

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. Small-scale 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 can improve the performance of small-scale fisheries by
7 buffering fisheries from environmental variability and management errors, while ensuring that
8 fishers reap the benefits of conservation investments. In the last 12 years, 18 old and new
9 community-based TURF-reserves gained legal recognition thanks to a 2014 regulation; their
10 effectiveness has not been formally evaluated. We combine causal inference techniques and
11 the Social-Ecological Systems framework to provide a holistic evaluation of community-based
12 TURF-reserves in three coastal communities in Mexico. We find that while reserves have not yet
13 achieved their stated goal of increasing the density of lobster and other benthic invertebrates, they
14 continue to receive significant support from the fishing communities. A lack of clear ecological
15 and socioeconomic effects likely results from a combination of factors. First, local fisheries are
16 already well managed, and it is unlikely that reserves might have a detectable effect. Second,
17 some of the reserves are not large enough to protect mobile species, like lobster. Third, some
18 of these reserves might be too young for the effects to show. Fourth, variable and extreme
19 oceanographic conditions have impacted harvested populations. However, these reserves have
20 shaped small-scale fishers' way of thinking about marine conservation, which can provide a
21 foundation for establishing additional, larger marine reserves needed to effectively conserve
22 mobile species.

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

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1 INTRODUCTION

26 Marine ecosystems around the world sustain significant impacts due to overfishing and unsustainable
27 fishing practices (Halpern et al., 2008; Worm et al., 2006; Pauly et al., 2005). In particular, small-scale
28 fisheries face great challenges since they tend to be hard to monitor and enforce (Costello et al., 2012).
29 Recent research shows that combining Territorial Use Rights for Fisheries (TURFs) with no-take marine
30 reserves (MRs) can greatly improve the performance of coastal fisheries and the health of the local resources
31 (Costello and Kaffine, 2010; Lester et al., 2017). Commonly known as TURF-Reserves, these systems
32 increase the benefits of spatial access rights allowing the maintenance of healthy resources (Afflerbach
33 et al., 2014; Lester et al., 2017). Although in theory these systems are successful (Costello and Kaffine,
34 2010; Smallhorn-West et al., 2018), there is little empirical evidence of their effectiveness and the drivers
35 of their success (Afflerbach et al., 2014; Lester et al., 2017).

36 The performance of these systems depends on how environmental and social factors combine and interact.
37 The science of marine reserves has largely focused on understanding the ecological effects of these areas,
38 which include increased biomass, species richness, and densities of organisms within the protected regions,
39 climate change mitigation, and protection from environmental variability (Lester et al., 2009; Giakoumi
40 et al., 2017; Sala and Giakoumi, 2017; Roberts et al., 2017; Micheli et al., 2012). Modelling studies show
41 that fishery benefits of marine reserves depend on initial stock status and the management under which the
42 fishery operates, as well as reserve size and the amount of larvae exported from these (Hilborn et al., 2006;
43 Krueck et al., 2017; De Leo and Micheli, 2015). Other research has focused on the relationship between
44 socioeconomic and governance structures and reserve effectiveness (Halpern et al., 2013; López-Angarita
45 et al., 2014; Mascia et al., 2017). However, to our knowledge, no studies exist that evaluate TURF-reserves
46 from both a social and ecological perspective. This is especially important in social-ecological coastal
47 systems dominated by close interactions and feedbacks between people and natural resources (Ostrom,
48 2009).

49 TURF-reserves can be created as community-based marine reserves, voluntarily established and enforced
50 by local communities. This bottom-up approach increases compliance and self-enforcement, and reserves
51 can yield benefits similar to systematically-designed reserves (Gelcich and Donlan, 2015; Espinosa-Romero
52 et al., 2014; Beger et al., 2004; Smallhorn-West et al., 2018). Community-based spatial closures occur
53 in different contexts, like the *kapu* or *ra'ui* areas in the Pacific Islands (Bohnsack et al., 2004; Johannes,
54 2002). However, MRs are difficult to enforce if they are not legally recognized, and fishers rely on the
55 exclusive access granted by the TURF. In an effort to bridge this normative gap, Mexican Civil Society
56 Organizations (CSOs) served as a link between fishers and government, and created a legal framework that
57 solves this governance issue. In Mexico, a new norm was created in 2014 allowing fishers to request the
58 legal recognition of community-based reserves as “Fish Refuges” (*Zona de Refugio Pesquero*; NOM-049-
59 SAG/PESC (2014)). Fish refuges can be implemented as temporal or partial reserves, which can protect
60 one, some, or all resources within their boundaries. Since 2012, old and new marine reserves have gained
61 legal recognition as Fishing Refuges. Of these, 18 were originally implemented as community-based
62 TURF-reserves. However, their effectiveness has not yet been formally evaluated and reported in the
63 scientific literature.

