Management and effectiveness of community-based marine reserves in small-scale fisheries

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ABSTRACT

Al finalizar revisiones

Keywords: Marine Protected Areas, Marine Conservation, Small-Scale Fisheries, Citizen Science, Mexico, Social-Ecological Systems

Last update: 2018-06-08

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). A common approach to manage the spatial distribution of fishing effort to recover stocks and preserve biodiversity is through the implementation of marine reserves (MRs). These areas allow bounded populations to recover by limiting all extractive activities (Halpern and Warner, 2002). The science of MRs has largely focused on understanding the ecological effects of these areas, which include increased biomass, 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 the fishery benefits of marine reserves depend on initial stock status and the type of management under which the fishery operates (Hilborn et al., 2006), as well as reserve size and the amount of larvae exported from these (Krueck et al., 2017). Other research has focused on the relationship between socioeconomic and governance structures and their relationship to ecological effectiveness (Halpern et al., 2013; López-Angarita et al., 2014; Mascia et al., 2017). However, few studies simultaneously evaluate MRs from all these perspectives (*e.g.* López-Angarita et al. (2014)).

While ecological factors like habitat representation within the MR or connectivity to other areas can determine the success of a MR, its effectiveness also depends on the socioeconomic and governance settings under which they are implemented and managed. Literature shows that many non-ecological characteristics can play an equally important role in the effectiveness of MRs. For example, age of a reserve (*i.e.* time since its implementation) and size were key to the effectiveness of MRs in Palau (Friedlander et al., 2017). In the Mediterranean, Di Franco et al. (2016) identify that surveillance and enforcement, presence of a management plan, and involvement of fishers in management and decision–making along with promotion of sustainable fishing practices were the key factors that increased stock health and income to fishers. At a global level, enforcement, age, size, and isolation are important factors that determine the effectiveness of the reserves (Edgar et al., 2014). However, MRs represent part of a more complex social-ecological systems governed by the interaction between humans and the environment. In this work we evaluate community-based marine reserves in Mexico using a triple bottom line approach by using ecological, socioeconomic, and governance indicators that provide a holistic view of the reserves (Halpern et al., 2013).

There are three main approaches to implement MRs, each of which have its pros and cons. We describe them in the context of Mexican MRs, but argue that at least the first two apply elsewhere. MRs were historically implemented and managed by a government agency, in this case the National Commission of Protected Areas (*Comisión Nacional de Áreas Marinas Protegidas*, CONANP). While CONANP has made efforts to have a participatory process, the implementation of these zones is still mainly a top-down process. A second approach is the implementation of community-based marine reserves within areas of exclusive access (*i.e.* TURFs), thus making them TURF-reserves (Afflerbach et al., 2014). Community-based spatial closures ocurr in other places, like the *kapu* or *ra'ui* areas in the Pacific Islands (Bohnsack et al., 2004; Johannes, 2002). This bottom-up approach increases compliance and self-enforcement (Gelcich and Donlan, 2015; Espinosa-Romero et al., 2014; Beger et al., 2004). However, these lack legal recognition and rely on the exclusive access granted by the TURF, making it impossible to enforce without one. In an effort to provide a legal framework for these reserves, Civil Society Organizations (CSOs) served as the bridge between fishers and government. In 2014, a new norm (NOM-049-SAG/PESC, 2014) allowed fishers to request the legal recognition of a community-based reserve under the name of "Fishing Refugia" (*Zona de Refugio Pesquero*). These can be implemented as temporal or partial eservs, which can protect one, some, or all resources within them. Since then, **39** of these have been implemented along the Pacific, Gulf of California, and Mexican Caribbean coastlines, but no formal evaluation of their effectiveness has taken place.

This work combines causal inference techniques and the social-ecological systems framework to provide a holistic evaluation of community-based marine reserves in three coastal communities in Mexico. The objective of this work is twofold. First, provide a triple bottom line evaluation of the effectiveness of community-based marine reserves that can inform similar processes in other countries. And second, perform the first formal evaluation of Fishing Refugia in Mexico and identify areas where improvement or adjustment might result in increased effectiveness. On both cases, we draw from the lessons learned and provide management recommendations to ensure or improve the effectiveness of community-based marine reserves.

