# Effectiveness of community-based TURF-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. Small-scale 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 can 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 12 years, 18 old and new community-based TURF-reserves gained legal recognition thanks to a 2014 regulation; their effectiveness has not been formally evaluated. We combine causal inference techniques and the 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 the density of lobster and other benthic invertebrates, they continue to receive significant support from the fishing communities. A lack of clear ecological and socioeconomic effects likely results from a combination of factors. First, some of these reserves might be too young for the effects to show. Second, the reserves are not large enough to protect mobile species, like lobster. Third, variable and extreme oceanographic conditions have impacted harvested populations. Fourth, local fisheries are already well managed, and it is unlikely that reserves might have a detectable effect in landings. However, these reserves may 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, small-scale fisheries face great challenges since they tend to be hard to monitor and enforce (Costello et al., 2012). One of the many important ways to improve the performance of coastal fisheries and health of the local resources is through the implementation of Territorial Use Rights for Fisheries (TURFs) that contain no-take marine reserves within them, thus creating TURF-reserve systems (Costello and Kaffine, 2010; Afflerbach et al., 2014; Lester et al., 2017).

TURFs are a fisheries management tool in which a well defined group of fishers have exclusive access to an explicitly delimited portion of the ocean. They promote a sense of stewardship and incentivise resource users to sustainably manage their resources (Gelcich et al., 2008; McCay, 2017). On the other hand, no-take marine reserves (marine reserves from hereinafter) are areas where all extractive activities are off-limits. These can be implemented to protect biodiversity but also as fishery management tools that restrict fishing effort and gears and therefore aid in the recovery of marine stocks. Commonly known as TURF-reserves, the combination of these tools should in theory increase the benefits of spatial access rights allowing the maintenance of healthy resources (Afflerbach et al., 2014; Gelcich and Donlan, 2015; Lester et al., 2017).

Research on TURFs has shown that these areas have higher abundance of targeted species than sites operating under open access (Gelcich et al., 2008; McCay et al., 2014; McCay, 2017). Likewise, conservation science has shown how marine reserves lead to increased biomass, species richness, and abundance within the protected regions (Lester et al., 2009; Giakoumi et al., 2017; Sala and Giakoumi, 2017), and that these may have a series of additional benefits like climate change mitigation, protection from environmental variability, and fisheries benefits Roberts et al. (2017); Micheli et al. (2012); Krueck et al. (2017). The benefits resulting from reserves established within TURFs should be captured exclusively by the group of fishers with exclusive access. Although in theory these systems are successful (Costello and Kaffine, 2010; Smallhorn-West et al., 2018), there is little empirical evidence of their effectiveness and the drivers of their success (Afflerbach et al., 2014; Lester et al., 2017).

TURF-reserve systems are inherently intrincate social-ecological systems, and their effectiveness must depend on how environmental and social factors combine and interact (Gelcich and Donlan, 2015). This is especially important in social-ecological coastal systems dominated by close interactions and feedbacks between people and natural resources (Ostrom, 2009). There is a growing body of literature focusing on the relations between socioeconomic and governance structures and reserve effectiveness (Halpern et al., 2013; López-Angarita et al., 2014; Mascia et al., 2017; Bergseth et al., 2018). However, to our knowledge, no studies exist that evaluate TURF-reserves from both a social and ecological perspective.

Moreover, a new Mexican norm was created in 2014 allowing fishers to request the legal recognition of community-based reserves as “Fish Refuges” (*Zona de Refugio Pesquero*; NOM-049-SAG/PESC (2014)). Since 2012, old and new marine reserves have gained legal recognition as Fishing Refuges. Of these, 18 were originally implemented as community-based TURF-reserves. However, their effectiveness has not yet been formally evaluated and reported in the scientific literature.

Here, we combine causal inference techniques and the Social-Ecological Systems (SES) framework to evaluate community-based TURF-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 holistic evaluation of the effectiveness of community-based TURF-reserves in terms of the changes in biological and socioeconomic indicators and the governance settings under which these develop, which may 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 where this tool is used to manage and rebuild their coastal fisheries.

# 2 METHODS

## 2.1 TURF-reserves in Mexico

Before discussing our data collection methods and describing our analyses, our case studies warrant some more background. Community-based marine reserves that are implemented within TURFs are a form of TURF-reserves, voluntarily established and enforced by local communities. This bottom-up approach increases compliance and self-enforcement, and reserves can yield benefits similar to systematically-designed reserves (Beger et al., 2004; Gelcich and Donlan, 2015; Smallhorn-West et al., 2018). 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, marine reserves are difficult to enforce if they are not legally recognized, and fishers rely on the exclusive access granted by the TURF.

