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

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ABSTRACT

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

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

1 INTRODUCTION

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

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

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

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

2 METHODS

2.1 Study area

We evaluate three TURF-reserves in Mexico (Fig 1A). The first one was created by the *Buzos y Pescadores de la Baja California* fishing cooperative, located in Isla Natividad in the Baja 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. While these reserves obtained legal recognition only in 2018, they have been well enforced since their implementation.

The other two TURF-reserves are located in Maria Elena and Punta Herrero, in the Yucatan Peninsula (Fig 1C). In contrast with Isla Nativdad, which hosts a well established fishing community, Maria Elena is a fishing camp –visited intermittently during the fishing season– belonging to the Cozumel fishing cooperative (*SCPP Cozumel*); Punta Herrero is home to the *SCPP José María Azcorra* cooperative, and similar to Isla Natividad hosts a local community. Their main fishery is the Caribbean spiny lobster (*Panulirus argus*), but they also target finfish in the off-season. Maria Elena and Punta Herrero established eight marine reserves in 2012, and four marine reserves in 2013, respectively. All these reserves have been legally recognized as Fishing Refuges since their creation (DOF, 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, 2017; McCay et al., 2014; Aceves-Bueno et al., 2017). Likewise, Maria Elena and Punta Herrero are representative of fishing cooperatives in the Mexican Caribbean, which are also part of a regional Federation. Together, these three communities provide an accurate representation of other fishing communities in each of their regions. While each region has additional communities that have established community-based TURF-reserves, available data would not allow us to perform the in-depth analysis that we undertake. Yet, given the similarities among communities and the socioeconomic and governance setting under which they opperate, it is safe to cautiously generalize our results to other communities in Mexico and other regions around the world.

2.2 Data collection

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

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

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

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

Data for the operationalization of the social-ecological system were collected at the community–level from official documents used in the creation and designation of the marine reserves (DOF, 2012b, 2013, 2018b) and based on the authors' experience and knowledge of the communities. These include information on the resource system, the resource units, actors, and the governance system itself (Table 2).

2.3 Data analysis

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:

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

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

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

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

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

We use the social-ecologycal system (SES) framework to evaluate each community as a means of providing an explanation to the biological and socioeconomic results. The use the SES framework standardizes our analysis and allows us to communicate our results in a common language across fields. We based our variable selection primarily on Leslie et al. (2015); Basurto et al. (2013), who have previously analyzed Mexican fishing cooperatives using this framework. We also incorporate other relevant information following Di Franco et al. (2016) and Edgar et al. (2014). Table 2 shows the selected variables, their definition and selected indicators.

3 RESULTS

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

3.1 Biological effects

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

3.2 Socioeconomic effects

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

3.3 Governance

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 opperate 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.

4 DISCUSSION

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

Some works evaluate marine reserves by performing inside-outside (Guidetti et al., 2014; Friedlander et al., 2017; Rodriguez and Fanning, 2017) or before-after comparisons (Betti et al., 2017). The first approach does not address temporal variability, and the second can not distinguish between the temporal trends in a reserve and the entire system (De Palma et al., 2018). Our approach to evaluate the temporal and spatial changes provides a more robust measure of reserve effectiveness. For example, we capture previously described patterns like the rapid increase observed for 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 effectivenes may be the young age of the reserves. Literature shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al., 2014; Babcock et al., 2010). Isla Natividad has the oldest reserves, and our SES analysis suggests that all communities have a well-established community-based enforcement system. With these characteristics, one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves (*i.e.* < 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; other community-based marine reserves in tropical ecosystems may take up to six years to show a spillover effect (da Silva et al., 2015).

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

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

While the evaluated reserves have failed to provide fishery benefits up to now, there are a number of additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range of species and vulnerable habitat, like coral reefs. These sites can serve as an insurance against uncertainty and errors in fisheries management, as well as environmental shocks (Hilborn et al., 2004, 2006; Micheli et al., 2012; Aalto et al., in press). Self-regulation of fishing effort (*i.e.* reduction in harvest) can serve as a way to compensate for future declines associated to environmental variation (Finkbeiner et al., 2018). Embarking in a marine conservation project can bring the community together, which promotes social cohesion and builds social capital (Fulton et al., 2019). 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 may not be generalizable to other reserves established under the same legal frameowrk (*i.e.* Fish Refges) in Mexico, and future research should aim at evaluating other areas that have also been established as bottom-up processes bt without the presence of TURFs (*e.g.* DOF (2012a)), or others established through a top-down process (*i.e.* DOF (2018a)).