64 Here, we combine causal inference techniques and the Social-Ecological Systems (SES) framework to
65 provide a holistic evaluation of community-based TURF-reserves in three coastal communities in Mexico.
66 These three case studies span a range of ecological and social conditions representative of different regions
67 of Mexico. The objective of this work is twofold. First, to provide a triple bottom line evaluation of the
68 effectiveness of community-based marine reserves, which may inform similar processes in other countries.

69 Second, to evaluate the effectiveness of TURF-reserves established as Fish Refuges in Mexico to identify
70 opportunities where improvement or adjustment might lead to increased effectiveness. We draw from
71 lessons learned in these three case studies and provide management recommendations to maximize the
72 effectiveness of community-based marine reserves in small-scale fisheries in Mexico and in other regions
73 around the world where this tool is used to manage and rebuild their coastal fisheries.

2 METHODS

74 2.1 Study area

75 We evaluate three TURF-reserves in Mexico (Fig 1A). The first one was created by the *Buzos y Pescadores*
76 *de la Baja California* fishing cooperative, located in Isla Natividad in the Baja California Peninsula (Fig
77 1B). The main fishery in the island is the spiny lobster (*Panulirus interruptus*), but other resources like
78 finfish, sea cucumber, red sea urchin, snail, and abalone are also an important source of income. In
79 2006, the community decided to implement two marine reserves within their fishing grounds to protect
80 commercially important invertebrate species; mainly lobster and abalone. While these reserves obtained
81 legal recognition only in 2018 (DOF, 2018b), they have been well enforced since their implementation.

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

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

101 2.2 Data collection

102 We use three main sources of information to evaluate these reserves across the ecological, socioeconomic,
103 and governance dimensions. Ecological data come from the annual ecological monitoring of reserve
104 and control areas, carried out by members from each community and personnel from the Mexican CSO
105 *Comunidad y Biodiversidad* (COBI). Trained divers record richness and abundances of fish and invertebrate
106 species along replicate transects (30 × 2 m each) at depths 5-20 m in the reserves and control sites
107 (Fulton et al., 2018, 2019; Suman et al., 2010). Size structures are also collected during fish surveys. We
108 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).

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

2.3 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:

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

Where year-level fixed effects are represented by $\gamma_t Year_t$, and $\beta Zone_i$ captures the difference between reserve ($Zone = 1$) and control ($Zone = 0$) sites. The interaction term $\lambda_t Year_t \times Zone_i$ 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-level fixed effects. $\epsilon_{i,t,j}$ represents the error term of the regression.

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

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

145 The model interpretation remains as for Eq 1, but in this case the *Treated* dummy variable indicates if
146 the community has a reserve (*Treated* = 1) or not (*Treated* = 0) and $\sigma_j Com$ captures community-level
147 fixed-effects. These regression models allow us to establish a causal link between the implementation
148 of marine reserves and the observed trends by accounting for temporal and spatial dynamics (De Palma
149 et al., 2018). The effect of the reserve is captured by the λ_t coefficient, and represents the difference
150 observed between the control site before the implementation of the reserve and the treated sites at time
151 t after controlling for other time and space variations (*i.e.* γ_t and β respectively). All model coefficients
152 were estimated via ordinary least-squares and heteroskedastic-robust standard errors (Zeileis, 2004). All
153 analyses were performed in R version 3.5.1 (2018-07-02) and R Studio version 1.1.456 (R Core Team,
154 2018).