In an effort to bridge this normative gap, Mexican Civil Society Organizations (CSOs) served as a link between fishers and government, and created a legal framework that solves this governance issue (*i.e.* Fish Refuges; 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. One of the ways in which fishing communities have takane advantage of this new tool is by implementing marine reserves within their TURFs. Our work focuses on some of these community-based TURF-reserves that occur in small-scale fisheries.

The common setup of community-based TURF-reserves in Mexico is the following. Fishers from a given community are assambled in fishing cooperatives which have exclusive fishing rights over a spatially delimited area (*i.e.* TURFs shown as blue polygons in Fig 1A). Each TURF is exlusively fished by one cooperative, and each community usually hosts no more than one cooperative. Fishing cooperatives interested in implementing marine reserves work with CSOs to implement marine reserves within their TURFs (*i.e.* TURF-reserves). Fishers then ask the government to grant legal recognition to their TURF-reserves under the name of Fish refuges following a series of studies outlined in NOM-049-SAG/PESC (2014).

## 2.2 Study areas

We evaluate three community-based no-take marine reserves implemented in Mexican TURF-managed fisheries, therefore making them TURF-reserves (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 California 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. These reserves obtained legal recognition only in 2018 (DOF, 2018b).

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; Punta Herrero is home to the *José María Azcorra* fishing 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 and four marine reserves in 2012 and 2013, respectively. These reserves have been legally recognized as Fishing Refuges since their creation (DOF, 2012b, 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 et al., 2014; McCay, 2017; 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 that have been historically manged with TURFs 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 causal inference analysis that we undertake. Yet, given the similarities among communities and the socioeconomic and governance setting under which they operate, it is safe to cautiously generalize our insights to other similar reserves in Mexico and elsewhere around the world.

The regulation governing the implementation of Fish refuges states that these are fishery management tools intended to have biological or socioeconomic benefits (NOM-049-SAG/PESC, 2014). For this reason, the main portion of our analyses focuses on a series of biological and socioeconomic indicators that may respond to reserve implementation. However, the effectiveness of conservation and fisheries management interventions also depends on the social and governance structures in place. We therefore incorporate a reduced version of the Social Ecological Systems framework (Ostrom, 2009) and evaluate variables and indicators known to aid and hinder the effectiveness of management interventions in conservation and fisheries. The incorporation of the SES is not intended to relate different levels of governance with reserve effectiveness, but rather provide a discussion of potential causes of individual reserve performance. The following two sections describe our data collection methods and analyses of biological and socioeconomic indicators as well as the SES analysis.

## 2.3 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 (30 $\times$ 2 m each) at depths 5-20 m in the reserves and control sites (Fulton et al., 2018, 2019; Suman et al., 2010). Size structures are also collected during fish surveys. We define control sites as regions where: i) habitat characteristics are similar to the corresponding reserves, ii) presumably had a similar probability of being selected as reserves during the design phase, and iii) are located within the TURF and therefore fishing occurs. 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 (Stewart-Oaten et al., 1986; De Palma et al., 2018).

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).

Data for the evaluation of the SES were collected at the community-level from official documents used in the creation and designation of the marine reserves (DOF, 2012b, 2013, 2018b) and based on the authors' experience and knowledge of the communities. These include information on the Resource Systems, Resource Units, Actors, and Governance System (Table 2).

### 2.4 Data analysis

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 (Francini-Filho and Moura, 2008; Moland et al., 2013; Villaseñor-Derbez et al., 2018). All sites were surveyed annually, and at least once before implementation of the reserves. We evaluate the effect that marine reserves have had on four ecological and two socioeconomic indicators (Table 1). 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.

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:

$$I_{i,t,j} = \alpha + \gamma_t Year_t + \beta Zone_i + \lambda_t Year_t \times Zone_i + \sigma_j Spp_j + \epsilon_{i,t,j} \tag{1}$$

Where year-level fixed effects are represented by $\gamma_t Year_t$, 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. When evaluating biomass and densities of the invertebrate or fish communities, we include $\sigma_j$ to control for species-level fixed effects. $\epsilon_{i,t,j}$ represents the error term of the regression.

