Enabling conditions for effective community-based marine reserves in small-scale fisheries

Juan Carlos Villaseñor-Derbez[1,*], **Eréndira Aceves-Bueno**[1,*], **Álvin Suarez-Castillo**[2], **Stuart Fulton**[2], **Jorge Torre**[2]

[1] *Bren School of Environmental Science and Management, University of California, Santa Barbara, Santa Barbara, CA, USA*
[2] *Comunidad y Biodiversidad A.C., Guaymas, Mexico*

Correspondence*:
Juan Carlos Villaseñor-Derbez, Bren Hall, University of California, Santa Barbara, Santa Barbara, CA, 93106
jvillasenor@bren.ucsb.edu

ABSTRACT

Coastal marine ecosystems provide livelihoods for small-scale fishers and coastal communities around the world. However, overfishing and unsustainable fishing practices threaten the marine environment and jeopardize the wellbeing of coastal communities. A common approach to protect the environment and recover overexploited stocks is to implement no-take marine reserves (areas where all extractive activities are off-limits). In small-scale fisheries, these are sometimes implemented as community-based reserves, where a group of fishers collectively agree to close an area to fishing. While we know that reductions in fishing effort are followed by a series of ecological benefits (increased biomass, abundance, and species diversity), we do not fully understand how environmental and governance dynamics influence the conservation and fisheries benefits of community-based marine reserves. In this work, we evaluate the ecological outcomes of four reserves established by three coastal communities in temperate and tropical ecosystems of Mexico. By combining causal inference techniques with an operationalization of the social-ecological systems framework, we identify the environmental and social conditions that enable reserve effectiveness. Our results show a strong interaction between environmental variation and community organization, which influences reserve effectiveness. For example, the most effective reserve had strong governance structures accompanied with low environmental variability. Thus, even when a community is well organized (and reserves are well enforced), environmental variation can hinder the benefits of a reserve, and vice versa. Our results are particularly relevant under present changing climate conditions, as they can better inform management and decision making.

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

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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 and recover stocks is through the implementation of marine reserves (*i.e.* areas where all fishing activities are off–limits; MRs) (Afflerbach et al., 2014; Krueck et al., 2017; Sala and Giakoumi, 2017).

Marine reserve science has largely focused on understanding the ecological effects of these areas, which include increased biomass, richness, and densities of organisms within the protected regions (Lester et al., 2009; Giakoumi et al., 2017; Sala and Giakoumi, 2017), climate change mitigation (Roberts et al., 2017), and protection from environmental variability (Micheli et al., 2012). However, there is considerably less literature focusing 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) or benefits to fisheries (Krueck et al., 2017); evaluations of marine reserves rarely provide a holistic view of the social-ecological system (López-Angarita et al., 2014). Here, we combine causal inference techniques (De Palma et al., 2018) and the social-ecological systems framework (Ostrom, 2009) to provide a comprehensive ecological and socioeconomic evaluation of four community-based marine reserves in three coastal communities in Mexico.

Marine Reserves in Mexico have been commonly implemented as "core zones" within Biosphere Reserves that are administered by 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 characterized by top–down approaches. This motivated Civil Society Organizations (CSOs) to work with coastal communities to implement community–based marine reserves (Uribe et al., 2010), which are usually established within a Territorial Use Rights for Fisheries (TURFs); thus making them TURF–reserves (Afflerbach et al., 2014). This bottom–up approach allows fishers to design their own reserves, which increases compliance and self–enforcement (Gelcich and Donlan, 2015; Espinosa-Romero et al., 2014; Beger et al., 2004). However, these reserves still lack legal recognition, making them vulnerable to poaching. 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*, FR). This new norm thus combines bottom–up approaches to design marine reserves, along with a legal recognition of the management intervention. Since then, *39* FR have been implemented along the Pacific, Gulf of California, and Mexican Caribbean coastlines, but no formal evaluation of their effectiveness has taken place.

