Effectiveness of community-based TURF-reserves in Mexican small-scale fisheries

Juan Carlos Villaseñor-Derbez[1,*], Eréndira Aceves-Bueno[1,2], Stuart Fulton[3], Álvin Suarez[3], Arturo Hernández-Velasco[3], Jorge Torre[3], Fiorenza Micheli[4]

[1] *Bren School of Environmental Science and Management, University of California, Santa Barbara, Santa Barbara, CA, USA*
[2] *Nicholas School of the Environment, Duke University, Beaufort, NC, USA*
[3] *Comunidad y Biodiversidad A.C., Guaymas, Sonora, Mexico*
[4] *Hopkins Marine Station and Center for Ocean Solutions, Stanford University, Pacific Grove, CA, USA*

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

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 use 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 13 years, 18 old and new community-based Mexican TURF-reserves gained legal recognition thanks to a regulation passed in 2012; 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 reserves have not yet achieved their stated goal of increasing the density of lobster and other benthic invertebrates. 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 catches. 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 (Pauly et al., 2005; Worm et al., 2006; Halpern et al., 2008). 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 approaches taken 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, thus creating TURF-reserve systems (Afflerbach et al., 2014; Gelcich and Donlan, 2015; Lester et al., 2017).

TURFs are a fisheries management tool in which a well-defined group of fishers (*e.g.* fishing cooperatives) 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; Costello and Kaffine, 2010; McCay et al., 2014). 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 to aid in the recovery of marine stocks. These instruments can be combined by establishing a marine reserve within a TURF, thus making them TURF-reserves (Afflerbach et al., 2014; Gelcich and Donlan, 2015; Lester et al., 2017).

Conservation science has shown how marine reserves may lead to increased biomass, species richness, and abundance within the protected regions (Lester et al., 2009), and that these may have a series of additional benefits such as mitigation and adaptation to climate change effects, protection from environmental variability, and fisheries benefits (Roberts et al., 2017; Micheli et al., 2012; Krueck et al., 2017). Likewise, research on TURFs has shown that these areas have higher abundance of targeted species than sites operating under open access and even similar to that of marine reserves (Gelcich et al., 2008, 2012). The benefits resulting from reserves established within TURFs (*i.e.* TURF-reserves) should be captured exclusively by the group of fishers with exclusive access (Gelcich and Donlan, 2015). Although in theory these systems are expected to be successful (Smallhorn-West et al., 2018), there is little empirical evidence of their effectiveness and the drivers of their success. Moreover, TURF-reserve systems are inherently intricate social-ecological systems, and their effectiveness must depend on how environmental and social factors combine and interact (Ostrom, 2009; Gelcich and Donlan, 2015). It is therefore important to consider not only the indicators of interest, but also the governance settings under which the reserves operate.

Recent changes in fisheries regulation in Mexico provide a ripe opportunity to study the effectiveness of community-based TURF-reserves in small-scale fisheries. In Mexico, a legal framework created in 2012 allows fishers to request legal recognition of community-based reserves as “Fish Refuges” (*Zona de Refugio Pesquero*, described in more detail below; NOM-049-SAG/PESC (2014)). Since 2012, 45 old and new marine reserves have gained legal recognition as Fish Refuges. Of these, 18 were originally implemented within TURFs. 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. 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 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 TURF-reserves in small-scale fisheries where this tool is used to manage and rebuild coastal fisheries.

2 METHODS

2.1 TURF-reserves in Mexico

Community-based marine reserves that are implemented within TURFs are a form of TURF-reserve, voluntarily established and enforced by local communities. This bottom-up approach can increase compliance and self-enforcement, and reserves can yield benefits similar to systematically-designed reserves (Beger et al., 2004; Smallhorn-West et al., 2018). Community-based spatial closures occur in different contexts, like the *kapu* or *ra'ui* areas in the Pacific Islands (Johannes, 2002; Bohnsack et al., 2004). However, community-based reserves can be hard to enforce if they are not legally recognized. In such conditions, TURF fishers must rely on the exclusive access of the TURF to maintain high levels of compliance.

In an effort to bridge this normative gap, Mexican Civil Society Organizations (CSOs) served as a link between fishers and government, and helped create a legal framework that solves this governance issue: Fish Refuges (NOM-049-SAG/PESC, 2014). Fish Refuges can be implemented as permanent, temporary or partial reserves, which can protect one, some, or all resources within their boundaries. One of the ways in which fishing communities have taken advantage of this new tool is by implementing temporary marine reserves within their TURFs with a defined expiration date (often five years). When the expiration date is reached, fishers can chose to open the reserves to fishing or re-establish them. Our work focuses on Fish Refuges implemented as community-based TURF-reserves in small-scale fisheries.

The most common setup of community-based TURF-reserves in Mexico is the following. Fishers from a given community are assembled 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 exclusively fished by one cooperative, and each community usually hosts no more than one cooperative. The profits from each TURF are shared amongst all fishers from the 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 as Fish Refuges after conducting a series of studies outlined in the regulation, namely the creation of a baseline of ecological data and historical performance of the fishery (NOM-049-SAG/PESC, 2014).

2.2 Study areas

We evaluate three community-based no-take TURF-reserve systems 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, 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. The objective of these reserves was "to protect and recover stocks of commercially important invertebrate species"; mainly lobster and abalone. The reserves obtained legal recognition in 2018 (DOF, 2018b).