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.

$$I_{i,t,j} = \alpha + \gamma_t Year_t + \beta Treated_i + \lambda_t Year_t \times Treated_i + \sigma_j Com_j + \epsilon_{i,t,j} \tag{2}$$

The model interpretation remains as for Eq 1, 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 $\lambda_t$ 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.* $\gamma_t$ and $\beta$ respectively). All model coefficients were estimated via ordinary least-squares and heteroskedastic-robust standard errors (Zeileis, 2004). All analyses were performed in R version 3.5.1 and R Studio version 1.1.456 (R Core Team, 2018). All data and code ara available in a GitHub repository.

We use the SES framework to evaluate each community. The use of this framework standardizes our analysis and allows us to communicate our results in a common language across fields by using a set of previously defined variables and indicators. We based our variable selection primarily on Leslie et al. (2015) and Basurto et al. (2013), who operationalized and analyzed Mexican fishing cooperatives using this framework. We also incorporate other relevant variables known to influence reserve performance following Di Franco et al. (2016) and Edgar et al. (2014). Table 2 shows the selected variables, their definition and values.

## 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 $\lambda_t$). 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 fourth year. These effects are in the order of 0.2 extra organisms $\mathrm{m}^{-2}$ for Isla Natividad and Punta Herrero, and 0.01 organisms $\mathrm{m}^{-2}$ for Maria Elena, but are not significantly different from zero ($p > 0.05$). Likewise, no significant changes were detected in fish biomass or invertebrate and fish densities (Fig. 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 with increased variation. Full tables with model coefficients are presented in the supplementary materials (S4 Table, S5 Table).

### 3.3 Governance

Our analysis of the SES (Table 2) shows that all analyzed communities share similarities known to foster sustainable resource management and increase reserve effectiveness. For example, fishers operate within clearly outlined TURFs (RS2, GS6.1.4.3) that provide exclusive access to resources and reserves. Along with their relatively small groups (A1 - Number of relevant actors), Isolation (A3), Operational rules (GS6.2), Social monitoring (GS9.1), and Graduated sanctions (GS10.1), these fisheries have solid governance structures that enable them to monitor their resources and enforce rules to ensure sustainable management. In general, success of conservation initiatives depends on the incentives of local communities to maintain a healthy status of the resources upon which they depend (Jupiter et al., 2017). Due to the clarity of access rights and isolation, the benefits of conservation directly benefit the members of the fishing cooperatives, which have favored the development of efficient community-based enforcement systems. However, our SES analysis also highlights factors that might hinder reserve performance or mask outcomes. While total reserve size ranges from 0.2% to 3.7% of the TURF area, individual reserves are often small (RS3), and relatively young (RS5). Additionally, fishers harvest healthy stocks (RS4.1), and it's unlikely that marine reserves will result in increased catches.

The fact that these reserves were re-established and just enacted shows a comitment of support (issue #16 on GH). the SES provides a static picture of governance and resource conditions, whereas the biological assessment reflects changes through time (Isse #31).

## 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 aside from 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. 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 as to why this happens.

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). While many ecology studies have used BACI sampling designs and respective analyses (*e.g.* (Stewart-Oaten et al., 1986)), few conservations studies have done so to evaluate the effect of an intervention (*i.e.* Francini-Filho and Moura (2008); Lester et al. (2009); Moland et al. (2013)) which has resulted in a call for more robust analyses in conservation science (Guidetti, 2002; Ferraro and Pattanayak, 2006).

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 lobster densities in Isla Natividad on the sixth year (*i.e.* 2012; Fig. 2A), 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 (S1 Table, S2 Table, S3 Table), supporting our findings of lack of change in the indicators used.

A first possible explanation for the lack of effectiveness 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.* $<$ 6 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; community-based marine reserves in tropical ecosystems may take six years or more 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, fishers may favor implementation of reserves that pose low fishing costs due to their small size or location. Our analysis of economic data supports this hypothesis, 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., 2004, 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. Furthermore, Gelcich et al. (2008) have shown that TURFs alone can have greater biomass and richness than areas operating under open access. This might reduce the difference between indicators from the TURF and reserve sites, making it difficult to detect such a small change. Further research should focus on evaluating sites in the reserve, TURF, and open access areas or similar Fish Refuges established without the presence of TURFs where the impact of the reserves might be larger.

Finally, extreme conditions, including prolonged hypoxia, heat waves, and storms have affected both the Pacific and Caribbean regions, with large negative impacts of coastal marine species and ecosystems (Cavole et al., 2016; Hughes et al., 2018; Breitburg et al., 2018). The coastal ecosystems where these reserves are located have been profoundly affected by these events (Micheli et al., 2012; Woodson et al., in press). Effects of protection might be eliminated by the mortalities associated with these extreme conditions.

