, to our knowledge, no studies exist that evaluate TURF-reserves from both a social and ecological perspective. This is especially important in social-ecological coastal systems dominated by close interaction and feedbacks between people and natural resources (Ostrom, 2009).

TURF-reserves can be created as community-based marine reserves, voluntarily established and enforced by local communities. This bottom-up approach increases compliance and self-enforcement (Gelcich and Donlan, 2015; Espinosa-Romero et al., 2014; Beger et al., 2004). Community-based spatial closures occur in different contexts, like the *kapu* or *ra'ui* areas in the Pacific Islands (Bohnsack et al., 2004; Johannes, 2002). However, without legal recognition no-take regulations are difficult to enforce and fishers rely on the exclusive access granted by the TURF. In an effort to bridge this normative gap, Civil Society Organizations (CSOs) have served as a link between fishers and government, and set out to create a legal framework that solve this governance issue. In Mexico, a new norm was created in 2014 allowing fishers to request the legal recognition of community-based reserves as "Fish Refuge" (*Zona de Refugio Pesquero*; NOM-049-SAG/PESC (2014)). Fish refuges can be implemented as temporal or partial reserves, which can protect one, some, or all resources within their boundaries. Since 2012, 45 of Fish Refuges have been created along the Pacific, Gulf of California, and Mexican Caribbean coastlines, with 18 of them implemented as TURF-reserves. However, their effectiveness has not yet been formally evaluated and reported in the scientific literature.

Here, we combine causal inference techniques and a social-ecological systems framework to provide a holistic evaluation of community-based marine reserves in three coastal communities in Mexico. These three case studies span a range of ecological and social conditions representative of different regions of Mexico. The objective of this work is twofold. First, to provide a triple bottom line evaluation of the effectiveness of community-based marine reserves that can inform similar processes in other countries. Second, to evaluate the effectiveness of TURF-reserves established as Fish Refuges in Mexico to identify opportunities where improvement or adjustment might lead to increased effectiveness. We draw from lessons learned in these three case studies and provide management recommendations to maximize the effectiveness of community-based marine reserves in small-scale fisheries in Mexico and in other regions around the world that are using this tool to manage and rebuild their coastal fisheries.

2 METHODS

2.1 Study area

We evaluate three TURF-reserves in Mexico (Fig 1A). The first one was created by the *Buzos y Pescadores de la Baja California* fishing cooperative, located in Isla Natividad in the Baja Peninsula (Fig 1B). The main fishery in the island is the spiny lobster (*Panulirus interruptus*), but other resources like finfish, sea cucumber, read sea urchin, snail, and abalone are also an important source of income. In 2006, the community decided to implement two marine reserves within their fishing grounds to protect commercially important invertebrate species; mainly lobster and abalone. While these reserves obtained legal recognition only in 2018, they have been well enforced since their implementation.

The other two TURF-reserves are located in Maria Elena and Punta Herrero, in the Yucatan Peninsula (Fig 1C). In contrast with Isla Nativdad, which hosts a well established fishing community, Maria Elena is a fishing camp –visited intermittently during the fishing season– belonging to the Cozumel fishing cooperative (*SCPP Cozumel*); Punta Herrero is home to the *SCPP José María Azcorra* cooperative, and similar to Isla Natividad hosts a local community. Their main fishery is the Caribbean spiny lobster (*Panulirus argus*), but they also target finfish in the off-season. Maria Elena and Punta Herrero established eight marine reserves in 2012, and four marine reserves in 2013, respectively. All these reserves have been legally recognized as Fishing Refuges since their creation (DOF, 2012, 2013).

These communities are representative of their region in terms of ecology, socioeconomic, and governance aspects. Isla Natividad, for example, is part of a greater group of fishing cooperatives belonging to a Federation of Fishing Cooperatives. This group has been identified as a cohesive group that often cooperates to better manage their resources (McCay, 2017; McCay et al., 2014; Aceves-Bueno et al., 2017). Likewise, Maria Elena and Punta Herrero are representative of fishing cooperatives in the Mexican Caribbean, which are also part of a regional Federation. Together, these three communities provide an accurate representation of other fishing communities in each of their regions. While each region has additional communities that have established community-based TURF-reserves, available data would not allow us to perform the in-depth analysis that we undertake. Yet, given the similarities among communities and the socioeconomic and governance setting under which they opperate, it is safe to cautiously generalize our results to other communities in Mexico and other regions around the world.