While there are ecological factors defining the success of a MR (*i.e.* habitat representation, initial state of protection, connectivity to other protected areas), their effectiveness also depends on the socioeconomic and governance settings under which they are implemented. 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), size, and habitat contained 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, Edgar et al. (2014) indicate that enforcement, age, size, and isolation were important factors determining effectiveness of the reserves.

The objective of this work is twofold: i) Provide the first evaluation of community–based marine reserves in Mexico, and ii) provide a comprehensive evaluation of the social–ecological system to identify how socioeconomic and governance characteristics relate to ecological effectiveness. With the purpose of providing a holistic evaluation, we combine ecological, socioeconomic, and governance indicators. We use causal inference techniques to provide a measurement of the effect of the management intervention, and combine it with the social–ecological systems framework (Ostrom, 2009).

2 MATERIALS AND METHODS

2.1 Study area

We focus our evaluation in three coastal communities from the Pacific coast of Baja California (n = 1) and the Mesoamerican Reef System (n = 2; Fig 1). Isla Natividad (IN) lies west of the Baja California Peninsula (Fig 1B), where kelp forests (*Macrocystis pyrifera*) and rocky reefs are the predominant habitats. The island is home to a fishing cooperative (*Sociedad Cooperative de Producción Pesquera Buzos y Pescadores de la Baja California SCL*), that holds a TURF for 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*, until 2010) are also important sources of income. In 2006, the community decided to implement two community–based marine reserves within their fishing grounds, seeking to recover depleted stocks of invertebrate species (mainly lobster and abalone). Until today, these reserves are yet to be legally recognized as Fishing Refugia.

The other two communities are Maria Elena (ME; Fig 1C) and Punta Herrero (PH; Fig 1D) in the Yucatan Peninsula, where coral reefs and mangroves are the representative coastal ecosystems. ME is a fishing camp –visited intermittently during the fishing season– belonging to the Cozumel fishing cooperative. PH is home to the "José María Azcorra" fishing cooperative. The main source of income to both communities is the Caribbean spiny lobster fishery (*Panulirus argus*), which is carried out within their respective TURFs. These communities 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. All these reserves are legally recognized as Fishing Refugia.

2.2 Data collection

To perform the evaluation of these reserves we use thee sources of information. 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). These monitorings record richness and abundances of fish and invertebrate species in the reserves and control sites. For fish census, size structures are also collected to derive biomass. We define control sites as regions with habitat characteristics similar to the corresponding reserves, and that presumably had the same probability of being selected as reserves during the design phase. From all the reserves in these three communities, we use the ones that have data 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) 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 reserves were surveyed annually from at least one year before implementation until 2016. Table 1 shows a summary of the number of reserves, year of implementation, and number of transects for each reserve.

Table 1. Summary of commuity–based marine reserves by community. Imp = Year of implementation, Start = Year of first sampling, number of fish transects in control (Cf) and reserve (Rf) sites, and number of invertebrate transects in Control (Ci) and Reserve (Ri) sites.

Community	Reserve - Control	Imp	Start	Cf	Rf	Ci	Ri
Isla Natividad	La Plana / Las Cuevas - La Dulce / Babencho	2006	2006	400	241	415	244
Maria Elena	Cabezo - Cabezo (Control)	2012	2012	44	45	27	21
Punta Herrero	El Faro - El Faro (Control)	2013	2013	39	40	24	32
Punta Herrero	Manchon - Manchon (Control)	2013	2012	43	45	27	41