The other two TURF-reserve systems 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 that belongs to the *Cozumel* fishing cooperative. Punta Herrero is home to the *José María Azcorra* fishing cooperative, and similar to Isla Natividad hosts a small 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 original implementation (DOF, 2012b, 2013) and subsequent re-establishments (DOF, 2017).

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 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 community-based TURF-reserves in Mexico and elsewhere.

The regulation governing the implementation of Fish Refuges states that these are fishery management tools intended to have conservation and fisheries 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 help provide context on the social-ecological system in which reserves develop. The following two sections describe our data collection methods and analyses.

2.3 Data collection

We use three main sources of information to evaluate these reserves across ecological, socioeconomic, and governance dimensions. Ecological data come from the annual ecological monitoring of reserve and control sites. Reserve sites are areas where no fishing occurs. Control sites are areas that meet the following criteria: i) habitat characteristics are similar to the corresponding reserves, ii) presumably had a similar probability of being selected as reserves during the design phase, iii) are located within the TURF, where fishing occurs, and iv) are not directly adjacent to the reserves. 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) and causally attribute the changes to the reserve (Francini-Filho and Moura, 2008; Villaseñor-Derbez et al., 2018).

The biological data are collected by members from each community and personnel from the Mexican CSO *Comunidad y Biodiversidad* (COBI). Trained divers record species richness and abundances of fish and invertebrate species along replicate transects (30×2 m each) at depths 5-20 m in the reserves and control sites, where a minimum of 4 transects per site are performed (Suman et al., 2010; Fulton et al., 2018, 2019). Size structures are also collected during fish surveys. All sites were surveyed annually, and at least once before implementation of the reserves. A summary of sampling effort is shown in the supplementary materials (Tables S1-S2).

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 TURF-managed cooperatives with and without marine reserves. In this case our treated unit are the cooperatives (*i.e.* communities) that have implemented a reserve within their TURF, and the controls are nearby communities that have a TURF but did not implement a reserve. These communities have TURFs adjacent to the TURFs of our focal communities. Cooperatives incorporated in this analysis have similar number of members, belong to larger regional-level Cooperative Federations, and are exposed to the same markets and institutional frameworks, making them plausible controls (McCay et al., 2014; 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). A table with summary statistics for this data is provided in the supplementary materials (Table S3, Figure S5).

Data for the evaluation of the SES were collected at the community-level from official documents used in the design, creation, and designation of the marine reserves. These include the technical studies that the cooperatives submit when they request recognition of their reserves, as well as the official enactments (DOF, 2012b, 2013, 2018b). We also complimented information based on the authors' experience and knowledge of the communities. We collected information on the Resource Systems, Resource Units, Actors, and Governance System (Table 2).

2.4 Data analysis

We evaluate the effect that the TURF-reserves have had on four ecological and two socioeconomic indicators shown in Table 1. Recall that reserves were implemented to protect lobster and other benthic invertebrates. However, we also use the available fish and invertebrate data to test for associated co-benefits.

We use a difference-in-differences analysis to evaluate these indicators. This approach is widely used in econometric literature to estimate the average treatment effect of an intervention, like the impact of minimum wage increases on employment rates (Card and Krueger, 1994). In our case it allows us to estimate the effect that the reserve had on each biological and socioeconomic indicator (Table 1) by comparing trends across time and treatments since reserve implementation (Moland et al., 2013; Villaseñor-Derbez et al., 2018). To perform difference-in-differences, we regress the indicator of interest on a dummy variable for treatment, a dummy variable for years, and the interaction term between these with a multiple linear regression of the form:

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

Where year-level fixed effects capturing a temporal trend are represented by $\gamma_t Year_t$, and $\beta Zone_i$ captures the difference between reserve ($Zone = 1$) and control ($Zone = 0$) sites. 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). Therefore, we would expect this term to be positive if the indicator increases because of the reserve. Finally, $\epsilon_{i,t}$ 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 (*i.e.* Isla Natividad and Maria Elena) has three counterfactual communities.

$$I_{i,t} = \alpha + \gamma_t Year_t + \beta Treated_i + \lambda_t Year_t \times Treated_i + \epsilon_{i,t} \quad (2)$$

The coefficient interpretations remains as for Eq. 1, but in this case the $Treated$ dummy variable indicates if the community has a reserve ($Treated = 1$) or not ($Treated = 0$). 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 site-specific dynamics (De Palma et al., 2018). For each indicator in each community, we fit one model (*e.g.* there are three models for lobster density, one for each community) for a total of 12 biological model fits and four socioeconomic model fits. Model coefficients were estimated via ordinary least-squares and used heteroskedastic-robust standard errors (Zeileis, 2004). All analyses were performed in R version 3.5.2 and R Studio version 1.1.456 (R Core Team, 2018). All data and code needed to reproduce our analyses are available in a GitHub repository at: `https://github.com/jcvdav/ReserveEffect`.

We use the SES framework to evaluate each community and create a narrative that provides context for 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. Due to the lack of sufficient information to quantitatively operationalize the social-ecological systems framework for these case studies (as in Leslie et al. (2015)), we followed a similar approach to Basurto et al. (2013), who used the SES framework as a classification system of the available information to qualitatively analyze fisheries systems. 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, and identified the key variables relevant to fishing cooperatives in Mexico. 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, along with definitions and values.