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. These sites can serve as an insurance against uncertainty and errors in fisheries management, as well as mild environmental shocks (Micheli et al., 2012; De Leo and Micheli, 2015; Roberts et al., 2017; 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). 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). Furthermore, the lack of effectiveness observed in these reserves should not be generalizable to other reserves established under the same legal framework (*i.e.* Fish Refuges) in Mexico, and future research should aim at evaluating other areas that have also been established as bottom-up processes but without the presence of TURFs (*e.g.* DOF (2012a)), or others established through a top-down process (*i.e.* DOF (2018a)).

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. Community-based marine reserves might have more benefits that result from indirect effects of the reserves, particularly providing resilience to shocks and management errors, and promoting social cohesion, which should be taken into account when evaluating the outcomes of TURF-reserves. Having full community support surely represents an advantage, but it is important that community-based TURF-reserves meet essential design principles such as size and placement so as to maximize their effectiveness.

## 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 AS conceived the idea. JC and EA analyzed data, discussed the results, and wrote the first draft. FM, SF, AS, JT, and AHV discussed the results and edited the manuscript. All authors provided valuable contributions.

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## FIGURE CAPTIONS

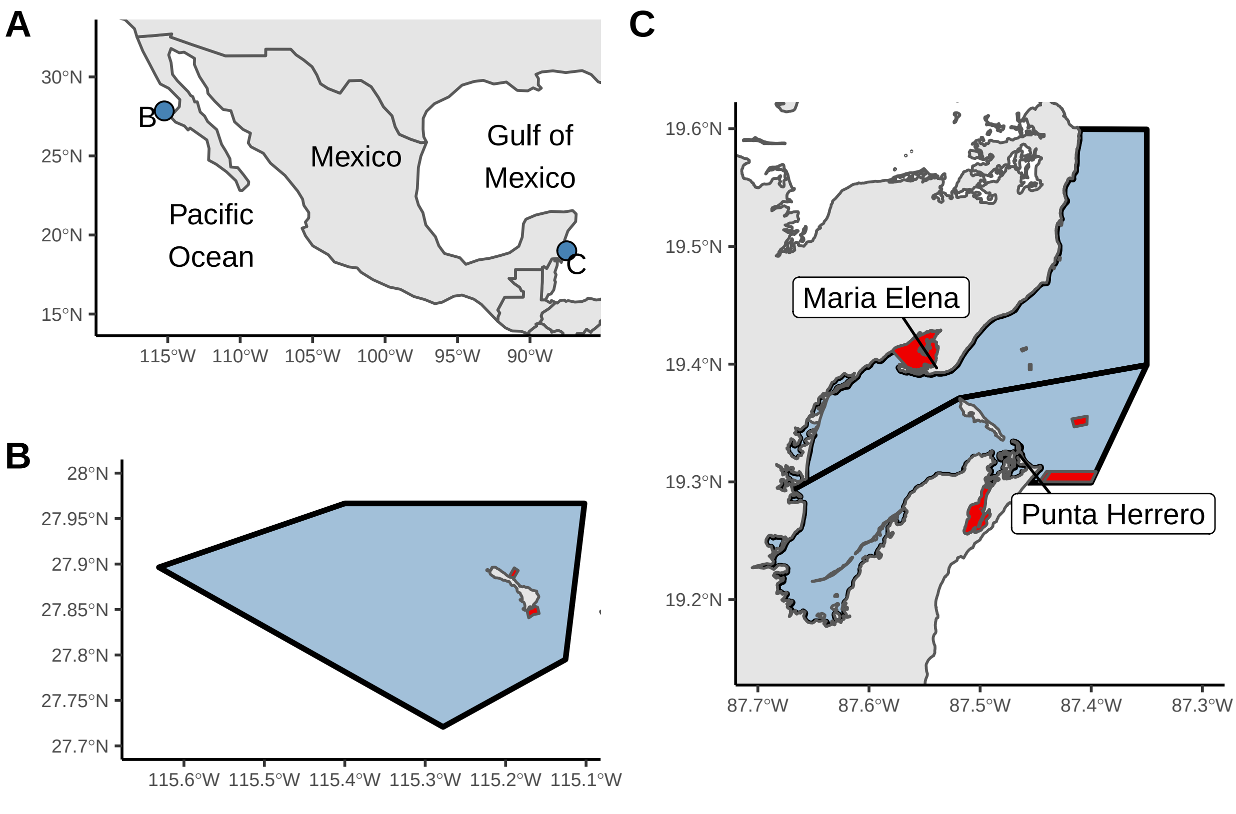


**Figure 1.** Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

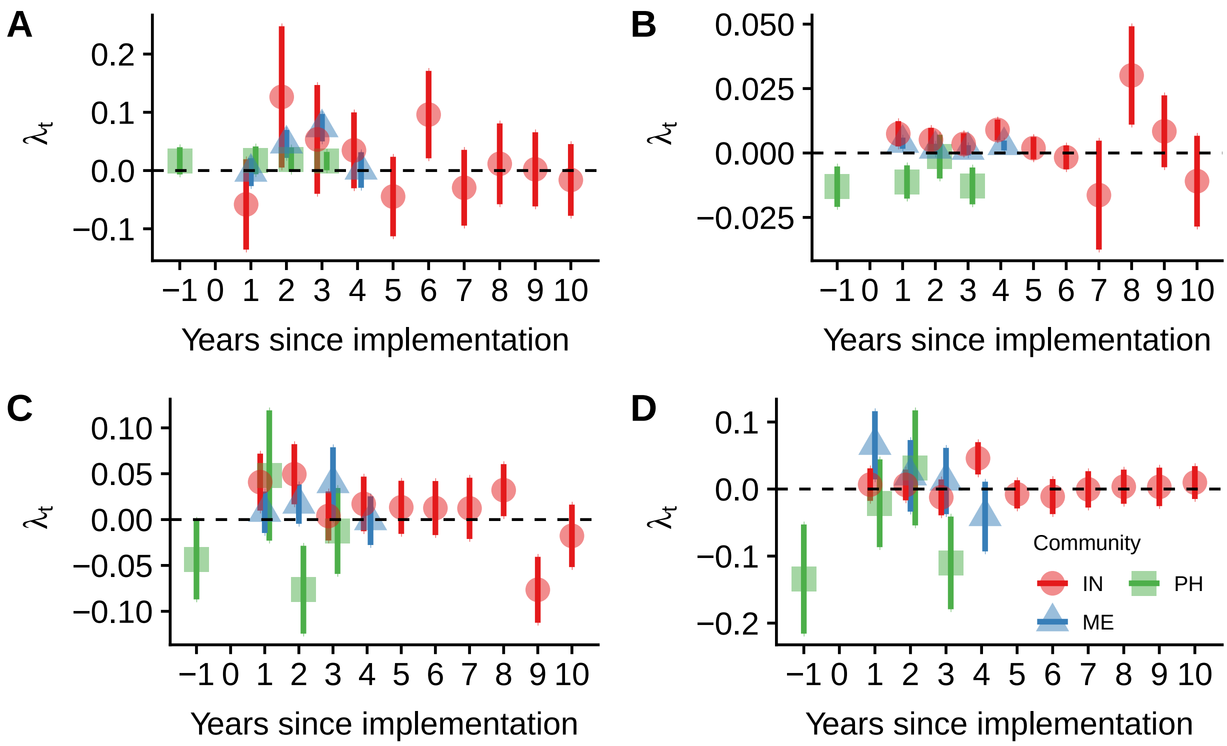


**Figure 2.** Effect sizes for marine reserves from Isla Natividad (IN; red cirlcles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey 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.