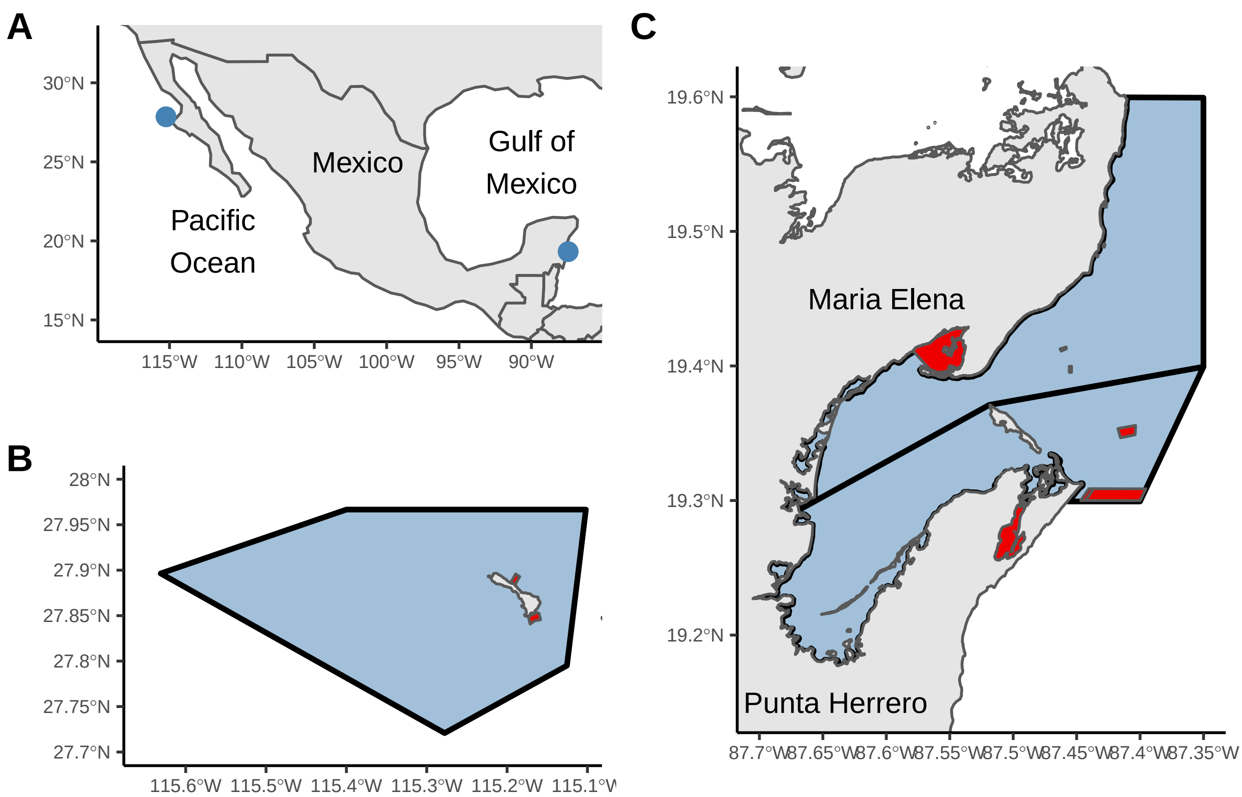

Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena (C) and Punta Herrero (D) are located in the yucatan Peninsula.

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. This information was aggregated by year, and economic values were adjusted by the Consumer Price Index (OECD, 2017) via Eq 1.

$$I_t = RI_t \times \frac{CPI_t}{CPI_T} \quad (1)$$

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).

Governance data were collected at the community-level. The information was compiled by combining key informants and the authors; experience and knowledge of the communities to collect the necessary information. These data contain information on the ecological system where the fishing activities develop, as well as the governance structures present in the cooperative. We also gathered information on the resource unit (*i.e.* lobsters) and the relevant actors present in each community (Leslie et al., 2015).

2.3 Data analysis

Following a framework that relates reserve objectives to performance indicators (Villaseñor-Derbez et al., 2018), we use five biological and two socioeconomic indicators to evaluate these marine reserves Table 2. We also use a set of governance indicators to analyze the governance structures of each cooperative (Leslie et al., 2015). The indicators (Table 3) focus on the resource system (four indicators), governance system (seven indicators), resource units (three indicators) and actors (three indicators).