3 RESULTS

The following sections present the effect that marine reserves had on the biological and socioeconomic indicators for each coastal community. Results are presented in terms of difference through time and across sites, relative to the control site on the year of implementation (*i.e.* the difference-in-differences estimate or effect size λ_t from Eqs. 1 and 2). 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 m^{-2} for Isla Natividad and Punta Herrero, and 0.01 organisms m^{-2} for Maria Elena, but are not significantly different from zero ($p > 0.05$). Likewise, no significant changes were detected in fish biomass or invertebrate and fish densities (Fig. 2B-D), where effect sizes oscillated around zero without clear trends. Figures and tables with time series of indicators and model coefficients are presented in the supplementary materials (Figures S1-S4, Tables S4-S6).

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 corresponds to the 2011 hypoxia events (Micheli et al., 2012). The only positive change observed in lobster landings is for Isla Natividad in 2014 ($p < 0.1$). The year of post-implementation data for Maria Elena does 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 only significant for the third year ($p < 0.05$). Full tables with model coefficients are presented in the supplementary materials (Tables S4-S5).

3.3 Governance

Our qualitative implementation of the social-ecological systems framework allowed us to systematically identify important differences between the case studies' governance systems and incorporate other characteristics of these fisheries neglected during the process of data collection (Table 2). We find that the 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); the largest reserve is only 4.37 km^2, and the smallest one is 0.09 km^2. Reserves are also relatively young (RS5). Additionally, fishers harvest healthy stocks (RS4.1), and it is unlikely that marine reserves will result in increased catches.

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. Lastly, the communities exhibit all the social enabling conditions for effective reserve and resource management. Here we discuss possible shortcomings in our analyses as well as possible explanations for the observed patterns.

While many ecology studies have used BACI sampling designs and respective analyses (*e.g.* Stewart-Oaten et al. (1986)), few conservation studies have done so to evaluate the effect of an intervention (*e.g.*

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, and captures previously described patterns. For example, the rapid increase observed for lobster densities in Isla Natividad on the sixth year (*i.e.* 2012; Fig. 2A), occurs 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. The use of causal inference techniques may help us support evidence-based conservation.

Our analyses of socioeconomic indicators has three limitations. First, we only look at landings and revenues by landings for communities with and without TURF-reserves. There are a number of other possible indicators that could show a change due to the implementation of the reserve. Notably, one often cited in the literature is additional benefits, such as tourism (Viana et al., 2017). However, it is unlikely that the evaluated communities will experience tourism benefits due to their remoteness and the lack of proper infrastructure to sustain tourism. A second limitation of our socioeconomic analysis is that we do not observe effort data, which may mask the effect of the reserve. For example, if catches remain relatively unchanged but fishing effort decreased, that would imply a larger catch per unit effort and thus higher profitability, provided that cost per unit effort does not increase. Likewise, it is possible that fishing effort increased around reserves to maintain the historical levels of landings. A final limitation applies to Maria Elena, where we only observe landings and income for one year after reserve implementation. While one would not expect to observe increased landings or income in such a short period, a spatial closure might cause total catches to decline, especially if effort is held constant.

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; RS5 in Table 2) 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), however, 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. As shown through the SES analysis, the size of the marine reserves appears small compared to the movement capacity of the main targeted species (RU1, RS3; Table 2). 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 reductions 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 and 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; reserves implemented in TURFs where fishing pressure is already optimally managed will still show a trade-off between fisheries and conservation objectives (Lester et al., 2017). 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 greater.

Finally, extreme conditions, including prolonged hypoxia, heat waves, and storms have affected both the Pacific and Caribbean regions, with large negative impacts on 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., 2018). Effects of protection might be eliminated by the mortalities associated with these extreme conditions.

While the evaluated reserves have failed to provide fishery benefits to date, there are a number of additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range of species and vulnerable habitat. Previous research focusing on these specific sites has shown that they serve as an insurance mechanism 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., 0). Self-regulation of fishing effort can serve as a way to compensate for future declines associated to environmental variation (Finkbeiner et al., 2018). Furthermore, embarking on 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 has allowed fishers to have greater bargaining power and leverage over fisheries management (Pérez-Ramírez et al., 2012). These additional benefits might explain why communities show a positive perception about their performance and continue to support their presence by re-establishing the reserves (Ayer et al., 2018).

Community-based TURF-reserves in small-scale fisheries may be helpful conservation and fishery management tools when appropriately implemented (Gelcich and Donlan, 2015). We must promote bottom-up design and implementation processes like the ones in the evaluated reserves, but without setting design principles aside. 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. Furthermore, conservation and advocacy groups should consider the opportunity costs of such interventions (*sensu* Smith et al. (2010)) and evaluate the potential of other approaches that may yield similar benefits.

In terms of fisheries regulation in Mexico, our work only evaluates Fish Refuges established within TURFs. Future research should aim at evaluating other Fish Refuges established as bottom-up processes but without the presence of TURFs (*e.g.* DOF (2012a)), others established through top-down processes (*i.e.* DOF (2018a)), as well as the relationship between governance and effectiveness across this gradient of approaches. For the particular case of the reserves that we evaluate, the possibility of expanding reserves or merging existing polygons into larger areas should be evaluated and proposed to the communities.

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.

FUNDING

JCVD received funding from UCMexus - CONACyT Doctoral Fellowship (CVU 669403) and the Latin American Fisheries Fellowship Program. AS, AHV, SF and JT received funding from Marisla Foundation, Packard Foundation, Walton Family Foundation, Summit Foundation, and Oak Foundation. FM was supported by NSF-CNH and NSF BioOce (grants DEB-1212124 and 1736830).