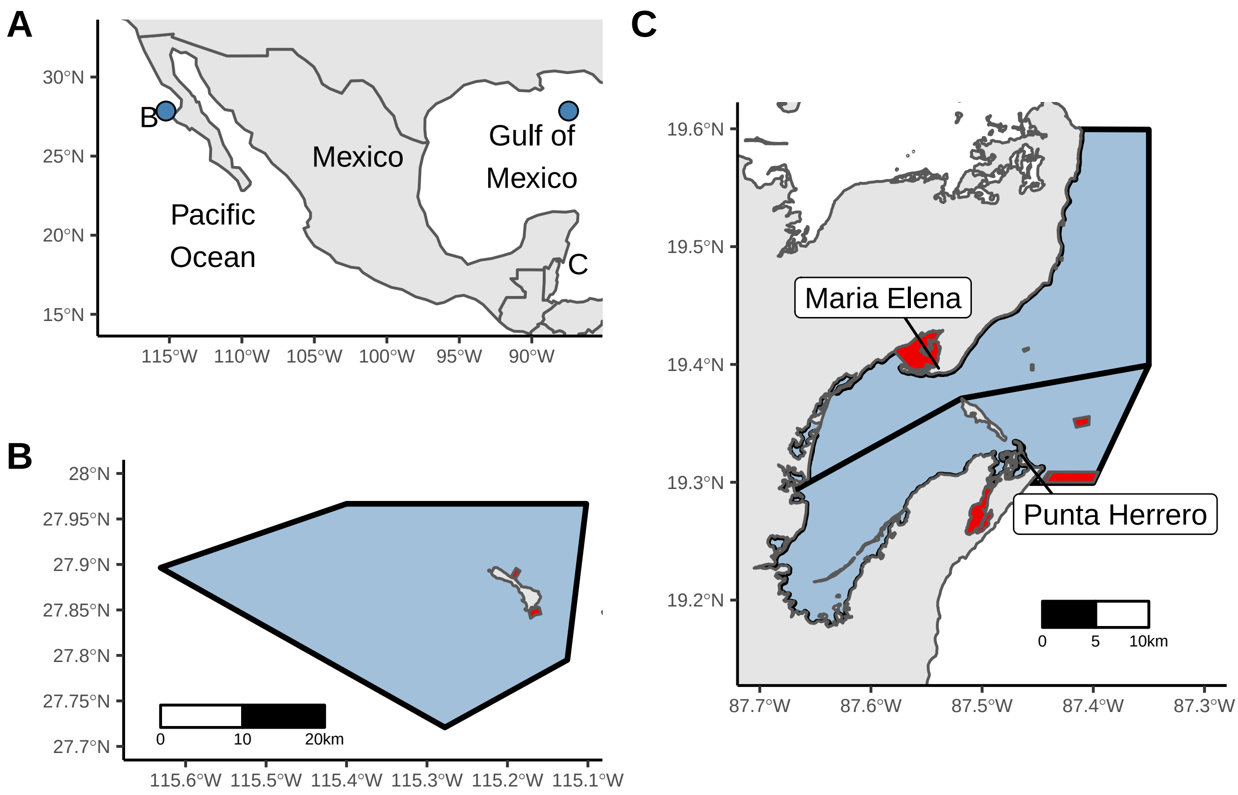

Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

2.2 Data collection

We use three main sources of information to evaluate these reserves across the ecological, socioeconomic, and governance dimensions. Ecological data come from the annual ecological monitoring of reserve and control areas, carried out by members from each community and personnel from the Mexican CSO *Comunidad y Biodiversidad* (COBI). Trained divers record richness and abundances of fish and invertebrate species along replicate transects (30x 2 m each) at depths 5-20 m in the reserves and control sites (Fulton et al., 2018, 2019). Size structures are also collected during fish surveys. We define control sites as regions with habitat characteristics similar to the corresponding reserves, and that presumably had a similar probability of being selected as reserves during the design phase. We focus our evaluation on sites where data are available for reserve and control sites, before and after the implementation of the reserve. This provides us with a Before-After-Control-Impact (*i.e.* BACI) sampling design that allows us to capture and control for temporal and spatial dynamics (De Palma et al., 2018; Ferraro and Pattanayak, 2006). BACI designs and causal inference techniques have proven effective to evaluate marine reserves, as they allow us to causally attribute observed changes to the intervention (Moland et al., 2013; Villaseñor-Derbez et al., 2018). All sites were surveyed annually, and at least once before implementation of the reserves. Table 1 shows a summary of the TURF-reserves included in this study.

Table 1. Summary of analyzed TURF-reserves by community.

Community	TURF area (km^{-2})	Reserve area (km^{-2})	Percent as reserves	Year of implementation
Isla Natividad	889.5	1.53	0.17%	2006
Maria Elena	353.1	0.10	0.03%	2012
Punta Herrero	299.7	0.43	0.14%	2013

Socioeconomic data come from landing receipts reported to the National Commission for Aquaculture and Fisheries (*Comisión Nacional de Acuacultura y Pesca*; CONAPESCA). Data contain monthly lobster landings (Kg) and revenues (MXP) for cooperatives with and without marine reserves. Cooperatives incorporated in this analysis belong to larger regional-level Cooperative Federations, and are exposed to the same markets and institutional frameworks, making them plausible controls (McCay, 2017; Ayer et al., 2018). Landings and revenues were aggregated at the cooperative-year level, and revenues were adjusted to represent 2014 values by the Consumer Price Index for Mexico (OECD, 2017) as:

$$I_t = RI_t \times \frac{CPI_t}{CPI_T} \tag{1}$$

Where I_t represents the adjusted income for year t as the product between the reported income for that year and the ratio between the consumer price index in that year (CPI_t) to the most recent year's consumer price index (CPI_T).

Data for the operationalization of the social-ecological system were collected at the community–level from official documents used in the creation and designation of the marine reserves (DOF, 2012, 2013, 2018) and based on the authors' experience and knowledge of the communities. These include information on the resource system, the resource units, actors, and the governance system itself (Table **??**).

2.3 Data analysis

We evaluate the effect that marine reserves have had on four ecological and two socioeconomic indicators (Table 2). Recall that reserves were implemented to protect lobster and other benthic invertebrates. However, we also use the available fish data to test for associated co-benefits.

Table 2. List of indicators used to evaluate the effectiveness of marine reserves, grouped by category.

Category	Indicator	Units
Biological	Lobster density	org m^{-2}
Biological	Invertebrate density	org m^{-2}
Biological	Fish biomass	Kg m^{-2}
Biological	Fish density	org m^{-2}
Socioeconomic	Income from target species	M MXP
Socioeconomic	Landings from target species	Metric Tonnes

We use a difference-in-differences analysis to evaluate these indicators. This approach allows us to estimate the effect that the reserve had by comparing trends across time and treatments (Moland et al., 2013; Villaseñor-Derbez et al., 2018). The analysis of ecological indicators is performed with a multiple linear regression of the form:

$$I_{itj} = \alpha + \gamma_t Year_t + \beta Zone_i + \lambda_t Year_t \times Zone_i + \sigma_j Spp_j + \epsilon \quad (2)$$

Where year-fixed effects are represented by $\gamma_{it} Year_t$, and $\beta Zone_i$ captures the difference between reserve ($Zone = 1$) and control ($Zone = 0$) sites. The interaction term $\lambda_{it} Year_t \times Zone_i$ represents the mean change in the indicator inside the reserve, for year t, with respect to the year of implementation in the control site (See Table 1). When evaluating biomass and densities of the invertebrate or fish communities, we include σ_j to control for species-fixed effects.