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

Category	Indicador
Biological	Abundance
	Richness
	Shannon's diversity index
	Biomass
	Abundance of target species (lobsters)
Socioeconomic	Income from target species
	Landings from target species

Table 3. Indicators used for the operationalization of the SES framework (Leslie et al., 2015)

Indicator	Isla Natividad	Maria Elena	Punta Herrero
Resource systems (RS)			
TURF presence	Yes	Yes	Yes
Type of ecosystem	Kelp Forest / Rocky Reefs	Coral Reef	Coral Reef
Intensity of environmental Disturbance	El nino event	Hurricanes	Hurricanes
Location	Island	Coastal	Coastal
Governance systems (GS)			
Fishing cooperative	Yes	Yes	Yes
Involved actors	COBI, Stanford, REBIVI	Alianza Kanan Kay, COBI, CONANP, Coop, CONAPESCA, Oceanus, FCyRH, FHMM,	Alianza Kanan Kay, COBI, CONANP, Coop, CONAPESCA, Oceanus, FCyRH, FHMM,
Presence of an inter-cooperative structure	Fedecoop	Non	Non
Fishing Regulations	Size limits, seasonal closures, quotas	Size limits, seasonal closures	Size limits, seasonal closures
Enforcement technology	Boats	Boats	Land enforcement
MR enforcement			
Cooperative regulations			
Resource Units (RU)			
Adult targeted species mobility	1km	30km	30km
Targeted species longevity (years)			
Price of targeted species			
Actors (A)			
Leadership			
Level of illegal fishing	1	1	3
Presence of alternative livelihoods			

Biological indicators are analyzed with a difference–in–differences analysis (Eq 2), which allows us to estimate the effect that the reserve has on the biological indicators by comparing trends across time and treatments (Moland et al., 2013; Villaseñor-Derbez et al., 2018). The analysis is performed with generalized linear models of the form:

$$I_i = \alpha_i + \gamma_{it} Year_t + \beta Zone_i + \lambda_{it} Year_t \times Zone_i + \sigma_j Spp_j + \epsilon \quad (2)$$

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 represent the mean change in the indicator inside the reserve, for year t, with respect to the first year of evaluation in the control site (See Table 1). When evaluating biomass and abundances, we include species–fixed effects (σ_j). For abundances and richness (*i.e.* count data) the model is estimated with a quasipoisson error distribution.

Socioeconomic indicators are evaluated with a similar approach (Eq 3), where landings and income before and after the implementation of the reserve are compared:

$$I_i = \beta_0 + \beta_1 Post \tag{3}$$

This approach does not allow for a causal attribution of the observed changes to the reserve, but still allows us to draw important information that can inform our conclusions. For both approaches, model coefficients are estimated via ordinary least–squares and heteroskedastic–robust standard errors (Zeileis, 2004).

3 RESULTS

Our methodological approach with biological indicators 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 reserve site at time t after controlling for other time and space variations (*i.e.* γ_t and β respectively). Here we present the effect that marine reserves had on each of the biological indicators for each coastal community, along with the trends in socioeconomic indicators of lobster catches and revenues. We also provide an overview of the state of the socioeconomic and governance settings of each community, and discuss how these dimensions might be intertwined with each other.

3.1 Biological

Effect sizes for biological indicators are shown in Figure 2, and Figure 3 shows the summarized biological effects by community. Isla Natividad shows inconsistent effects across data sources (*i.e.* fish vs. invertebrates). For example, the reserve had a small effect on fish abundances (Fig 2A), where only year 2010 showed significant effect sizes in fish abundances ($p < 0.05$) and all other years oscillated above and under zero ($p > 0.05$). However, invertebrate abundances (Fig 2B) presented a positive trend relative to the control site before implementation ($p < 0.05$) for all but 1 year (2008). Maria Elena and Punta Herrero showed no significant increase in fish and invertebrate abundances ($p < 0.05$), except for invertebrates in Punta Herrero for 2014 –right after the implementation of the reserves– which showed a significant increase (*i.e.* $\lambda_{2014} = 2.5$, $p < 0.05$). Full tables with model coefficients are presented in the supplementary materials (**S1 Table**, **S2 Table**, **S3 Table**).