ACKNOWLEDGMENTS

The authors wish to acknowledge 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 fishing communities here mentioned, who participated in the data-collection process. The authors wish to acknowledge comments by the reviewers and editor, which significantly improved the quality of this work.

REFERENCES

Aalto, E. A., Micheli, F., Boch, C. A., Espinoza Montes, J. A., Woodson, C. B., and De Leo, G. A. (0). Catastrophic mortality, allee effects, and marine protected areas. *The American Naturalist* 0, 000–000. doi:10.1086/701781

Aceves-Bueno, E., Cornejo-Donoso, J., Miller, S. J., and Gaines, S. D. (2017). Are territorial use rights in fisheries (TURFs) sufficiently large? *Marine Policy* 78, 189–195. doi:10.1016/j.marpol.2017.01.024

Afflerbach, J. C., Lester, S. E., Dougherty, D. T., and Poon, S. E. (2014). A global survey of turf-reserves, territorial use rights for fisheries coupled with marine reserves. *Global Ecology and Conservation* 2, 97–106. doi:10.1016/j.gecco.2014.08.001

Ayer, A., Fulton, S., Caamal-Madrigal, J. A., and Espinoza-Tenorio, A. (2018). Halfway to sustainability: Management lessons from community-based, marine no-take zones in the mexican caribbean. *Marine Policy* 93, 22–30. doi:10.1016/j.marpol.2018.03.008

Babcock, R. C., Shears, N. T., Alcala, A. C., Barrett, N. S., Edgar, G. J., Lafferty, K. D., et al. (2010). Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proc Natl Acad Sci USA* 107, 18256–18261. doi:10.1073/pnas.0908012107

Basurto, X., Gelcich, S., and Ostrom, E. (2013). The social–ecological system framework as a knowledge classificatory system for benthic small-scale fisheries. *Global Environmental Change* 23, 1366–1380. doi:10.1016/j.gloenvcha.2013.08.001

Beger, M., Harborne, A. R., Dacles, T. P., Solandt, J.-L., and Ledesma, G. L. (2004). A framework of lessons learned from community-based marine reserves and its effectiveness in guiding a new coastal management initiative in the philippines. *Environ Manage* 34, 786–801. doi:10.1007/s00267-004-0149-z

Bohnsack, J. A., Ault, J. S., and Causey, B. (2004). Why have no-take marine protected areas? In *American Fisheries Society Symposium*. vol. 42, 185–193

Breitburg, D., Levin, L. A., Oschlies, A., Grégoire, M., Chavez, F. P., Conley, D. J., et al. (2018). Declining oxygen in the global ocean and coastal waters. *Science*

Card, D. and Krueger, A. B. (1994). Minimum wages and employment: A case study of theFast-food industry in new jersey and pennsylvania. *AER* 84, 772–793

Cavole, L. M., Demko, A. M., Diner, R. E., Giddings, A., Koester, I., Pagniello, C. M., et al. (2016). Biological impacts of the 2013–2015 warm-water anomaly in the northeast pacific: Winners, losers, and the future. *Oceanography* 29, 273–285

Costello, C. and Kaffine, D. T. (2010). Marine protected areas in spatial property-rights fisheries. *Australian Journal of Agricultural and Resource Economics* 54, 321–341. doi:10.1111/j.1467-8489.2010.00495.x

Costello, C., Ovando, D., Hilborn, R., Gaines, S. D., Deschenes, O., and Lester, S. E. (2012). Status and solutions for the world's unassessed fisheries. *Science* 338, 517–520. doi:10.1126/science.1223389

da Silva, I. M., Hill, N., Shimadzu, H., Soares, A. M. V. M., and Dornelas, M. (2015). Spillover effects of a community-managed marine reserve. *PLoS ONE* 10, e0111774. doi:10.1371/journal.pone.0111774

De Leo, G. A. and Micheli, F. (2015). The good, the bad and the ugly of marine reserves for fishery yields. *Philos Trans R Soc Lond, B, Biol Sci* 370. doi:10.1098/rstb.2014.0276

De Palma, A., Sanchez Ortiz, K., Martin, P. A., Chadwick, A., Gilbert, G., Bates, A. E., et al. (2018). Challenges with inferring how land-use affects terrestrial biodiversity: Study design, time, space and synthesis. *Advances in ecological research* doi:10.1016/bs.aecr.2017.12.004

Di Franco, A., Thiriet, P., Di Carlo, G., Dimitriadis, C., Francour, P., Gutiérrez, N. L., et al. (2016). Five key attributes can increase marine protected areas performance for small-scale fisheries management. *Sci Rep* 6, 38135. doi:10.1038/srep38135

DOF (1993). Norma oficial mexicana 006-pesc-1993, para regular el aprovechamiento de todas las especies de langosta en las aguas de jurisdiccion federal del golfo de mexico y mar caribe, asi como del oceano pacifico incluyendo el golfo de california. *Diario Oficial de la Federación*

DOF (2012a). Acuerdo por el que se establece una red de zonas de refugio en aguas marinas de jurisdicción federal frente a la costa oriental del estado de baja california sur, en el corredor marino de san cosme a punta coyote. *Diario Oficial de la Federación*

DOF (2012b). Acuerdo por el que se establece una red de zonas de refugio pesquero en aguas marinas de jurisdicción federal ubicadas en el área de sian ka an, dentro de la bahía espíritu santo en el estado de quintana roo. *Diario Oficial de la Federación*

