

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

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2 ABSTRACT

3 Coastal marine ecosystems provide livelihoods for small-scale fishers and coastal communities
4 around the world. Artisanal fisheries face great challenges since they are difficult to monitor,
5 enforce, and manage. Combining territorial user rights for fisheries (TURF) with no-take marine
6 reserves to create TURF-reserves is believed to improve the performance of small-scale fisheries
7 by buffering fisheries from environmental variability and management errors, while ensuring
8 that fishers reap the benefits of conservation investments. In the last six years, and following a
9 2012 regulation, 18 TURF-reserves have been implemented in Mexico; their effectiveness has
10 not been formally evaluated. We combine causal inference techniques and a social-ecological
11 systems framework to provide a holistic evaluation of community-based TURF reserves in three
12 coastal communities in Mexico. We find that while reserves have not yet achieved their stated
13 goal of increasing lobster densities, they continue to receive significant support from the fishing
14 communities. A lack of ecological and socioeconomic effects likely results from a combination of
15 factors. First, the lobster fisheries are already well managed, and it is unlikely that reserves might
16 have a detectable effect. Second, some of the reserves are not large enough to protect lobsters'
17 home ranges. Third, some of these reserves might be too young for the effects to show. However,
18 these reserves have shaped small-scale fishers' way of thinking about marine conservation,
19 which can provide a foundation for establishing additional, larger marine reserves needed to
20 effectively conserve mobile species.

21 **Keywords:** TURF-reserves, Causal Inference, Social-Ecological Systems, Marine Protected Areas, Marine Conservation, Small-Scale
22 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 resources (Costello and Kaffine, 2010; Lester et al., 2017). Commonly known as TURF-Reserves, these systems increase the benefits of spatial access rights allowing the maintenance 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

66 opportunities where improvement or adjustment might lead to increased effectiveness. We draw from
67 lessons learned in these three case studies and provide management recommendations to maximize the
68 effectiveness of community-based marine reserves in small-scale fisheries in Mexico and in other regions
69 around the world that are using this tool to manage and rebuild their coastal fisheries.

2 METHODS

70 2.1 Study area

71 We evaluate three TURF-reserves in Mexico (Fig 1A). The first one was created by the *Buzos y Pescadores*
72 *de la Baja California* fishing cooperative, located in Isla Natividad in the Baja California Peninsula (Fig
73 1B). The main fishery in the island is the spiny lobster (*Panulirus interruptus*), but other resources like
74 finfish, sea cucumber, red sea urchin, snail, and abalone are also an important source of income. In
75 2006, the community decided to implement two marine reserves within their fishing grounds to protect
76 commercially important invertebrate species; mainly lobster and abalone. While these reserves obtained
77 legal recognition only in 2018, they have been well enforced since their implementation.

78 The other two TURF-reserves are located in Maria Elena and Punta Herrero, in the Yucatan Peninsula
79 (Fig 1C). In contrast with Isla Nativdad, which hosts a well established fishing community, Maria Elena
80 is a fishing camp –visited intermittently during the fishing season– belonging to the Cozumel fishing
81 cooperative (*SCPP Cozumel*); Punta Herrero is home to the *SCPP José María Azcorra* cooperative, and
82 similar to Isla Natividad hosts a local community. Their main fishery is the Caribbean spiny lobster
83 (*Panulirus argus*), but they also target finfish in the off-season. Maria Elena and Punta Herrero established
84 eight marine reserves in 2012, and four marine reserves in 2013, respectively. All these reserves have been
85 legally recognized as Fishing Refuges since their creation (DOF, 2012b, 2013).

86 These communities are representative of their region in terms of ecology, socioeconomic, and governance
87 aspects. Isla Natividad, for example, is part of a greater group of fishing cooperatives belonging to
88 a Federation of Fishing Cooperatives. This group has been identified as a cohesive group that often
89 cooperates to better manage their resources (McCay, 2017; McCay et al., 2014; Aceves-Bueno et al.,
90 2017). Likewise, Maria Elena and Punta Herrero are representative of fishing cooperatives in the Mexican
91 Caribbean, which are also part of a regional Federation. Together, these three communities provide an
92 accurate representation of other fishing communities in each of their regions. While each region has
93 additional communities that have established community-based TURF-reserves, available data would not
94 allow us to perform the in-depth analysis that we undertake. Yet, given the similarities among communities
95 and the socioeconomic and governance setting under which they operate, it is safe to cautiously generalize
96 our results to other communities in Mexico and other regions around the world.