2 MATERIALS AND METHODS

2.1 Study area

We evaluate community-based marine reserves from three coastal communities located in the Pacific coast of Baja California and the Mexican Caribbean (Fig 1). All communities are organized as fishing cooperatives that hold Territorial Use Rights for Fisheries (TURFs). Isla Natividad (IN) lies west of the Baja California Peninsula (Fig 1B), where kelp forests and rocky reefs are the predominant habitats. The island is home to the *Sociedad Cooperativa de Producción Pesquera (SCPP) Buzos y Pescadores de la Baja California*, whose main resource by value is the spiny lobster (*Panulirus interruptus*). However, other resources like finfish (yellow-tail jack, *Seriola lalandi*), sea cucumber (*Parastichopus parvimensis*), red sea urchin (*Mesocentrotus franciscanus*), snail (*Megastraea turbanica* y *M. undosa*), and abalone (*Haliotis spp*) are also important sources of income. In 2006, the community decided to implement two community-based marine reserves within their fishing grounds to protect commercially important invertebrate species (mainly lobster and abalone). Until today, these reserves are yet to be legally recognized as Fishing Refugia but count with full support from the community.

The other two communities are Maria Elena (ME) and Punta Herrero (PH; Fig 1C) in the Yucatan Peninsula, where coral reefs and mangroves are the representative coastal ecosystems. ME is a fishing camp –visited intermitently during the fishing season– belonging to the Cozumel fishing cooperative (*SCPP Cozumel*); PH is home to the *SCPP José María Azcorra* cooperative. Their main fishery is the Caribbean spiny lobster (*Panulirus argus*), but they also target finfish in the off season, mainly snappers (Lutjanidae) and groupers (Serranidae). ME established eight marine reserves in 2012, and PH established four marine reserves in 2013 and an additional community-based (*i.e.* not legally recognized). All these reserves are legally recognized as Fishing Refugia.

2.2 Data collection

We use three main sources of information to evaluate these reserves. 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 in the reserves and control sites. Size structures are also collected for fish evaluations. 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 for sites where data are avaialable 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 reserves included in this study.