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

CONFLICT OF INTEREST STATEMENT

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

AUTHOR CONTRIBUTIONS

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

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REFERENCES

Aalto, E., Micheli, F., Boch, C., Espinoza-Montes, A., Woodson, C., and De Leo, G. (in press). Marine protected areas lower risk of abalone fishery collapse following widespread catastrophic mortality events. *American Naturalist*

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

Arafeh-Dalmau, N., Torres-Moye, G., Seingier, G., Montaño-Moctezuma, G., and Micheli, F. (2017). Marine spatial planning in a transboundary context: Linking baja california with california's network of marine protected areas. *Front Mar Sci* 4. doi:10.3389/fmars.2017.00150

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

Betti, F., Bavestrello, G., Bo, M., Asnaghi, V., Chiantore, M., Bava, S., et al. (2017). Over 10 years of variation in mediterranean reef benthic communities. *Marine Ecology* 38, e12439. doi:10.1111/maec.12439

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

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

[Dataset] DOF, D. (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

DOF, D. (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, D. (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, D. (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, D. (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*

DOF, D. (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

Espinosa-Romero, M. J., Rodriguez, L. F., Weaver, A. H., Villanueva-Aznar, C., and Torre, J. (2014). The changing role of ngos in mexican small-scale fisheries: From environmental conservation to multi-scale governance. *Marine Policy* 50, 290–299. doi:10.1016/j.marpol.2014.07.005

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

Friedlander, A. M., Golbuu, Y., Ballesteros, E., Caselle, J. E., Gouezo, M., Olsudong, D., et al. (2017). Size, age, and habitat determine effectiveness of palau's marine protected areas. *PLoS ONE* 12, e0174787. doi:10.1371/journal.pone.0174787

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., 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

Giakoumi, S., Scianna, C., Plass-Johnson, J., Micheli, F., Grorud-Colvert, K., Thiriet, P., et al. (2017). Ecological effects of full and partial protection in the crowded mediterranean sea: a regional meta-analysis. *Sci Rep* 7, 8940. doi:10.1038/s41598-017-08850-w

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., Baiata, P., Ballesteros, E., Di Franco, A., Hereu, B., Macpherson, E., et al. (2014). Large-scale assessment of mediterranean marine protected areas effects on fish assemblages. *PLoS ONE* 9, e91841. doi:10.1371/journal.pone.0091841

Halpern, B. S., Klein, C. J., Brown, C. J., Beger, M., Grantham, H. S., Mangubhai, S., et al. (2013). Achieving the triple bottom line in the face of inherent trade-offs among social equity, economic return, and conservation. *Proc Natl Acad Sci USA* 110, 6229–6234. doi:10.1073/pnas.1217689110

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

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

López-Angarita, J., Moreno-Sánchez, R., Maldonado, J. H., and Sánchez, J. A. (2014). Evaluating linked social-ecological systems in marine protected areas. *Conserv Lett* 7, 241–252. doi:10.1111/conl.12063

Mascia, M. B., Fox, H. E., Glew, L., Ahmadia, G. N., Agrawal, A., Barnes, M., et al. (2017). A novel framework for analyzing conservation impacts: evaluation, theory, and marine protected areas. *Ann N Y Acad Sci* 1399, 93–115. doi:10.1111/nyas.13428

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., Castrejón, M., Gutiérrez, N. L., and Defeo, O. (2016). The marine stewardship council certification in latin america and the caribbean: A review of experiences, potentials and pitfalls. *Fisheries Research* 182, 50–58. doi:10.1016/j.fishres.2015.11.007

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

Rodriguez, A. G. and Fanning, L. M. (2017). Assessing marine protected areas effectiveness: A case study with the tobago cays marine park. *OJMS* 07, 379–408. doi:10.4236/ojms.2017.73027

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

Sala, E. and Giakoumi, S. (2017). No-take marine reserves are the most effective protected areas in the ocean. *ICES Journal of Marine Science* doi:10.1093/icesjms/fsx059

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

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., Micheli, F., Boch, C., M, A.-N., Hernandez, A., Vera, L., et al. (in press). Harnessing environmental variability as a climate change adaptation for small-scale fisheries. *Conservation Letters*