While the number of fish species oscillated above and bellow zero through time for all reserves, none of these changes were statistically significant ($p > 0.05$) indicating that the reserves had no effect on fish species richness (Fig 2C). For invertebrate species in Isla Natividad, all effect sizes were negative, but only significant for 2008, 2009, 2011, and 2014 ($p < 0.05$; Fig 2D). For Maria Elena and Punta Herrero, the data do not show significant changes in invertebrate species richness ($p > 0.05$).

Effect sizes for Shannon's diversity index for fish (Fig 2E) in Isla Natividad oscillated between $\lambda_{2011} = -0.45$ and $\lambda_{2010} = -0.005$, but were not significantly different from null hypotheses of no change (*i.e.* $\lambda_t = 0$; $p > 0.05$). For invertebrates in that same community (Fig 2F), Shannon's diversity index showed a significant decrease between 2008 and 2014, with largest decrease observed for 2011 ($\lambda_{2011} = -0.91$;

$p < 0.05$). In the case of Maria Elena and Punta Herrero, Shannon's diversity index for fish showed increases in the order of $\lambda_t = 1$. For Maria Elena and Punta Herrero, these effects were only statistically significant for 2014, and 2014 and 2015 ($p < 0.05$).

Biomass was only evaluated for fish data (Fig 2G). In Isla Natividad, fish biomass presented a steady but small increase ($p > 0.05$), and exhibited an increased variability in biomass between 2013 and 2016. Maria Elena and Punta Herrero also showed small, non-statistically significant increases in fish biomass ($p > 0.05$). The last biological indicator is abundance of target species, *Panulirus interruptus* and *P. argus*, for the Pacific and Caribbean, respectively (Fig 2H). Isla Natividad presented small constantly-positive effects but were not significantly different from the reference point of control site before the implementation of the reserve ($p > 0.05$. Maria Elena showed significant increases in lobster densities in the order of $\lambda_t = 10$ ($p < 0.05$). Finally, Punta Herrero presented alternating negative and positive effects, but these were not different from the baseline case ($p > 0.05$).

3.2 Socioeconomic

Lobster landings and revenue showed a increase after the implementation of the reserves for Isla Natividad and Maria Elena (Fig 4). However, the differences in catches and and revenue were not different in the periods before and after the implementation ($p > 0.05$) except for revenues in Isla Natividad, which presented a significant increase of 14.37 (M MXP; $p < 0.05$). All regression coefficients are presented in **S4 Table**.

3.3 Governance

Although we have little information on the social dimension of these fisheries, using the SES framework indicators (Table 3), we can analyze the performance of each governance system with respect to MR enforcement (Table 4). 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 Isla Natividad. 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 system. In contrast, the communities of Maria Elena and Punta Herrero are located near other fishing communities and cities. In Maria Elena, the fishing pressure caused by outsiders can be reduced by implementing a strong enforcement system (in water and land) supported by CSOs and the local government (CONANP). Lastly, the community of Punta Herrero shows the highest levels of illegal activities which can be attributed to its connectedness to other communities and the lack of appropriate technologies for enforcement.