DOF (2013). Acuerdo por el que se establece una red de zonas de refugio pesquero en aguas marinas de jurisdicción federal ubicadas en las áreas de banco chinchorro y punta herrero en el estado de quintana roo. *Diario Oficial de la Federación*

DOF (2017). Acuerdo por el que se amplía la vigencia del similar que establece una red de zonas de refugio pesquero en aguas marinas de jurisdicción federal ubicadas en el área de sian ka an, dentro de la bahía espíritu santo en el estado de quintana roo, publicado el 30 de noviembre de 2012. *Diario Oficial de la Federación*

DOF (2018a). Acuerdo por el que se establece el área de refugio para la tortuga amarilla (caretta caretta) en el golfo de ulloa, en baja california sur. *Diario Oficial de la Federación* doi:http://www.dof.gob.mx/nota_detalle.php?codigo=5525056&fecha=05/06/2018

DOF (2018b). Acuerdo por el que se establece una red de dos zonas de refugio pesquero parciales permanentes en aguas marinas de jurisdicción federal adyacentes a isla natividad, ubicada en el municipio de mulegé, en el estado de baja california sur. *Diario Oficial de la Federación*

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., et al. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature* 506, 216–220. doi:10.1038/nature13022

Ferraro, P. J. and Pattanayak, S. K. (2006). Money for nothing? a call for empirical evaluation of biodiversity conservation investments. *PLoS Biol* 4, e105. doi:10.1371/journal.pbio.0040105

Finkbeiner, E., Micheli, F., Saenz-Arroyo, A., Vazquez-Vera, L., Perafan, C., and Cárdenas, J. (2018). Local response to global uncertainty: Insights from experimental economics in small-scale fisheries. *Global Environmental Change* 48, 151–157. doi:10.1016/j.gloenvcha.2017.11.010

Francini-Filho, R. and Moura, R. (2008). Evidence for spillover of reef fishes from a no-take marine reserve: An evaluation using the before-after control-impact (BACI) approach. *Fisheries Research* 93, 346–356. doi:10.1016/j.fishres.2008.06.011

Fulton, S., Caamal-Madrigal, J., Aguilar-Perera, A., Bourillón, L., and Heyman, W. D. (2018). Marine conservation outcomes are more likely when fishers participate as citizen scientists: Case studies from the mexican mesoamerican reef. *CSTP* 3. doi:10.5334/cstp.118

Fulton, S., Hernandez-Velasco, A., Suarez-Castillo, A., Fernandez-Rivera Melo, F., Rojo, M., Saenz-Arroyo, A., et al. (2019). From fishing fish to fishing data: the role of artisanal fishers in conservation and resource management in mexico. In *Viability and Sustainability of Small-Scale Fisheries in Latin America and The Caribbean*, eds. S. Salas, M. J. Barragán-Paladines, and R. Chuenpagdee (Cham: Springer International Publishing), vol. 19 of *MARE Publication Series*. 151–175. doi:10.1007/978-3-319-76078-0_7

Gelcich, S. and Donlan, C. J. (2015). Incentivizing biodiversity conservation in artisanal fishing communities through territorial user rights and business model innovation. *Conserv Biol* 29, 1076–1085. doi:10.1111/cobi.12477

Gelcich, S., Fernández, M., Godoy, N., Canepa, A., Prado, L., and Castilla, J. C. (2012). Territorial user rights for fisheries as ancillary instruments for marine coastal conservation in chile. *Conserv Biol* 26, 1005–1015. doi:10.1111/j.1523-1739.2012.01928.x

Gelcich, S., Godoy, N., Prado, L., and Castilla, J. C. (2008). Add-on conservation benefits of marine territorial user rights fishery policies in central chile. *Ecol Appl* 18, 273–281. doi:10.1890/06-1896.1

Green, A., Chollett, I., Suarez, A., Dahlgren, C., Cruz, S., Zepeda, C., et al. (2017). *Biophysical Principles for Designing a Network of Replenishment Zones for the Mesoamerican Reef System*. Technical report

Guidetti, P. (2002). The importance of experimental design in detecting the effects of protection measures on fish in mediterranean MPAs. *Aquatic Conserv: Mar. Freshw. Ecosyst.* 12, 619–634. doi:10.1002/aqc.514

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., et al. (2008). A global map of human impact on marine ecosystems. *Science* 319, 948–952. doi:10.1126/science.1149345

Hilborn, R., Micheli, F., and De Leo, G. A. (2006). Integrating marine protected areas with catch regulation. *Can. J. Fish. Aquat. Sci.* 63, 642–649. doi:10.1139/f05-243

Hilborn, R., Stokes, K., Maguire, J.-J., Smith, T., Botsford, L. W., Mangel, M., et al. (2004). When can marine reserves improve fisheries management? *Ocean and Coastal Management* 47, 197 – 205. doi:https://doi.org/10.1016/j.ocecoaman.2004.04.001

Hughes, T. P., Anderson, K. D., Connolly, S. R., Heron, S. F., Kerry, J. T., Lough, J. M., et al. (2018). Spatial and temporal patterns of mass bleaching of corals in the anthropocene. *Science*

Johannes, R. E. (2002). The renaissance of community-based marine resource management in oceania. *Annual Review of Ecology and Systematics* 33, 317–340