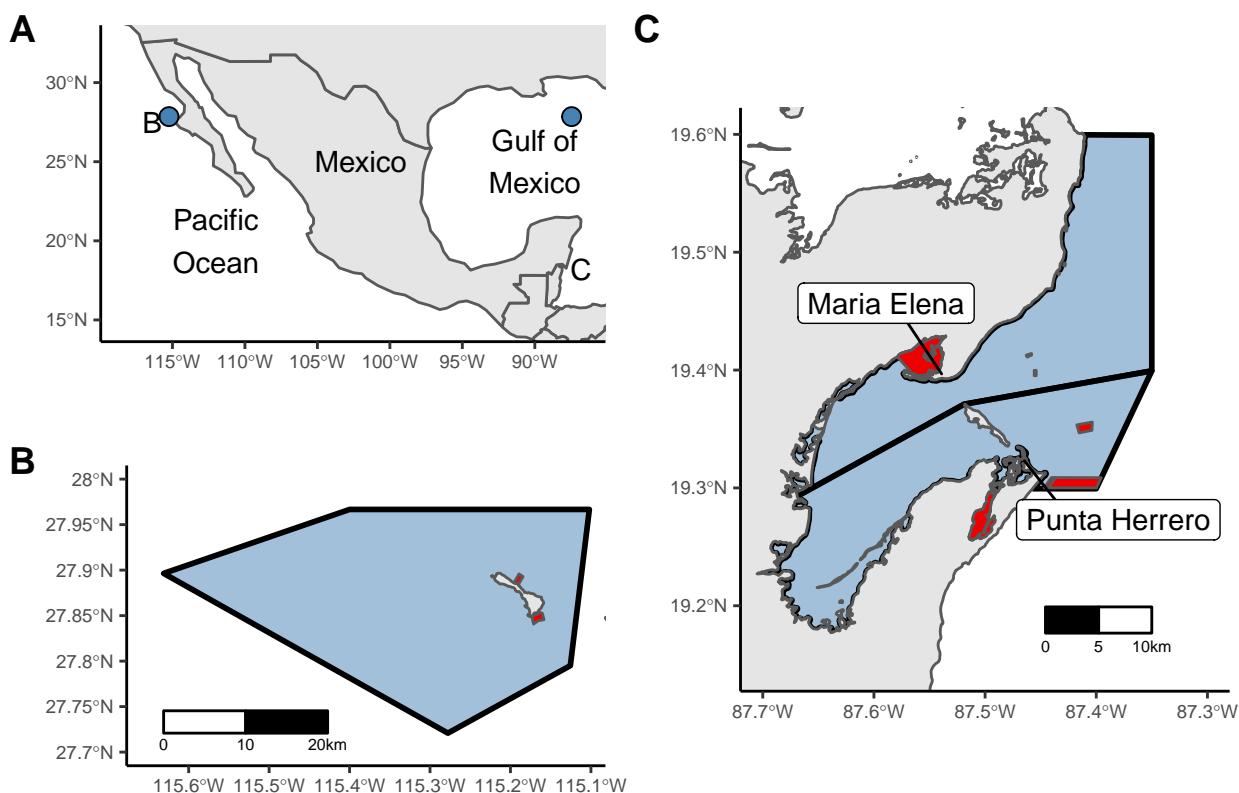


Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

97 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

115 incorporated in this analysis belong to larger regional-level Cooperative Federations, and are exposed to
 116 the same markets and institutional frameworks, making them plausible controls (McCay, 2017; Ayer et al.,
 117 2018). Landings and revenues were aggregated at the cooperative-year level, and revenues were adjusted to
 118 represent 2014 values by the Consumer Price Index for Mexico (OECD, 2017) as:

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

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

122 Data for the operationalization of the social-ecological system were collected at the community-level
 123 from official documents used in the creation and designation of the marine reserves (DOF, 2012b, 2013,
 124 2018b) and based on the authors' experience and knowledge of the communities. These include information
 125 on the resource system, the resource units, actors, and the governance system itself (Table 2).