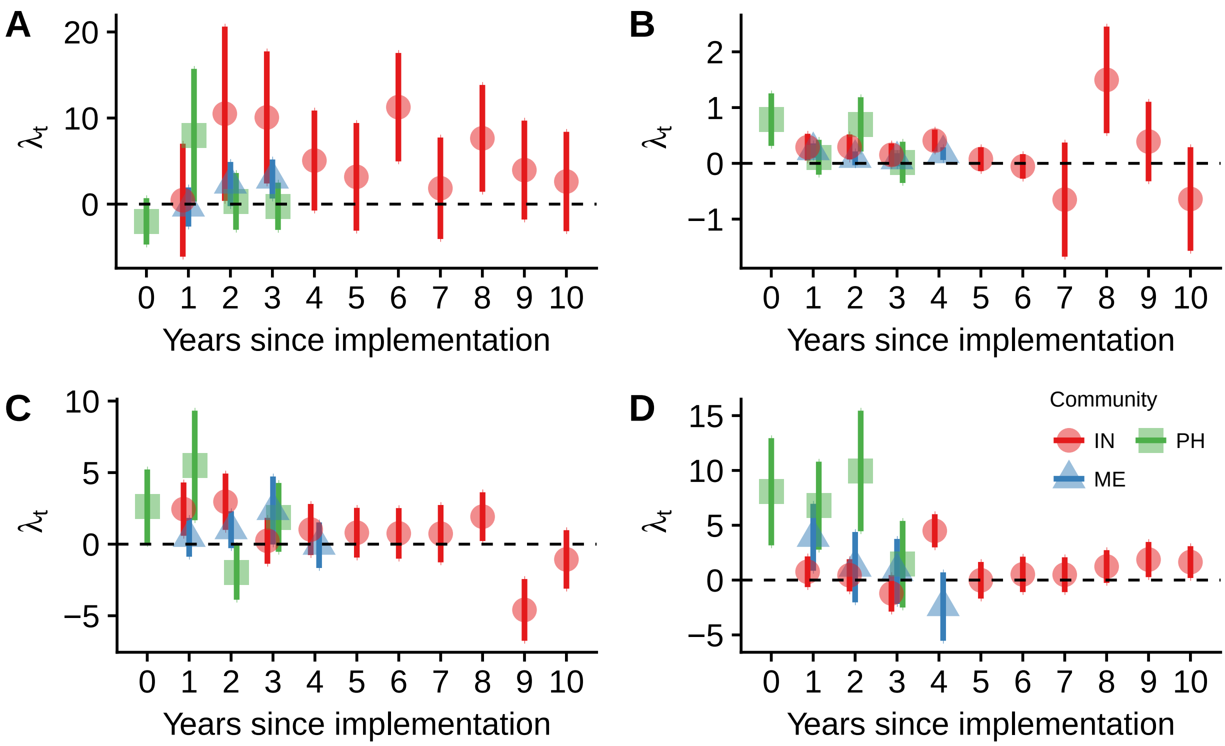

Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red cirlcles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for community-levell indicators. Plots are ordered by survey type (left: fish; right: invertebrates) and indicators: Abundance (A, B), Richness (C, D), Shannon's diversity index (E, F),fish biomass (G), and lobster (*Panulirus spp*) abundances (H). Points are jittered hotizontally to avoid overplotting. Points indicate the effect size, and errorbars are heteroskedastic-robust standard errors.

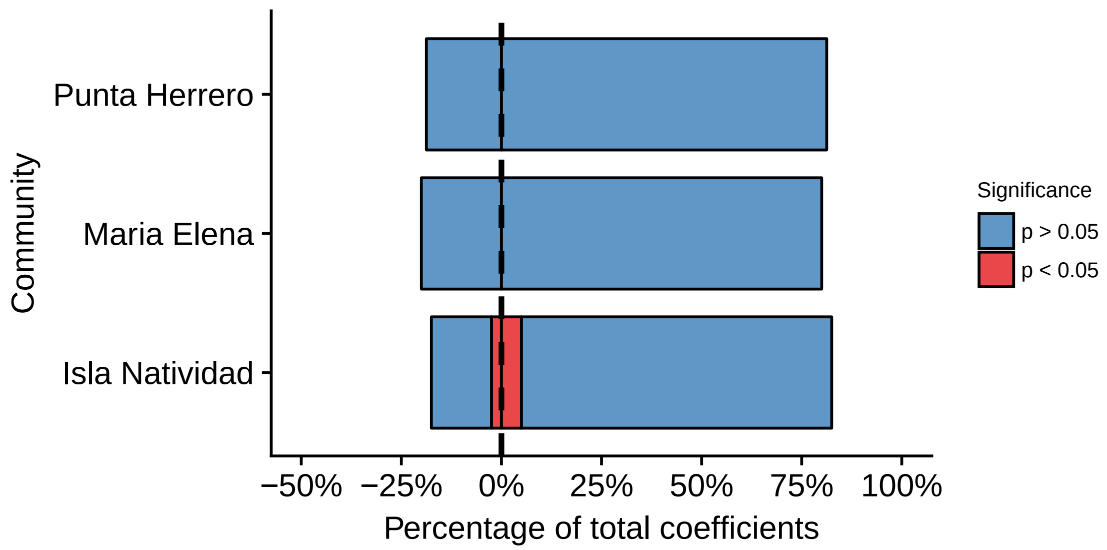

Figure 3. Summarized effects of the marne reserves by direction (positive - negative) and significance.