Jupiter, S. D., Epstein, G., Ban, N. C., Mangubhai, S., Fox, M., and Cox, M. (2017). A social–ecological systems approach to assessing conservation and fisheries outcomes in fijian locally managed marine areas. *Soc Nat Resour* 30, 1096–1111. doi:10.1080/08941920.2017.1315654

Krueck, N. C., Ahmadia, G. N., Possingham, H. P., Riginos, C., Treml, E. A., and Mumby, P. J. (2017). Marine reserve targets to sustain and rebuild unregulated fisheries. *PLoS Biol* 15, e2000537. doi:10.1371/journal.pbio.2000537

Leslie, H. M., Basurto, X., Nenadovic, M., Sievanen, L., Cavanaugh, K. C., Cota-Nieto, J. J., et al. (2015). Operationalizing the social-ecological systems framework to assess sustainability. *Proc Natl Acad Sci U S A* 112, 5979–5984. doi:10.1073/pnas.1414640112

Lester, S., Halpern, B., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B., Gaines, S., et al. (2009). Biological effects within no-take marine reserves: a global synthesis. *Mar. Ecol. Prog. Ser.* 384, 33–46. doi:10.3354/meps08029

Lester, S., McDonald, G., Clemence, M., Dougherty, D., and Szuwalski, C. (2017). Impacts of turfs and marine reserves on fisheries and conservation goals: theory, empirical evidence, and modeling. *BMS* 93, 173–198. doi:10.5343/bms.2015.1083

McCay, B. (2017). Territorial use rights in fisheries of the northern pacific coast of mexico. *BMS* 93, 69–81. doi:10.5343/bms.2015.1091

McCay, B. J., Micheli, F., Ponce-Díaz, G., Murray, G., Shester, G., Ramirez-Sanchez, S., et al. (2014). Cooperatives, concessions, and co-management on the pacific coast of mexico. *Marine Policy* 44, 49–59. doi:10.1016/j.marpol.2013.08.001

Micheli, F., Saenz-Arroyo, A., Greenley, A., Vazquez, L., Espinoza Montes, J. A., Rossetto, M., et al. (2012). Evidence that marine reserves enhance resilience to climatic impacts. *PLoS ONE* 7, e40832. doi:10.1371/journal.pone.0040832

Moland, E., Olsen, E. M., Knutsen, H., Garrigou, P., Espeland, S. H., Kleiven, A. R., et al. (2013). Lobster and cod benefit from small-scale northern marine protected areas: inference from an empirical before-after control-impact study. *Proceedings of the Royal Society B: Biological Sciences* 280, 20122679–20122679. doi:10.1098/rspb.2012.2679

NOM-049-SAG/PESC (2014). Norma oficial mexicana nom-049-sag/pesc-2014, que determina el procedimiento para establecer zonas de refugio para los recursos pesqueros en aguas de jurisdicción federal de los estados unidos mexicanos. *DOF*

[Dataset] OECD (2017). Inflation CPI

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science* 325, 419–422. doi:10.1126/science.1172133

Ovando, D., Dougherty, D., and Wilson, J. R. (2016). Market and design solutions to the short-term economic impacts of marine reserves. *Fish Fish* 17, 939–954. doi:10.1111/faf.12153

Pauly, D., Watson, R., and Alder, J. (2005). Global trends in world fisheries: impacts on marine ecosystems and food security. *Philosophical Transactions of the Royal Society B: Biological Sciences* 360, 5–12. doi:10.1098/rstb.2004.1574

Pérez-Ramírez, M., Ponce-Díaz, G., and Lluch-Cota, S. (2012). The role of msc certification in the empowerment of fishing cooperatives in mexico: The case of red rock lobster co-managed fishery. *Ocean Coast Manag* 63, 24–29. doi:10.1016/j.ocecoaman.2012.03.009

R Core Team (2018). *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria

Rees, S. E., Pittman, S. J., Foster, N., Langmead, O., Griffiths, C., Fletcher, S., et al. (2018). Bridging the divide: Social–ecological coherence in marine protected area network design. *Aquatic Conservation: Marine and Freshwater Ecosystems*

Roberts, C. M., OLeary, B. C., McCauley, D. J., Cury, P. M., Duarte, C. M., Lubchenco, J., et al. (2017). Marine reserves can mitigate and promote adaptation to climate change. *Proc Natl Acad Sci USA* 114, 6167–6175. doi:10.1073/pnas.1701262114

Rossetto, M., Micheli, F., Saenz-Arroyo, A., Montes, J. A. E., and De Leo, G. A. (2015). No-take marine reserves can enhance population persistence and support the fishery of abalone. *Can. J. Fish. Aquat. Sci.* 72, 1503–1517. doi:10.1139/cjfas-2013-0623

Smallhorn-West, P. F., Bridge, T. C. L., Malimali, S., Pressey, R. L., and Jones, G. P. (2018). Predicting impact to assess the efficacy of community-based marine reserve design. *Conserv Lett* , e12602doi:10.1111/conl.12602

Smith, M. D., Lynham, J., Sanchirico, J. N., and Wilson, J. A. (2010). Political economy of marine reserves: understanding the role of opportunity costs. *Proc Natl Acad Sci USA* 107, 18300–18305. doi:10.1073/pnas.0907365107

Stewart-Oaten, A., Murdoch, W. W., and Parker, K. R. (1986). Environmental impact assessment: ”pseudoreplication” in time? *Ecology* 67, 929–940. doi:10.2307/1939815