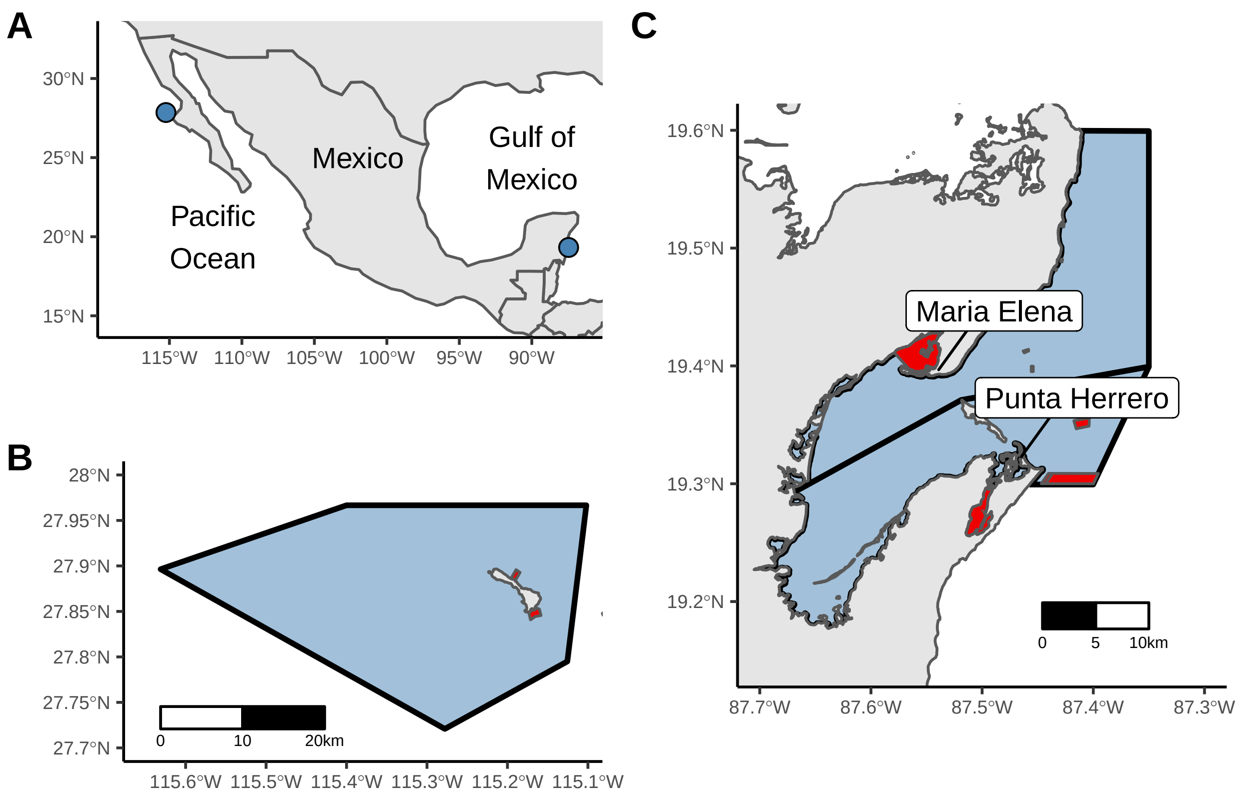

Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena (C) and Punta Herrero (D) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

Table 1. Summary of commuity–based marine reserves by community.

Community	TURF area (km^2)	Reserve area (km^2)	Percent as reserves	Year of implementation
Isla Natividad	889.5	1.53	0.1720067	2006
Maria Elena	353.1	0.10	0.0283206	2012
Punta Herrero	299.7	0.43	0.1434768	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 value (MXP) from 2000 to 2014 for cooperatives with and without marine reserves (**Fig S1**). This information was aggregated by year, and economic values were adjusted by the Consumer Price Index (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 qualitative analysis 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) 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 (**S1 Table**).

2.3 Data analysis

We evaluate the effect that marine reserves have had on four biological 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, since finfish are targeted in the off-season and because there might be associated cobenefits of the reserve.

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

Category	Indicador	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 the biological indicators. This approach allows us to estimate the effect that the reserve has on the biological indicators by comparing trends across time and treatments (*i.e.* reserve / control sites Moland et al. (2013); Villaseñor-Derbez et al. (2018)). The analysis is performed with a multiple linear regression of the form:

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

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 entire benthic or fsh communities, we include σ_j to control for species-fixed effects.

Socioeconomic indicators are evaluated with a similar approach (Eq 3). Due to data constrains, only Isla Natividad and Maria Elena are evaluated in this case. We constructed panel-data information with yearly (2001 - 2014) lobster landings and income of the studied and neighbouring communities that have similar management strategies, and belong to larger Cooperative Federtations (McCay, 2017; Ayer et al., 2018). Neighbouring communities are used as counterfacutals that allow us to control for unobserved time-invariants. Each "treated" community (Isla Natividad and Maria Elena) has three counterfactual communities.

$$I = \alpha + \gamma_t Year_t + \beta Treated_i + \lambda_t Year_t \times Treated_i + \sigma_j Com_j + \epsilon \tag{3}$$

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 regressions allows us to make 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_2018).

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 dimensions might be related to the effectiveness of the reserves.

3.1 Biological

Indicators showed idiosyncratic responses through time for each community. Figure 2A shows positive effect sizes for lobster densities for Isla Natividad and Punta Herrero during the first years, but the effect is eroded through time. These effects are in the order of 0.2 extra organisms m^{-2} but are not significantly different from zero ($p > 0.05$). Lobster densities were only significantly positive for Isla Natividad on the sixth year (*i.e.* 2012; $p < 0.05$), a year after the hypoxia events described by Micheli et al. (2012) caused mass mortality of organisms. 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 (**S2 Table**, **S3 Table**, **S4 Table**).