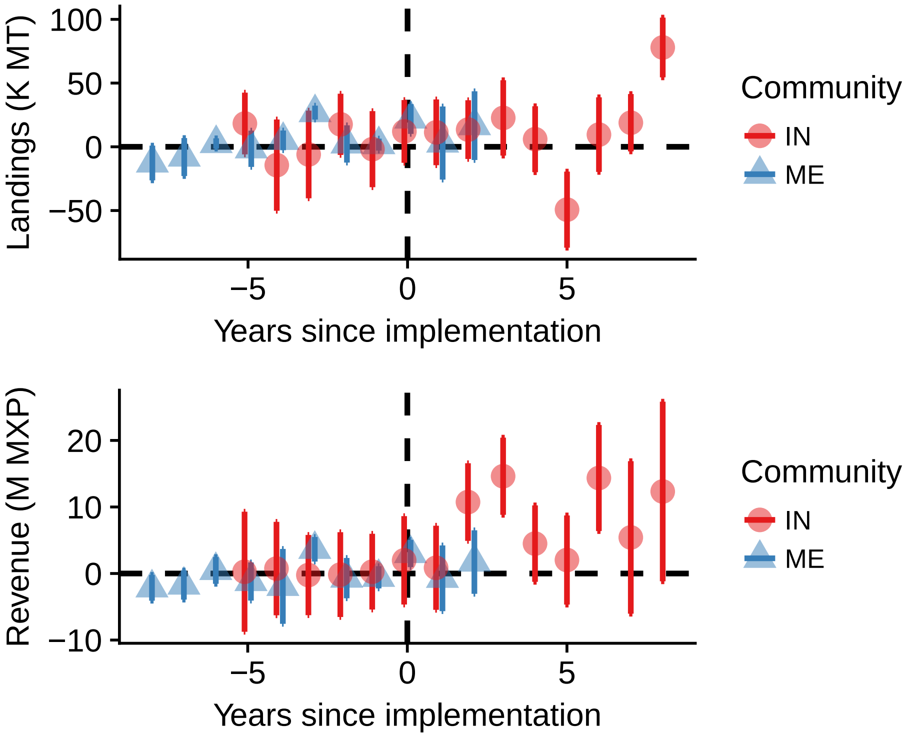

Figure 4. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles)

Table 4. Analysis of the fishing cooperatives based on the Social-Ecological systems framework (McGinnis and Ostrom, 2014).

	Indicator	Isla Natividad	Maria Elena	Punta Herrero
Resource systems (RS)				
RS2 – Clarity of system boundaries	TURF presence	High	High	High
RS3 – Size of resource system				
RS5 – Productivity of system	Type of ecosystem	High	High	High
RS7 – Predictability of system dynamics	Intensity of environmental disturbance	Low (ENSO)	High	High
RS9 – Location	Proximity to other communities/cities	Isolated	Not Isolated	Not Isolated
Governance systems (GS)				
GS1 – Government organizations	Presence of fishing cooperatives	Yes	Yes	Yes
GS2 – Nongovernment organizations	Involved actors	Yes	Yes	Yes
GS3 – Network structure	Presence of an inter-cooperative structure	Yes	No	No
GS4 – Property-rights systems	TURF presence	Yes	Yes	Yes
GS5 – Operational-choice rules	Fishing Regulations / MPA enforcement / Enforcement technolofy	Yes	Yes	Yes
GS6 – Collective-choice rules	Cooperative regulations	Yes	Yes	Yes
GS7 – Constitutional-choice rules				
Resource units (RU)				
RU1 – Resource unit mobility	Targeted species home range	Low	Medium	Medium
RU2 – Growth or replacement rate	Max age of targeted species	Low	Medium	Medium
RU4 – Economic value	Price of targeted species	high	High	high
Actors (A)				
A1 – Number of relevant actors		98		
A2 – Socioeconomic attributes				
A5 – Leadership/entrepreneurship	Leadership	High	High	High
A6 – Norms (trust-reciprocity)/social capital— (Based on illegal fishing)	Level of illegal fishing	High	High	Low
A8 – Importance of resource (dependence)	Presence of alternative livelihoods	High	High	High