Socioeconomic indicators are evaluated with a similar approach. Due to data constrains, we only evaluate socioeconomic data for Isla Natividad (2000 - 2014) and Maria Elena (2006 - 2013). Neighboring communities are used as counterfactuals that allow us to control for unobserved time-invariants. Each focal community (Isla Natividad and Maria Elena) has three counterfactual communities.

$$I = \alpha + \gamma_t Year_t + \beta Treated_i + \lambda_t Year_t \times Treated_i + \sigma_j Com_j + \epsilon \quad (3)$$

The model interpretation remains as for Eq 2, but in this case the $Treated$ dummy variable indicates if the community has a reserve ($Treated = 1$) or not ($Treated = 0$) and $\sigma_j Com$ captures community-level fixed-effects. These regression models allow us to establish a causal link between the implementation of marine reserves and the observed trends by accounting for temporal and spatial dynamics (De Palma et al., 2018). The effect of the reserve is captured by the λ_t coefficient, and represents the difference observed between the control site before the implementation of the reserve and the treated sites at time t after controlling for other time and space variations (*i.e.* γ_t and β respectively). All model coefficients were estimated via ordinary least-squares and heteroskedastic-robust standard errors (Zeileis, 2004). All analyses were performed in R 3.5.0 and R Studio 1.1.453 (R Core Team, 2018). Data and code are available on github.com.

We use the social-ecologycal system (SES) framework to evaluate each community as a means of providing an explanation to the biological and socioeconomic results. The use the SES framework standardizes our analysis and allows us to communicate our results in a common language across fields. We based our variable selection primarily on Leslie et al. (2015); Basurto et al. (2013), who have previously analyzed Mexican fishing cooperatives using this framework. Table **??** shows the selected variables, the definition and selected indicators, and scoring system.

We followed Leslie et al. (2015) to operationalize the implementation of the SES framework. We analyzed 18 variables, that fitted within the first, second and third tires of the framework. Categorical variables were scored as binary data (0 or 1). Numerical variables were given a continuous score, and then standardized between 0 and 1. For example, the score on number of actors was relative to the smallest values (21 fishers) which was given a score of one since theoretically smaller groups facilitate coordination (Viana et al., 2018). Each component of the SES framework had a cumulative value of 1 and was equally weighted to calculate the final score.

3 RESULTS

The following sections present the effect that marine reserves had on each of the biological and socioeconomic indicators for each coastal community. Results are presented in terms of the difference through time and across sites, relative to the control site on the year of implementation (*i.e.* effect size λ_t). We also provide an overview of the governance settings of each community, and discuss how these might be related to the effectiveness and performance of the reserves.

3.1 Biological effects

Indicators showed ambiguous responses through time for each reserve. Figure 2A shows positive effect sizes for lobster densities in Isla Natividad and Punta Herrero during the first years, but the effect is eroded through time. In the case of Maria Elena, positive changes were observed in the third and forth year. These effects are in the order of 0.2 extra organisms m^{-2} for Isla Natividad and Punta Herrero, and 0.01 organisms m^{-2} for Maria Elena, but are not significantly different from zero ($p > 0.05$). Likewise, no changes were detected in fish biomass or invertebrate and fish densities (2B-D), where effect sizes oscillated around zero without clear trends. Full tables with model coefficients are presented in the supplementary materials (S1 Table, S2 Table, S3 Table).

3.2 Socioeconomic effects

Lobster landings and revenue were only available for Isla Natividad and Maria Elena (Fig 3). For all years before implementation, the effect sizes are close to zero, indicating that the control and treatment sites have similar pre-treatment trends, suggesting that these are plausible controls. However, effect sizes do not change after the implementation of the reserve. Interestingly, the negative effect observed for Isla Natividad on year 5 correspond to the 2011 hypoxia events. The only positive change observed in lobster landings is for Isla Natividad in 2014 ($p < 0.1$). The three years of post-implementation data for Maria Elena do not show a significant effect of the reserve. Isla Natividad shows higher revenues after the implementation of the reserve, as compared to the control communities. However, these changes are not significant and are associated to increased variation. Full tables with model coefficients are presented in the supplementary materials (S4 Table, S5 Table.