155 We use the SES framework to evaluate each community. The use of this framework standardizes our
156 analysis and allows us to communicate our results in a common language across fields by using a set
157 of previously defined variables and indicators. We based our variable selection primarily on Leslie et al.
158 (2015) and Basurto et al. (2013), who operationalized and analyzed Mexican fishing cooperatives using this
159 framework. We also incorporate other relevant variables known to influence reserve performance following
160 Di Franco et al. (2016) and Edgar et al. (2014). Table 2 shows the selected variables, their definition and
161 values.

3 RESULTS

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

167 3.1 Biological effects

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

176 3.2 Socioeconomic effects

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

185 associated with increased variation. Full tables with model coefficients are presented in the supplementary
186 materials (S4 Table, S5 Table).

187 **3.3 Governance**

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

4 DISCUSSION

202 Our results indicate that these TURF-reserves have not increased lobster densities. Additionally, no
203 co-benefits were identified when using other ecological indicators aside from the previously reported
204 buffering effect that reserves can have to environmental variability in Isla Natividad (Micheli et al., 2012).
205 The socioeconomic indicators pertaining landings and revenues showed little to no change after reserve
206 implementation. Despite the lack of evidence of the effectiveness of these reserves, most of the communities
207 show a positive perception about their performance and continue to support their presence (Ayer et al.,
208 2018). Understanding the social-ecological context in which these communities and their reserves operate
209 might provide insights as to why this happens.

210 Some works evaluate marine reserves by performing inside-outside (Guidetti et al., 2014; Friedlander
211 et al., 2017; Rodriguez and Fanning, 2017) or before-after comparisons (Betti et al., 2017). The first
212 approach does not address temporal variability, and the second can not distinguish between the temporal
213 trends in a reserve and the entire system (De Palma et al., 2018). Our approach to evaluate the temporal
214 and spatial changes provides a more robust measure of reserve effectiveness. For example, we capture
215 previously described patterns like the rapid increase observed for lobster densities in Isla Natividad on the
216 sixth year (*i.e.* 2012; Fig. 2A), a year after the hypoxia events described by Micheli et al. (2012), which
217 caused mass mortality of sedentary organisms such as abalone and sea urchins, but not lobster and finfish.
218 Yet, our empirical approach assumes control sites are a plausible counterfactual for treated sites. This
219 implies that treated sites would have followed the same trend as control sites, had the reserves not been
220 implemented. Nonetheless, temporal trends for each site don't show any significant increases (S1 Table, S2
221 Table, S3 Table), supporting our findings of lack of change in the indicators used.

222 A first possible explanation for the lack of effectiveness may be the young age of the reserves. Literature
223 shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al.,
224 2014; Babcock et al., 2010). Isla Natividad has the oldest reserves, and our SES analysis suggests that all
225 communities have a well-established community-based enforcement system. With these characteristics,

226 one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves
227 (*i.e.* < 6 years old) and effects may not yet be evident due to the short duration of protection, relative to the
228 life histories of the protected species; community-based marine reserves in tropical ecosystems may take
229 six years or more to show a spillover effect (da Silva et al., 2015).

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

241 Even if reserves had appropriate sizes and were placed in optimal locations, there are other plausible
242 explanations for the observed patterns. For instance, marine reserves are only likely to provide fisheries
243 benefits if initial population sizes are low and the fishery is poorly managed (Hilborn et al., 2004, 2006).
244 Both lobster fisheries were certified by the Marine Stewardship Council (Pérez-Ramírez et al., 2016).
245 Additionally, lobster fisheries are managed via species-specific minimum catch sizes, seasonal closures,
246 protection of “berried” females, and escapement windows where traps are allowed (DOF, 1993). It is
247 uncertain whether such a well-managed fishery will experience additional benefits from marine reserves.
248 Furthermore, Gelcich et al. (2008) have shown that TURFs alone can have greater biomass and richness
249 than areas operating under open access. This might reduce the difference between indicators from the
250 TURF and reserve sites, making it difficult to detect such a small change. Further research should focus on
251 evaluating sites in the reserve, TURF, and open access areas or similar Fish Refuges established without
252 the presence of TURFs where the impact of the reserves might be larger.