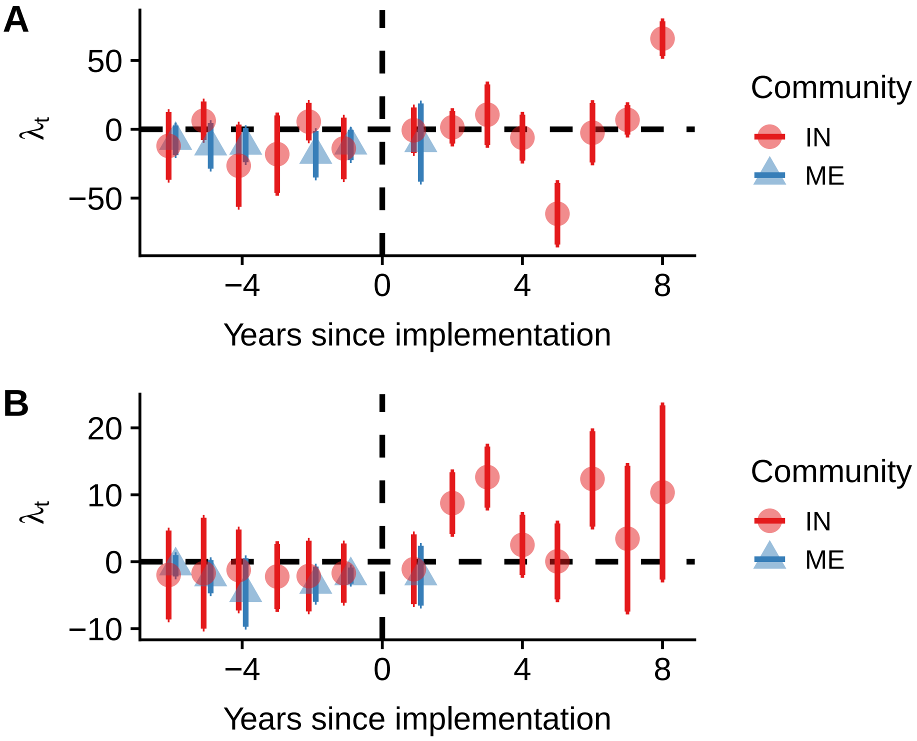


**Figure 3.** Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points indicate the effect size and standard errors. Years have been centered to year of implementation.

**Table 1.** List of indicators used to evaluate the effectiveness of marine reserves, grouped by category.

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

**Table 2.** Variables for the Social-Ecological System analysis (IN = Isla Natividad, ME = Maria Elena, PH = Punta Herrero). Alphanumeric codes follow Basurto et al. (2013); an asterisk (*) denotes variables incorporated based on Di Franco et al. (2016) and Edgar et al. (2014).

| Variable | Narrative |
|---|---|
| **Resource System (RS)** | |
| RS2 - Clarity of system boundaries: Clarity of geographical boundaries of TURF and reserves | Individual TURF and reserve boundaries are explicitly outlined in official documents that include maps and coordinates. Reserve placement is decided by the community. Fishers use GPS units and landmarks. |
| RS3 - Size of resource system: TURF Area ($Km^2$) | IN = 889.5; ME = 353.1; PH = 299.7 |
| RS3 - Size of resource system: Reserve area (Evaluated reserve area; $Km^2$) | IN = 2 (1.3); ME = 10.48(0.09); PH = 11.25 (4.37) |
| RS4.1 - Stock status: Status of the main fishery | Lobster stocks are well managed, and are (IN) or have been (ME, PH) MSC certified. |
| *RS5 - Age of reserves: Years since reserves were implemented | IN = 12; ME = 6; PH = 5 |
| **Resource Unit (RU)** | |
| RU5 - Number of units (catch diversity): Number of targeted species | Lobster is their main fishery of these three communities, but they also target finfish. Additionally, fishers from Isla Natividad target other sedentary benthic invertebrates. |
| **Actors (A)** | |
| A1 - Number of relevant actors: Number of fishers | IN = 98; ME = 80; PH = 21 |
| *A3 - Isolation: Level of isolation of the fishing grounds | Their fishing grounds and reserves are highly isolated and away from dense urban centers. |
| **Governance system (G)** | |
| GS6.1.4.3 - Territorial use communal rights : Presence of institutions that grant exclusive harvesting rights | Each community has exclusive access to harvest benthic resources, including lobster. These take the form of Territorial User Rights for Fisheries granted by the government to fishing cooperatives. |
| GS6.2 - Operational rules: Rules implemented by individuals atuhorized to partake on collective activities | Fishers have rules in addition to what the legislation mandates. These include larger minimum catch sizes, lower quotas, and assigning fishers to specific fishing grounds within their TURF. |
| GS9.1 - Social monitoring: Monitoring of the activities performed by cooperative members and external fishers | Fishing cooperatives have a group that monitors and enforces formal and internal rules. They ensure fishers of their fishing cooperative adhere to the established rules, and that foreign vessels do not poach their TURF and reserves. |
| GS9.2 - Biophysical monitoring: Monitoring of biological resources, including targeted species | Fishers perform annual standardized underwater surveys in the reserves and fishing grounds. Recently, they have installed oceanographic sensors to monitor oceanographic variables. |
| GS10.1 - Graduated sanctions | Fishers have penalties for breaking collective-choice rules or fishing inside the reserves. These may range from scoldings and warnings to not being allowed to harvest a particular resource or being expelled from the cooperative. |