3.3 Governance

Although we have little information on the social dimension of these fisheries, we can use the social-ecological systems framework to analyze the performance of each governance system (Table **??**). Our analysis shows that all communities analyzed share similarities in their Governance system which is based on cooperatives (GS5.2.3.2), with strong rules in use that include Operational rules (GS6.2), Collective-choice rules (GS6.3), Constitutional rules (GS6.3), and even Territorial use communal rights (GS6.1.4.3). However, we identified important differences in terms of the actors, resource systems, and resource units. Although all communities show a high level of leadership (A5), the level of trust (A6.1) is lower in Punta Herrero. In general, the presence and success of conservation initiatives depends on the incentives of local communities to maintain a healthy status of the resources they depend upon (Jupiter et al., 2017). The enabling conditions for conservation seem to be strongly present in all communities. Due to the clarity of access rights and isolation, the benefits of conservation directly benefit the members of the fishing cooperative. These conditions have favored the development of efficient community-based enforcement systems.

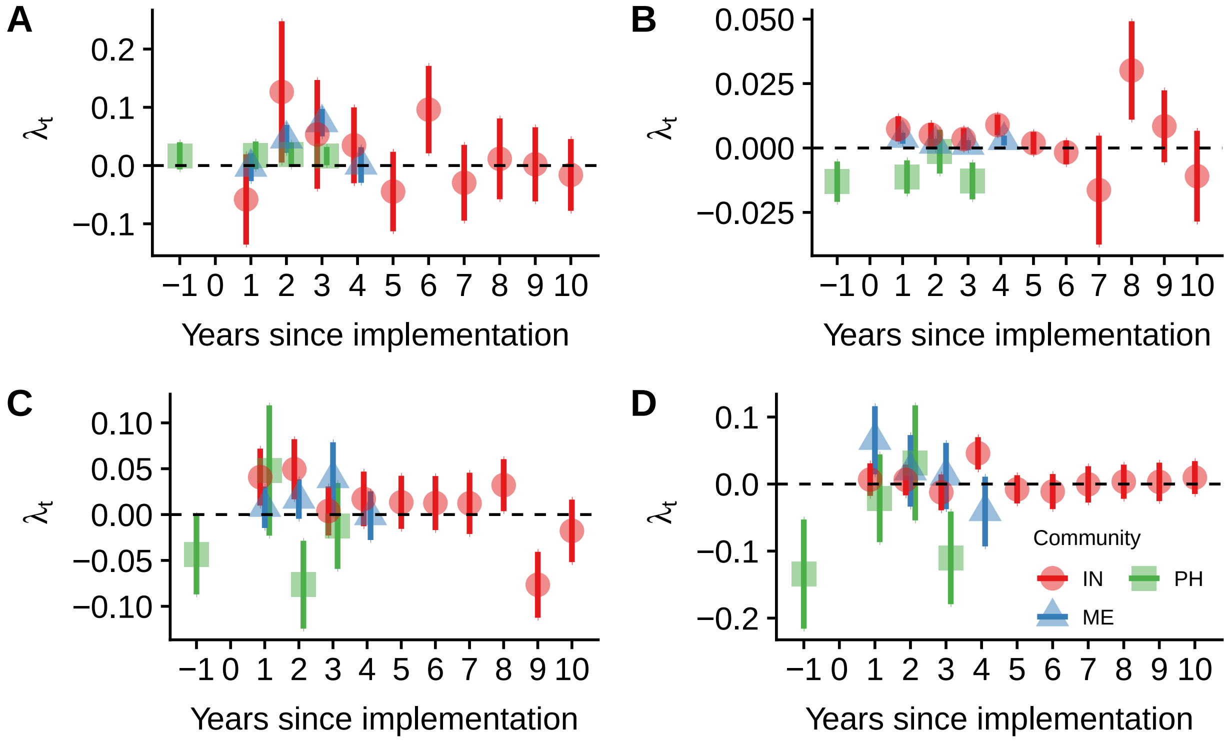

Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red cirlcles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type (left column: invertebrates; right column: fish). Points are jittered hotizontally to avoid overplotting. Points indicate the effect size and standard errors. Years have been centered to year of implementation.