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

259 While the evaluated reserves have failed to provide fishery benefits up to now, there are a number of
260 additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range
261 of species and vulnerable habitat. These sites can serve as an insurance against uncertainty and errors in
262 fisheries management, as well as mild environmental shocks (Micheli et al., 2012; De Leo and Micheli,
263 2015; Roberts et al., 2017; Aalto et al., in press). Self-regulation of fishing effort (*i.e.* reduction in harvest)
264 can serve as a way to compensate for future declines associated to environmental variation (Finkbeiner et al.,
265 2018). Embarking in a marine conservation project can bring the community together, which promotes
266 social cohesion and builds social capital (Fulton et al., 2019). Showing commitment to marine conservation
267 and sustainable fishing practices allows fishers to have greater bargaining power and leverage over fisheries
268 management (Pérez-Ramírez et al., 2012). Furthermore, the lack of effectiveness observed in these reserves
269 should not be generalizable to other reserves established under the same legal framework (*i.e.* Fish Refuges)

270 in Mexico, and future research should aim at evaluating other areas that have also been established as
271 bottom-up processes but without the presence of TURFs (*e.g.* DOF (2012a)), or others established through
272 a top-down process (*i.e.* DOF (2018a)).

273 Community-based marine reserves in small-scale fisheries can be helpful conservation and fishery manage-
274 ment tools when appropriately implemented. Lessons learned from these cases can guide implementation
275 of community-based marine reserves elsewhere. For the particular case of the marine reserves that we
276 evaluate, the possibility of expanding reserves or merging existing polygons into larger areas should be
277 evaluated and proposed to the communities. Community-based marine reserves might have more benefits
278 that result from indirect effects of the reserves, particularly providing resilience to shocks and management
279 errors, and promoting social cohesion, which should be taken into account when evaluating the outcomes
280 of TURF-reserves. Having full community support surely represents an advantage, but it is important that
281 community-based TURF-reserves meet essential design principles such as size and placement so as to
282 maximize their effectiveness.