126 2.3 Data analysis

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

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

130 We use a difference-in-differences analysis to evaluate these indicators. This approach allows us to
 131 estimate the effect that the reserve had by comparing trends across time and treatments (Moland et al.,
 132 2013; Villaseñor-Derbez et al., 2018). The analysis of ecological indicators is performed with a multiple
 133 linear regression of the form:

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

134 Where year-fixed effects are represented by $\gamma_t Year_t$, and $\beta Zone_i$ captures the difference between
 135 reserve ($Zone = 1$) and control ($Zone = 0$) sites. The interaction term $\lambda_t Year_t \times Zone_i$ represents the
 136 mean change in the indicator inside the reserve, for year t , with respect to the year of implementation in the
 137 control site (See Table ??). When evaluating biomass and densities of the invertebrate or fish communities,
 138 we include σ_j to control for species-fixed effects.

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

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

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

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

Table 2. Variables used in our Social-Ecological System analysis (IN = Isla Natividad, ME = Maria Elena, PH = Punta Herrero).

Variable	Narrative
Resource System	
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, and members trace reserve boundaries themselves. Fishers use GPS units and landmarks.
Size of resource system - TURF Area (Km ²)	IN = 889.5; ME = 353.1; PH = 299.7
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)
Age of reserves - Years since reserves were implemented	IN = 12; ME = 6; PH = 5
Resource Unit	
Number of units (catch diversity) - Diversity 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 (sea cucumber, urchin, snail, abalone).
Stock status - Status of the main fishery	Lobster stocks are well managed; along with neighboring cooperatives, the lobster fishery is MSC certified.
Actors	
Number of relevant actors - Number of fishers	IN = 98; ME = 80; PH = 21
Isolation - Level of isolation of the fishing grounds	Their fishing grounds and reserves are highly isolated and away from dense urban centers.
Governance system	
Property 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.
Operational rules - Rules implemented by individuals authorized 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. They've established penalties for breaking these rules or fishing inside the reserves.
Social monitoring - Monitoring of the activities performed by cooperative members and external fishers	Fishing cooperatives have a designated group that monitors and enforces formal and internal operational rules. They ensure fishers of their fishing cooperative adhere to the established rules, and that foreign vessels do not poach their TURF and reserves.
Biophysical monitoring - Monitoring of biological resources, including targeted species	Every year since the creation of the reserves, along with Comunidad y Biodiversidad, fishers perform standardized underwater surveys in the reserves and fishing grounds. In more recent years, they have installed oceanographic sensors to monitor for changes in oceanographic variables (e.g. Temperature, Dissolved Oxygen, Salinity, pH).

3 RESULTS

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

165 **3.1 Biological effects**

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

174 **3.2 Socioeconomic effects**

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

185 **3.3 Governance**

186 Although we have little information on the social dimension of these fisheries, we can use the social-
187 ecological systems framework to analyze the performance of each governance system (Table ??). Our
188 analysis shows that all communities analyzed share similarities in their Governance system which is based
189 on cooperatives (GS5.2.3.2), with strong rules in use that include Operational rules (GS6.2), Collective-
190 choice rules (GS6.3), Constitutional rules (GS6.3), and even Territorial use communal rights (GS6.1.4.3).
191 However, we identified important differences in terms of the actors, resource systems, and resource units.
192 Although all communities show a high level of leadership (A5), the level of trust (A6.1) is lower in Punta
193 Herrero. In general, the presence and success of conservation initiatives depends on the incentives of local
194 communities to maintain a healthy status of the resources they depend upon (Jupiter et al., 2017). The
195 enabling conditions for conservation seem to be strongly present in all communities. Due to the clarity
196 of access rights and isolation, the benefits of conservation directly benefit the members of the fishing
197 cooperative. These conditions have favored the development of efficient community-based enforcement
198 systems.