Suman, C. S., Saenz-Arroyo, A., Dawson, C., and Luna, M. C. (2010). *Manual de Instruccion de Reef Check California: Guia de instruccion para el monitoreo del bosque de sargazo en la Peninsula de Baja California* (Pacific Palisades, CA, USA: Reef Check Foundation)

Viana, D. F., Halpern, B. S., and Gaines, S. D. (2017). Accounting for tourism benefits in marine reserve design. *PLoS ONE* 12, e0190187. doi:10.1371/journal.pone.0190187

Villaseñor-Derbez, J. C., Faro, C., Wright, M., Martínez, J., Fitzgerald, S., Fulton, S., et al. (2018). A user-friendly tool to evaluate the effectiveness of no-take marine reserves. *PLOS ONE* 13, 1–21. doi:10.1371/journal.pone.0191821

Woodson, C. B., Micheli, F., Boch, C., Al-Najjar, M., Espinoza, A., Hernandez, A., et al. (2018). Harnessing marine microclimates for climate change adaptation and marine conservation. *Conservation Letters* , e12609doi:10.1111/conl.12609

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., et al. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science* 314, 787–790. doi:10.1126/science.1132294

Zeileis, A. (2004). Econometric computing with hc and hac covariance matrix estimators. *J Stat Softw* 11. doi:10.18637/jss.v011.i10