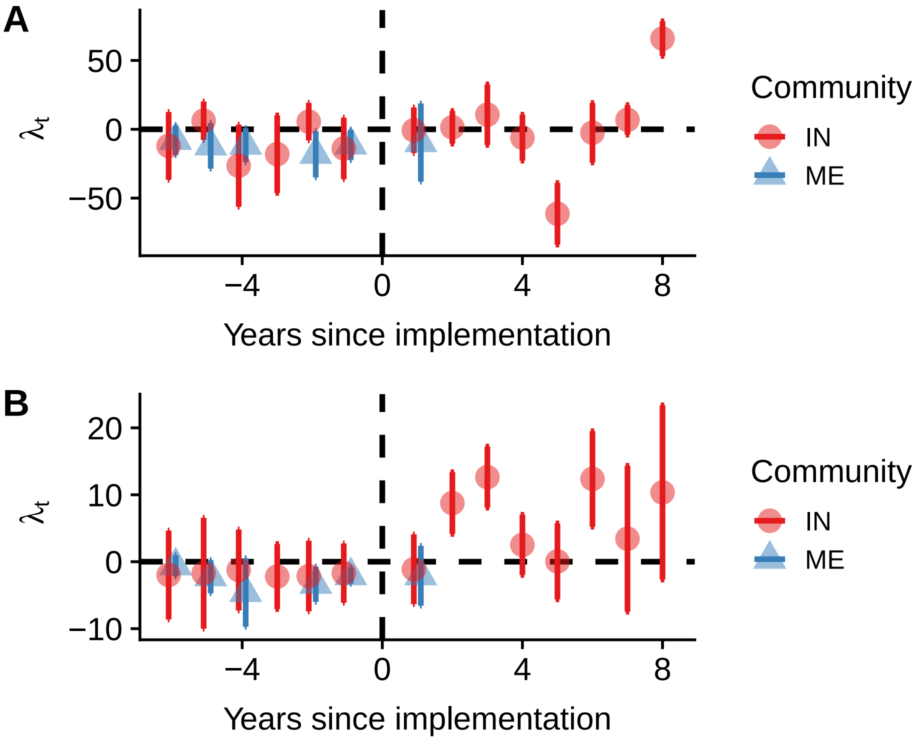

Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points indicate the effect size and standard errors. Years have been centered to year of implementation.

4 DISCUSSION

Our results indicate that these TURF-reserves have not increased lobster densities. Additionally, no co-benefits were identified when using other ecological indicators other than the previously reported buffering effect that reserves can have to environmental variability in Isla Natividad (Micheli et al., 2012). The socioeconomic indicators pertaining landings and revenues showed little to no change after reserve implementation. The coastal ecosystems where these reserves are located have been profoundly affected by climatic and oceanographic extremes, including warming events, extreme storms and prolonged hypoxia (Micheli et al., 2012), (**Beas et al. in prep., Woodson et al in review**). Despite the lack of evidence of the effectiveness of these reserves, most of the communities show a positive perception about their performance and continue to support their presence (Ayer et al., 2018). Understanding the social-ecological context in which these communities and their reserves operate might provide insights to this.

Some works evaluate marine reserves by performing inside-outside (Guidetti et al., 2014; Friedlander et al., 2017; Rodriguez and Fanning, 2017) or before-after comparisons (Betti et al., 2017). The first approach does not address temporal variability, and the second can not distinguish between the temporal trends in a reserve and the entire system (De Palma et al., 2018). Our approach to evaluate the temporal and spatial changes provides a more robust measure of reserve effectiveness. For example, we capture previously described patterns like the rapid increase observed for changes in lobster densities for Isla Natividad on the sixth year (*i.e.* 2012; Fig. 2A) occur a year after the hypoxia events described by Micheli et al. (2012), which caused mass mortality of sedentary organisms such as abalone and sea urchins, but not lobster and finfish. Yet, our empirical approach assumes control sites are a plausible counterfactual for treated sites. This implies that treated sites would have followed the same trend as control sites, had the reserves not been implemented. Nonetheless, temporal trends for each site don't show any significant increases, supporting our findings of lack of change in the indicators used.