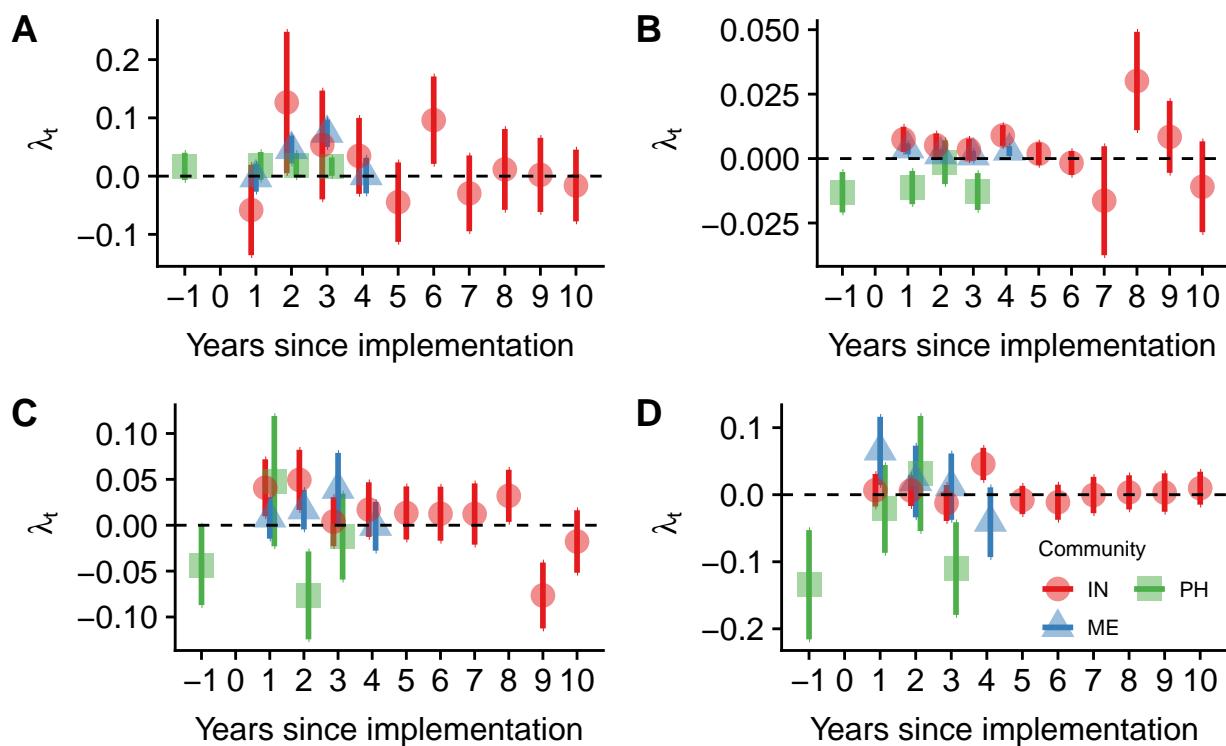


Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red circles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp.*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

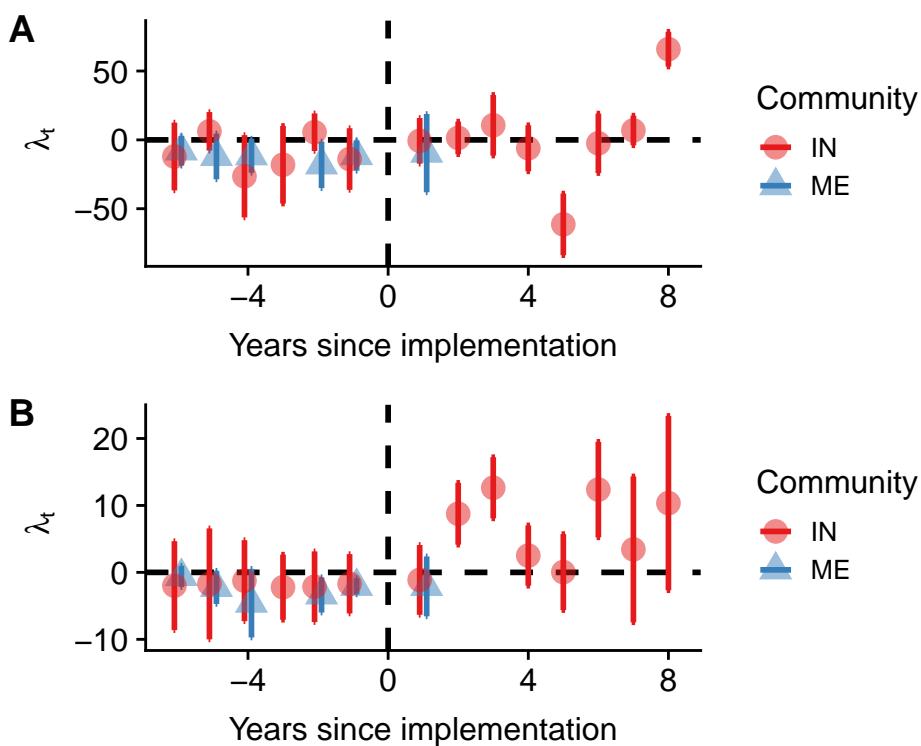


Figure 3. Effect sizes for lobster catches (A) and revenues (B) in Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points indicate the effect size and standard errors. Years have been centered to year of implementation.