FIGURE CAPTIONS

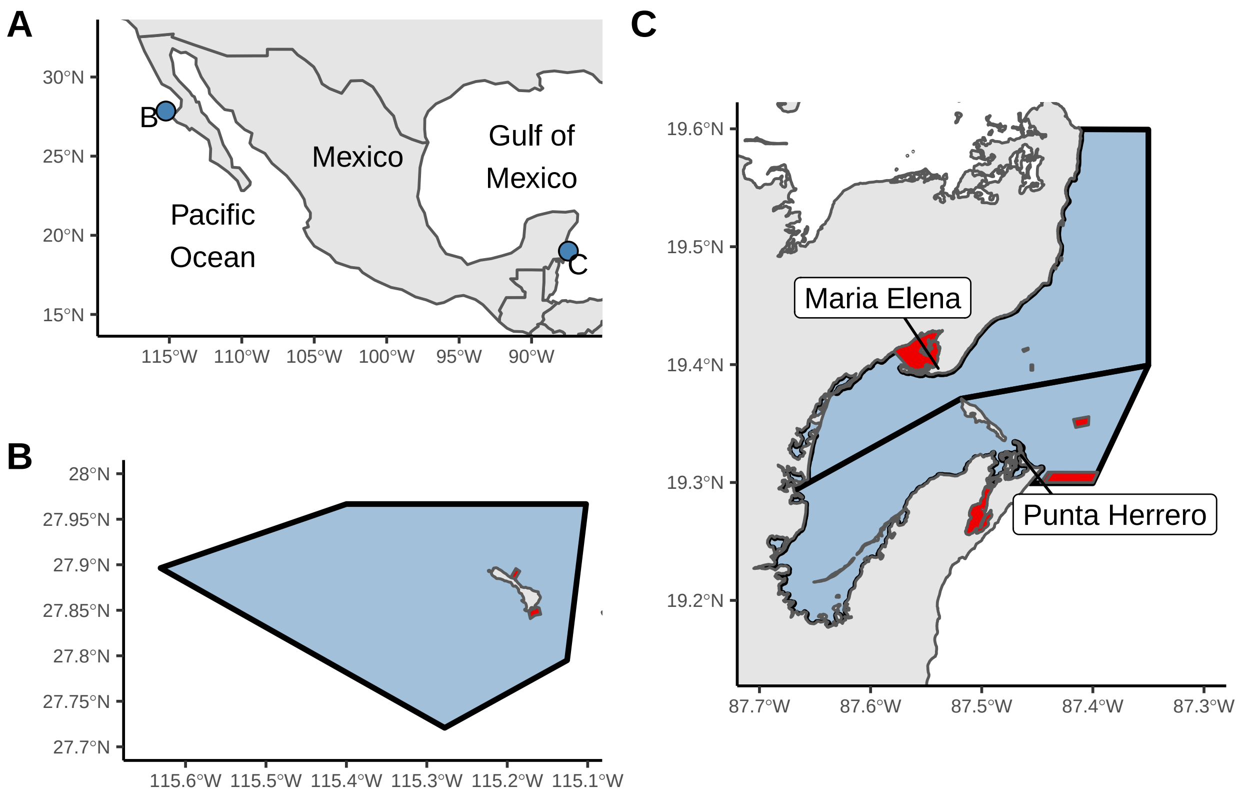

Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

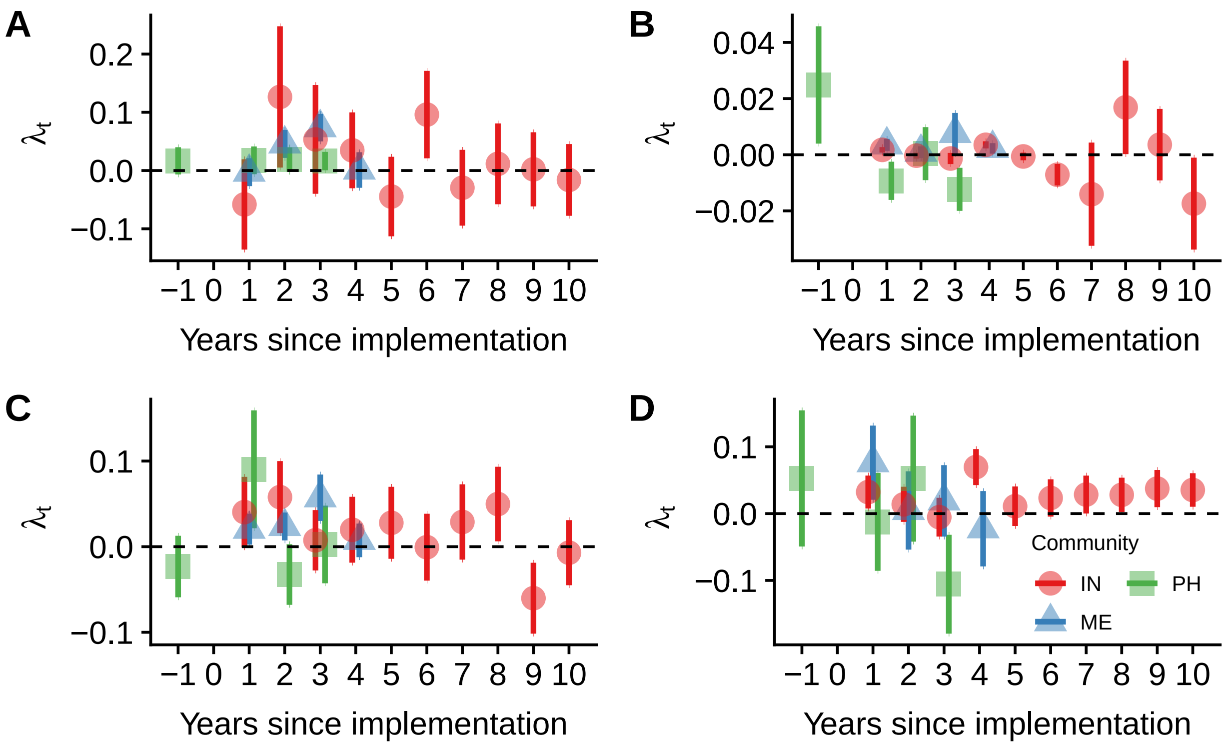

Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red circles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

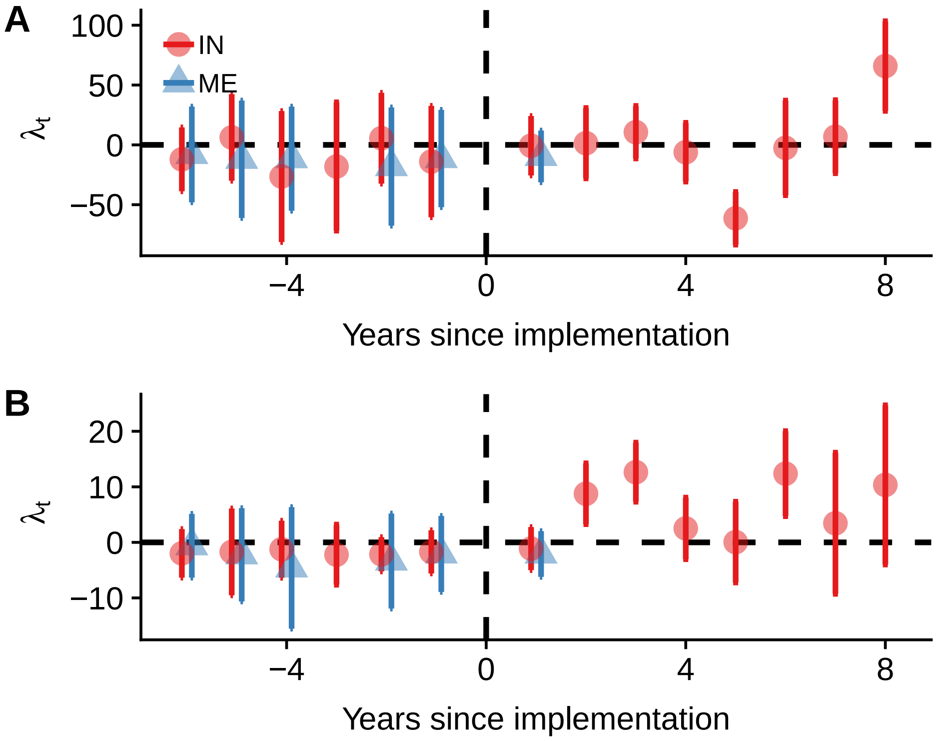

Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points are jittered hotizontally to avoid overplotting. Points indicate the effect size and error bars are heteroskedastic-robust standard errors. Years have been 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). The presented narrative applies equally for all communities unless otherwise noted.

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)	
RU1 - Resource unit mobility	Adult spiny lobsters can move between 1 and 10 Km, while larvae can have displacements in the order of hundreds of Km (Aceves-Bueno et al., 2017; Green et al., 2017).
RU5 - Number of units (catch diversity): Number of targeted species	Lobster is their main fishery of these three communities, but they also target finfish (2 spp each). Additionally, fishers from Isla Natividad target other sedentary benthic invertebrates (4 spp).
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. IN lies 545 Km south from Tijuana, and ME and PH 230 Km south from Cancun, where the nearest international airports are located.
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 are: 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 (Consejo de vigilancia) 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.