3.2 Socioeconomic

Lobster landings and revenue were only available for Isla Natividad and Maria Elena (Fig 3). For all years befor implementation, the effect sizes are close to zero, indicating that the control and treatment sites track each other well and that these are plausible controls. However, effect sizes do not change after the implementation of the reserve. Again, the negative coeficient 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 trhee years of post-implementation data for Maria Elena do not show a significant effect of the reserve. Isla Natividad shows higher revenues after the iumplementation of the reserve, as compared to the control communities. However, these changes are also not ignificantly different and present an increased variation. All regression coefficients for each community and indicator are presented in **S5 Table**.

3.3 Governance

Although we have little information on the social dimension of these fisheries, we can use the social-ecologycal systems framework (**S1 Table**) to analyze the performance of each governance system (**S6 Table**).

Our analysis shows that all of the systems 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. The value of lobster in isla natividad is higher than the lobster sold from Punta Herrero and Maria Elena (RU4), which can reduce the pressure on harvest. Lastly, in terms of actors, 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 an efficient community-based enforcement systems.

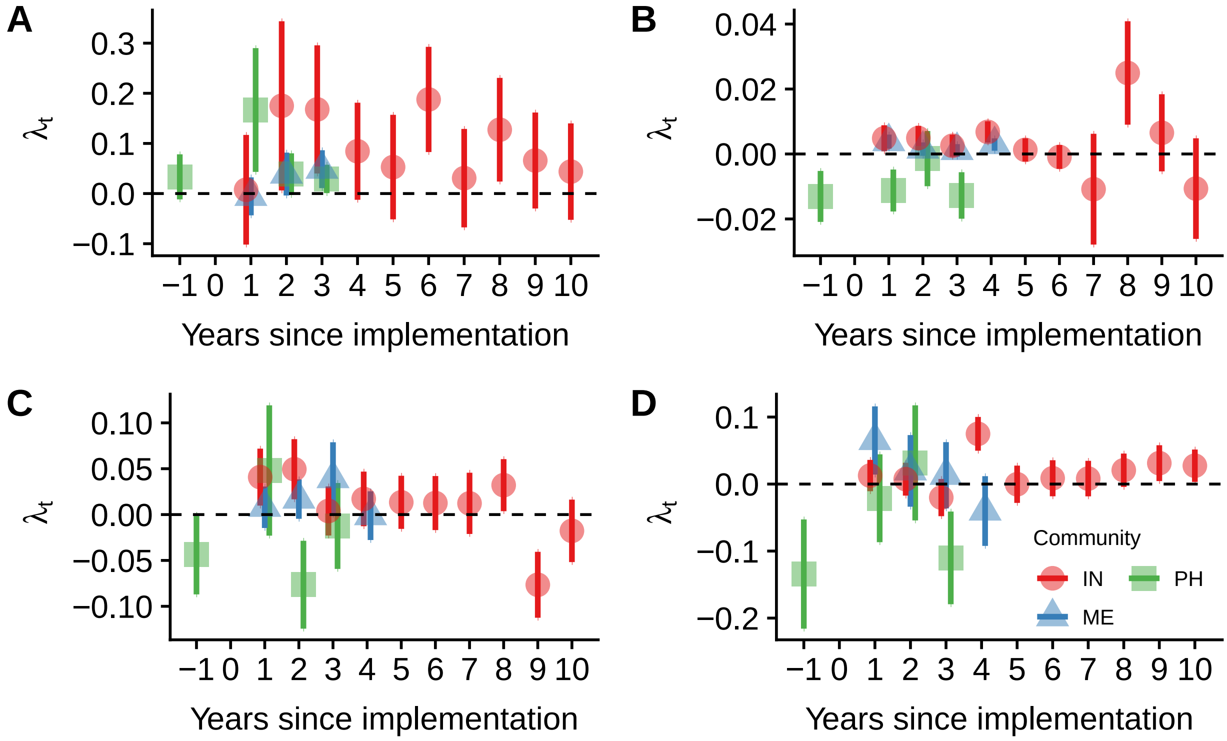

Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red cirlcles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