4 DISCUSSION

Our results show idiosyncratic biological effects of the reserves across communities and indicators. However, many of these effects were not statistically significant, indicating no effect of the reserve 3. The socioeconomic indicators pertaining to landings and revenues associated to those landings showed little or no temporal change before and after reserve implementation. These contrasting effects, however, might be clarified when understanding the social-ecological context in which these communities and their reserves sit. In this section, we discuss potential shortcomings in our analysis, and provide plausible explanations to the observed biological and socioeconomic basing on previous literature and our analysis of the social-ecological system.

The contrasting biological effectiveness observed is perhaps explained by our approach to evaluate the temporal and spatial changes of each indicator. 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 times, we can control for time and space dynamics, and provide a better identification of the effect. However, it is worth looking deeper into each case, and identifying other plausible explanations.

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. While other communities are certainly within reach, these are known to be well organized fishing communities with successful resource management (McCay, 2017; McCay et al., 2014). The reserve at Isla Natividad presented the largest percentage of significantly positive changes in biological indicators (19%), but an important portion of was also negative (15%). With the age, relative isolation, and enforcement level of this reserve, it would be expected for it to be considerably effective. The potential gap in performance can be attributed to perturbations that do not distinguish reserve boundaries, such as environmental variability (**no recuerdo esta cita**). The region is known to be under the influence of recurrent hypoxia and high-temperature events known to cause massive adult mortalities (Micheli et al., 2012).

Maria Elena and Punta Herrero are relatively young reserves (See Table 1). From these, the Maria Elena exhibited the highest performance in terms of biological indicators (15% significantly positive). In contrast, Punta Herrero had a similar proportion of positive and negative effects.

The way in which we measure changes in catches and revenues can not identify whether the observed differences are simply caused by pre-existing temporal trends or by the implementation of the reserve. Yet, there were no detectable changes in these indicators, except for landings in Isla Natividad. Other research has shown that reserves in Isla Natividad yield fishery benefits for the abalone fishery (Rossetto et al., 2015). Since the trend was not detected in catches –directly related to abundance and fishing effort– it is plausible that these differences are purely explained by an increase in market-level prices.

The fact that there was no detectable change in catches for Maria Elena and Punta Herrero can be explained by a combination of factors related to the design, management, age, or ecological factors. Reserves in these communities are relatively small and young, and may need more time for lobster abundances to increase enough to export larvae or adult organisms. Other community-based marine reserves in tropical ecosystems have taken up to six years to show a spillover effect (da Silva et al., 2015). A complimentary explanation lies in the results observed for the governance system. The lack of enforcement in Punta Herrero, for example, could explain the lack of effectiveness observed in their reserves.

Limitations. . .

Our results show that community-based marine reserves can be effective if the environmental and social settings allow it. By studying the social-ecological system as a whole, we can provide a wider range of explanations to the patterns observed. It is interesting that even under the best enabling social conditions, climate variability can hinder the effect of a reserve –Although it is interesting to imagine what the state of that fishery had been if the reserve and organized cooperative were not present–. On the contrary, we show how under low climate variability, absence of proper governance structures can limit the effectiveness and benefits of a reserve. Whether the combination of a stable environment and governance structures are additive or multiplicative represents an interesting area for future research, especially under a changing climate.

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.

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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

Timeseries of indicators for IN

S2 Figure

Timeseries of indicators for ME

S3 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