A first possible explanation for the lack of effectivenes may be the young age of the reserves. Literature shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al., 2014; Babcock et al., 2010). Isla Natividad has the oldest reserves, and our SES analysis suggests that all communities have a well-established community-based enforcement system. With these characteristics, one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves (*i.e.* $<$ 5 years old) and effects may not yet be evident due to the short duration of protection, relative to the life histories of the protected species; other community-based marine reserves in tropical ecosystems may take up to six years to show a spillover effect (da Silva et al., 2015).

Another key condition for effectiveness is reserve size (Edgar et al., 2014), and the lack of effectiveness can perhaps be attributed to poor ecological coherence in reserve design (*sensu* Rees et al. (2018). Previous research has shown that reserves in Isla Natividad yield fishery benefits for the abalone fishery (Rossetto et al., 2015). Abalone are less mobile than lobsters, and perhaps the reserves provide enough protection to these sedentary invertebrates, but not lobsters. Design principles developed by Green et al. (2017) for marine reserves in the Caribbean state that reserves "should be more than twice the size of the home range of adults and juveniles", and suggest that reserves seeking to protect spiny lobsters should have at least 14 km across. Furthermore, may favor implementation of reserves that pose low fishing costs due to their small size or location. Our analysis of economic data supports this, as neither landings nor revenues showed the expected short-term costs associated to the first years of reserve implementation (Ovando et al., 2016).

Even if reserves had appropriate sizes and were placed in optimal locations, there are other plausible explanations for the observed patterns. For instance, marine reserves are only likely to provide fisheries benefits if initial population sizes are low and the fishery is poorly managed (Hilborn et al., 2006). Both lobster fisheries were certified by the Marine Stewardship Council (Pérez-Ramírez et al., 2016). Additionally, lobster fisheries are managed via species-specific minimum catch sizes, seasonal closures, protection of "berried" females, and escapement windows where traps are allowed (DOF, 1993). It is uncertain whether such a well-managed fishery will experience additional benefits from marine reserves. Additionally, Gelcich et al. (2008) has shown that TURFs alone can have greater biomass and richness than areas operating under open acces. These increased attributes perhaps minimize the difference between TURF and reserve. Further research should focus on evaluating sites in the reserve, TURF, and open access areas.

While the evaluated reserves have failed to provide fishery benefits up to now, there are a number of additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range of species and vulnerable habitat, like coral reefs. These sites can serve as an insurance against uncertainty and errors in fisheries management, as well as environmental shocks (Hilborn et al., 2004, 2006; Micheli et al., 2012) (**,Aalto et al, in press**). Self-regulation of fishing effort (*i.e.* reduction in harvest) can serve as a way to compensate for future declines associated to environmental variation (Finkbeiner et al., 2018). Embarking in a marine conservation project can bring the community together, which promotes social cohesion and builds social capital (Fulton et al., 2019). Furthermore, showing commitment to marine conservation and sustainable fishing practices allows fishers to have greater bargaining power and leverage over fisheries management (Pérez-Ramírez et al., 2012).

Previous studies have evaluated the potential of implementing marine reserves in Baja California and connect them to the existing network in California (Arafeh-Dalmau et al., 2017). Community-based marine reserves in small-scale fisheries can be helpful conservation and fishery management tools when appropriately implemented. Lessons learned from these cases can guide implementation of community-based marine reserves elsewhere. For the particular case of the marine reserves that we evaluate, the possibility of expanding reserves or merging existing polygons into larger areas should be evaluated and proposed to the communities. At the broader scale, having full community support surely represents an advantage, but it is important that marine reserves meet essential design principles such as size and placement. Community-based marine reserves might have more benefits that result from indirect effects of the reserves, which should be taken into account when evaluating the outcomes of similar projects.

CONFLICT OF INTEREST STATEMENT

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

AUTHOR CONTRIBUTIONS

JC and EA analyzed and interpreted data, discussed the results, and wrote the first draft. AHV coordinated fieldwork and collected the data. AS, AHV, SF, JT, and FM discussed the results and edited the manuscript.

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FIGURE CAPTIONS