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

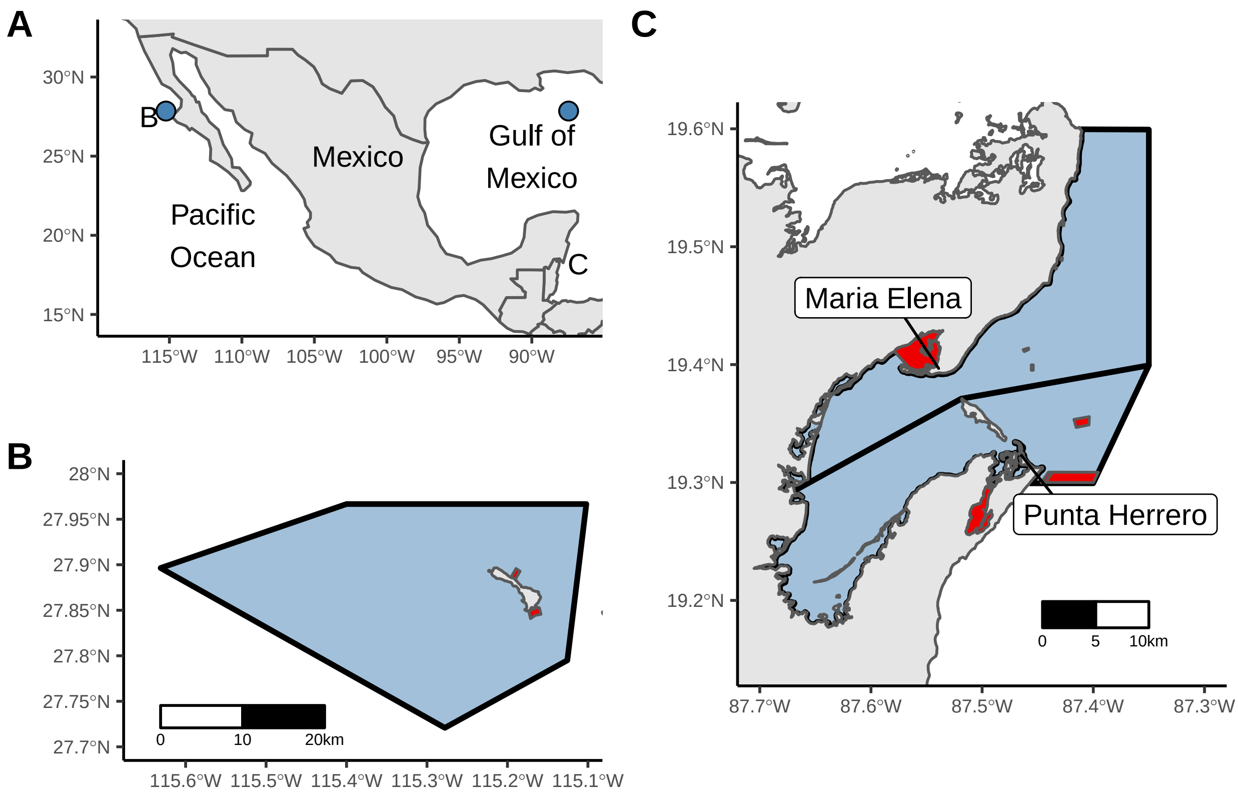

Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

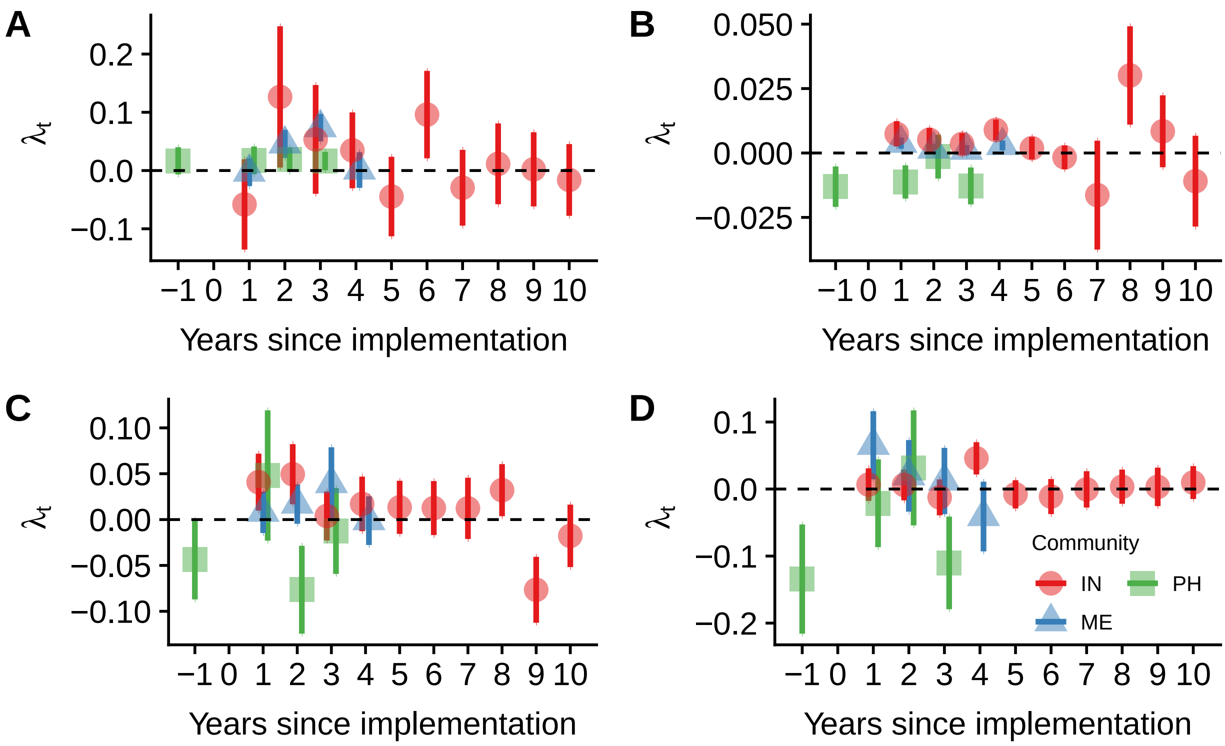

Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red cirlcles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

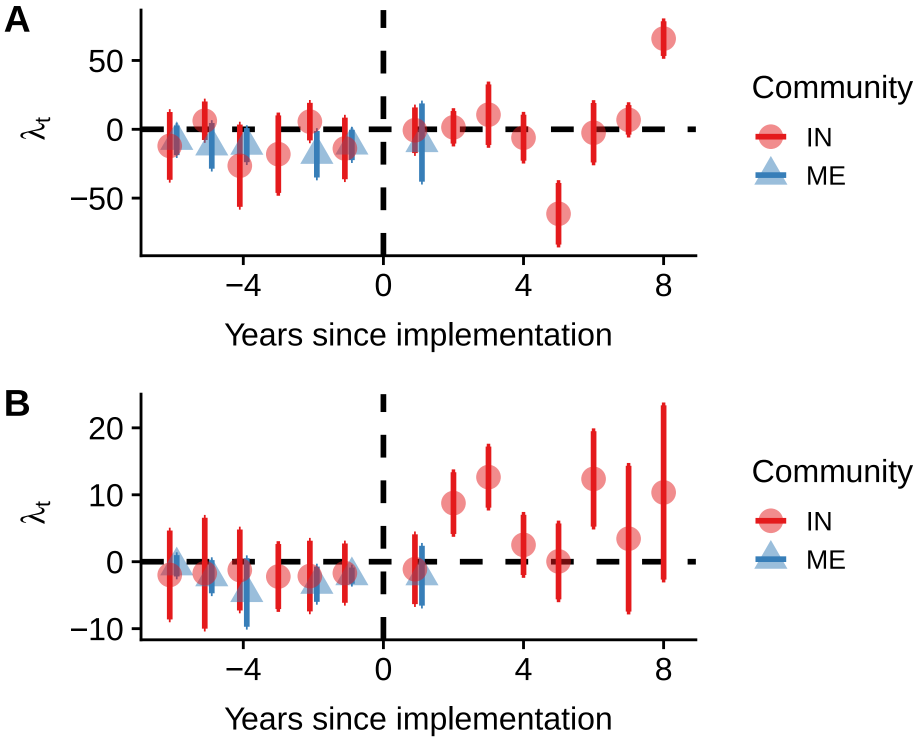

Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points indicate the effect size and standard errors. Years have been centered to year of implementation.

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

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

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) orhave 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 mandate. 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 internalrules. 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.