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), (Beas et al. in prep., Woodson et al in review). 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

218 implies that treated sites would have followed the same trend as control sites, had the reserves not been
219 implemented. Nonetheless, temporal trends for each site don't show any significant increases (S1 Table, S2
220 Table, S3 Table), supporting our findings of lack of change in the indicators used.

221 A first possible explanation for the lack of effectiveness may be the young age of the reserves. Literature
222 shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al.,
223 2014; Babcock et al., 2010). Isla Natividad has the oldest reserves, and our SES analysis suggests that all
224 communities have a well-established community-based enforcement system. With these characteristics,
225 one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves
226 (*i.e.* < 6 years old) and effects may not yet be evident due to the short duration of protection, relative to the
227 life histories of the protected species; other community-based marine reserves in tropical ecosystems may
228 take up to six years to show a spillover effect (da Silva et al., 2015).

229 Another key condition for effectiveness is reserve size (Edgar et al., 2014), and the lack of effectiveness
230 can perhaps be attributed to poor ecological coherence in reserve design (*sensu* Rees et al. (2018)). Previous
231 research has shown that reserves in Isla Natividad yield fishery benefits for the abalone fishery (Rossetto
232 et al., 2015). Abalone are less mobile than lobsters, and perhaps the reserves provide enough protection
233 to these sedentary invertebrates, but not lobsters. Design principles developed by Green et al. (2017) for
234 marine reserves in the Caribbean state that reserves "should be more than twice the size of the home range
235 of adults and juveniles", and suggest that reserves seeking to protect spiny lobsters should have at least 14
236 km across. Furthermore, fishers may favor implementation of reserves that pose low fishing costs due to
237 their small size or location. Our analysis of economic data supports this, as neither landings nor revenues
238 showed the expected short-term costs associated to the first years of reserve implementation (Ovando et al.,
239 2016).

240 Even if reserves had appropriate sizes and were placed in optimal locations, there are other plausible
241 explanations for the observed patterns. For instance, marine reserves are only likely to provide fisheries
242 benefits if initial population sizes are low and the fishery is poorly managed (Hilborn et al., 2006).
243 Both lobster fisheries were certified by the Marine Stewardship Council (Pérez-Ramírez et al., 2016).
244 Additionally, lobster fisheries are managed via species-specific minimum catch sizes, seasonal closures,
245 protection of "berried" females, and escapement windows where traps are allowed (DOF, 1993). It is
246 uncertain whether such a well-managed fishery will experience additional benefits from marine reserves.
247 Additionally, Gelcich et al. (2008) have shown that TURFs alone can have greater biomass and richness
248 than areas operating under open access. These increased attributes perhaps minimize the difference between
249 TURF and reserve, making it difficult to detect such a small difference. Further research should focus on
250 evaluating sites in the reserve, TURF, and open access areas or similar Fish Refuges established without
251 the presence of TURFs where the impact of the reserves might be larger.

252 While the evaluated reserves have failed to provide fishery benefits up to now, there are a number of
253 additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range of
254 species and vulnerable habitat, like coral reefs. These sites can serve as an insurance against uncertainty
255 and errors in fisheries management, as well as environmental shocks (Hilborn et al., 2004, 2006; Micheli
256 et al., 2012) (,Aalto et al, in press). Self-regulation of fishing effort (*i.e.* reduction in harvest) can serve as
257 a way to compensate for future declines associated to environmental variation (Finkbeiner et al., 2018).
258 Embarking in a marine conservation project can bring the community together, which promotes social
259 cohesion and builds social capital (Fulton et al., 2019). Furthermore, showing commitment to marine
260 conservation and sustainable fishing practices allows fishers to have greater bargaining power and leverage
261 over fisheries management (Pérez-Ramírez et al., 2012). Furthermore, the lack of effectiveness observed

262 in these reserves may not be generalizable to other reserves established under the same legal frameowrk
263 (*i.e.* Fish Refges) in Mexico, and future research should aim at evaluating other areas that have also
264 been established as bottom-up processes bt without the presence of TURFs (*e.g.* DOF (2012a)), or others
265 established through a top-down process (*i.e.* DOF (2018a)).

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

CONFLICT OF INTEREST STATEMENT

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

AUTHOR CONTRIBUTIONS

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

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FIGURE CAPTIONS