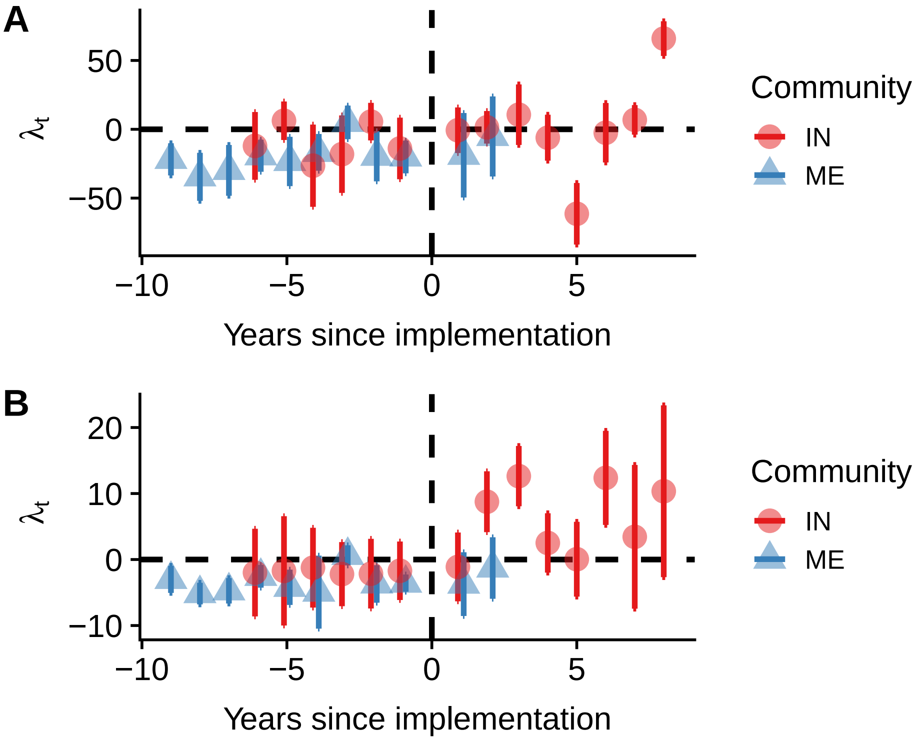

Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles)

4 DISCUSSION

Our results indicate that marine reserves are ineffective in increaseing population sizes of lobsters in the three communities. The socioeconomic indicators pertaining to landings and revenues showed little to no change after reserve implementation. While lobster densities represent an indicator directly tied to reserve objectives, we also evaluate other biological indicators to test for additional effects and fail to detect any effect other then the already reported buffering effect that Natividad reserves can have to environmental variability (Micheli et al., 2012). The lack of expected effectiveness poses the question, why do these communities continue to support the reserves? Understanding the social-ecological context in which these communities and their reserves operate might provide insights to this question. Here, we discuss plausible explanations to lack of effectiveness observed onthe biological and socioeconomic indicators based on our social-ecological system analysis. We also discuss potential shortcomings in our analysis, and provide management recommendations to improve reserve effectiveness.

Our approach to evaluate the temporal and spatial changes of each indicator provides a more robust measure of reserve effectiveness. Some works have solely focused on an inside-outside comparison of indicators (Guidetti et al., 2014; Friedlander et al., 2017; Rodriguez and Fanning, 2017), which do not address temporal variability (De Palma et al., 2018). Other works have compared the trend observed within a reserve through time (Betti et al., 2017), which cannot distinguish between the temporal trends in a reserve and the entire system (De Palma et al., 2018). By accounting for trends between sites and through time, we can control for time and space dynamics, and provide a better identification of the effect.

Age, isolation, and enforcement are important factors influencing effectiveness of a marine reserve (Edgar et al., 2014). Isla Natividad has the oldest reserve, is fairly isolated, and has a well-established community-based enforcement system. Neighbouring fishing communities are known to be well organized with successful resource management of their resources (McCay, 2017; McCay et al., 2014). Maria Elena and Punta Herrero are relatively young reserves (Table 1). With the age, relative isolation, and enforcement level of these reserves, one would expect to observe effectiveness. However, another key feature of effective MRs is size (Edgar et al., 2014); the lack of effectiveness is perhaps attributed to the reserves being too small 1. Furthermore, perturbations that do not distinguish reserve boundaries, such as the environmental variability observed in Isla Natividad can also hinder effectiveness. The possibility of increasing reserve size or merging existing networks into a larger reserve should be evaluated.

Our analysis of landings and revenues does not identify detectable changes in these indicators. However, previous research has shown that reserves in Isla Natividad yield fishery benefits for the abalone fishery (Rossetto et al., 2015). Abalone are sesile invertebrates with less mobility (compared to lobsters), and thus current reserve size might not be enough for lobster's range of mobility even when accounting for reserve age in Isla Natividad. Other community-based marine reserves in tropical ecosystems have taken up to six years to show a spillover effect (da Silva et al., 2015). Reserves in Maria Elena and Punta Herrero are relatively small and young, and may need more time for abundances to increase enough to export larvae or adult organisms.

There are two plausible explanations for the observed lack of effectiveness. First, marine reserves are only likely to provide fisheries benefits if initial population sizes are low (Hilborn et al., 2006) and are poorly managed. However, both stocks were at some point certified by the Marine Stewardship Council (Pérez-Ramírez et al., 2016). The fishery is managed via species-specific minimum catch sizes, seasonal closures, and escapement windows on traps DOF (1993). A second plausible explanation lies in the design of these areas. Small reserves are unlikely to have a significant effect on population size if they fail to protect a significant portion of it. Intuitvely, small reserves have little or no displacement of fishing effort, and fishers do not perceive the short-term costs associated to the first years of reserve implementation (Ovando et al., 2016). On either case, these might explain why reserves were not effective, but not why they still receive community support.

While reserves fail to provide conservation or economic benefits, they provide the community with access to funding from the filantrhopic sector. Furthermore, reserves can provide a joint enterprise to bring a community together, which promotes social cohesion and social capital.

A second explanation lies on the the mismatch of objectives during the design process. The Project description (*Estudios Técnicos Justificativos*) of each MR provides little information about the followed design guidelines. However, all reserves had to be approved by the community members. Depending on the communities' capacity to look for longterm benefits, many communities may favor implementation of reserves on sites that represent a low fishing cost. As a consequence, the marine reserve have low impacts in reducng the fishing effort. Having small reserves in areas that do not compromise fishing profits might explain why this communities still support their reserves.

Although our case studies fulfilled the social requirements for effective marine reserves ("high enforcement, presence of a management plan, fisher engagement in management, and promotion of sustainable fishing"; Di Franco et al. (2016)), our results show that proper reserve design is crucial for effective marine reserves. The social-ecological systems framework allowed a systematic diganosis and compare all the different case studies, allowing us to identify and tease appart possible explanations (Basurto et al., 2013).

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 manuscript. AS, SF and JT edited the manuscript and discussed the results.

FUNDING

JCVD CONACyT + LAFF ASC SF JT

ACKNOWLEDGMENTS

The authors wish to acknowledge Arturo Hernández and Imelda Amador for contributions on the governance data, as well as pre-processing biological data. This study would have not been possible without the effort by members of the communities here mentioned, who collected the biological data.

SUPPLEMENTAL DATA

Supplementary Material should be uploaded separately on submission, if there are Supplementary Figures, please include the caption in the same file as the figure. LaTeX Supplementary Material templates can be found in the Frontiers LaTeX folder

S1 Figure

Map of control and treated sites in A and control and treated landings in B

S2 Figure

Timeseries of indicators for IN

S3 Figure

Timeseries of indicators for ME

S4 Figure

Timeseries of indicators for PH

S1 Table

Coefficient estimates for Isla Natividad

S2 Table

Coefficient estimates for Maria Elena

S3 Table

Coefficient estimates for Punta Herrero

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FIGURE CAPTIONS