CONFLICT OF INTEREST STATEMENT

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

AUTHOR CONTRIBUTIONS

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

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FIGURE CAPTIONS

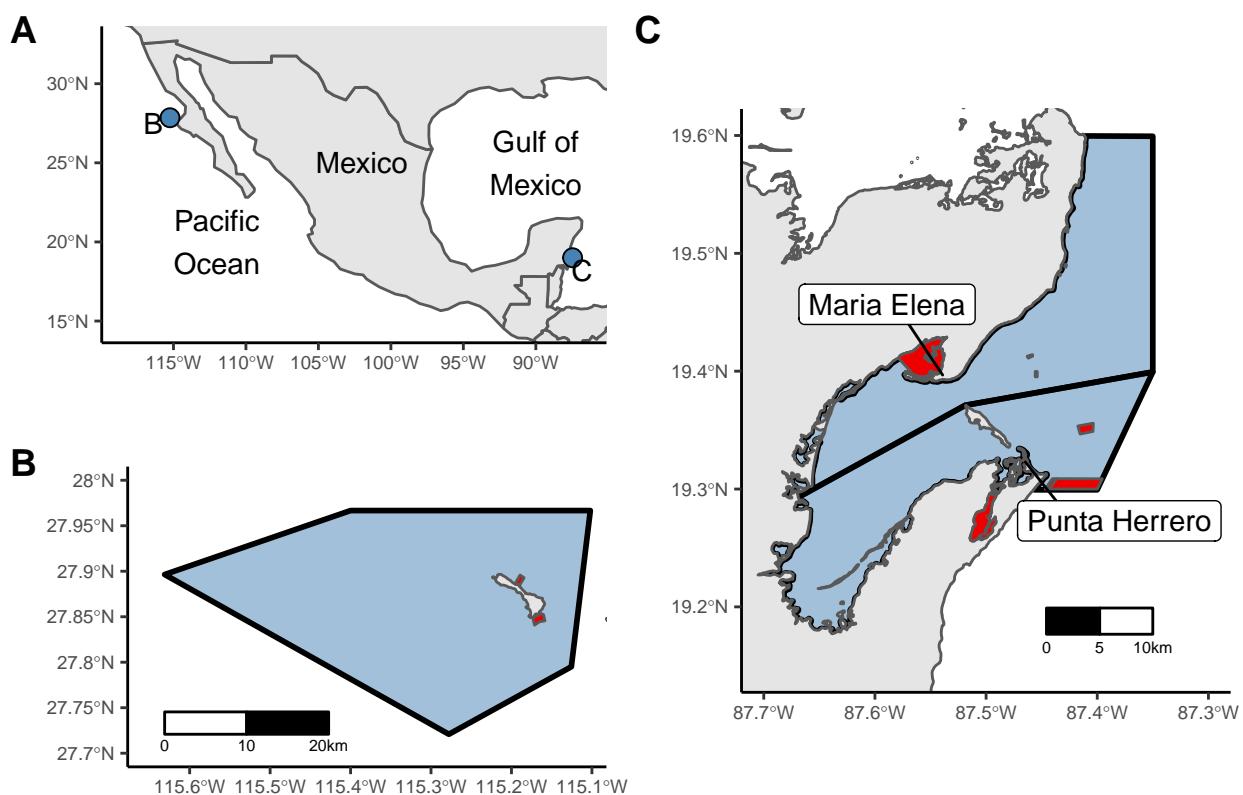


Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

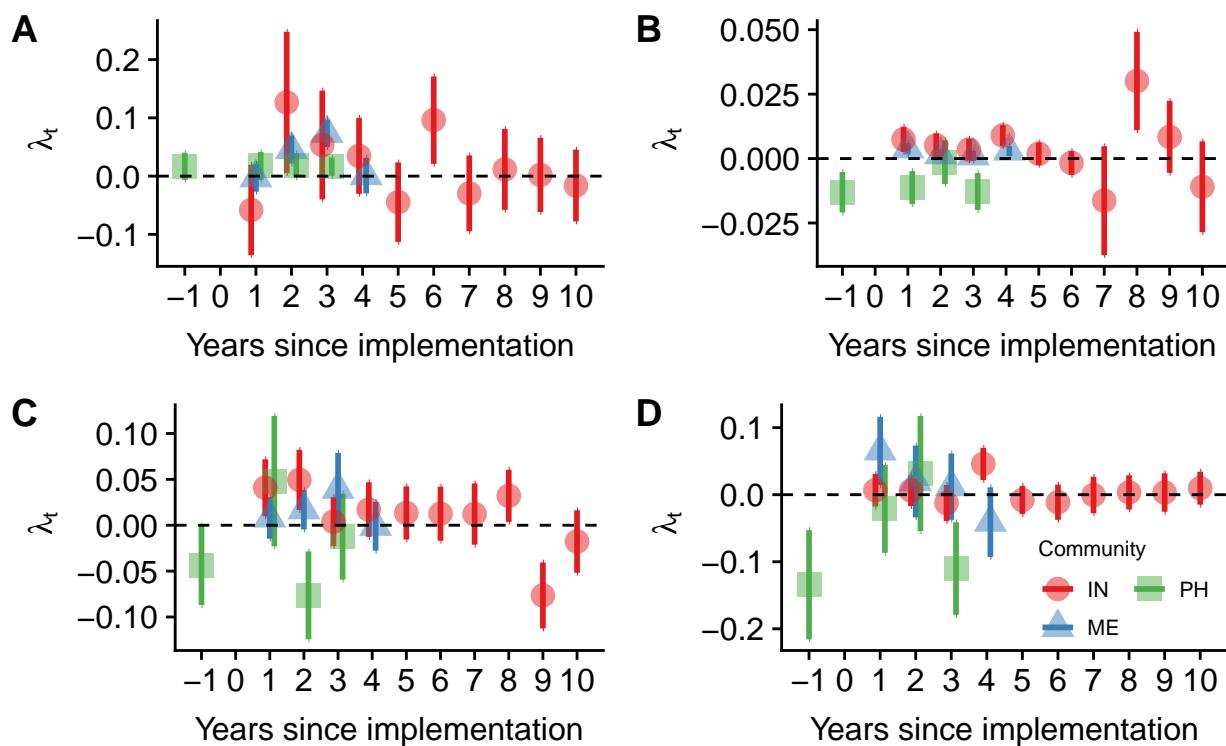


Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red circles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp.*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

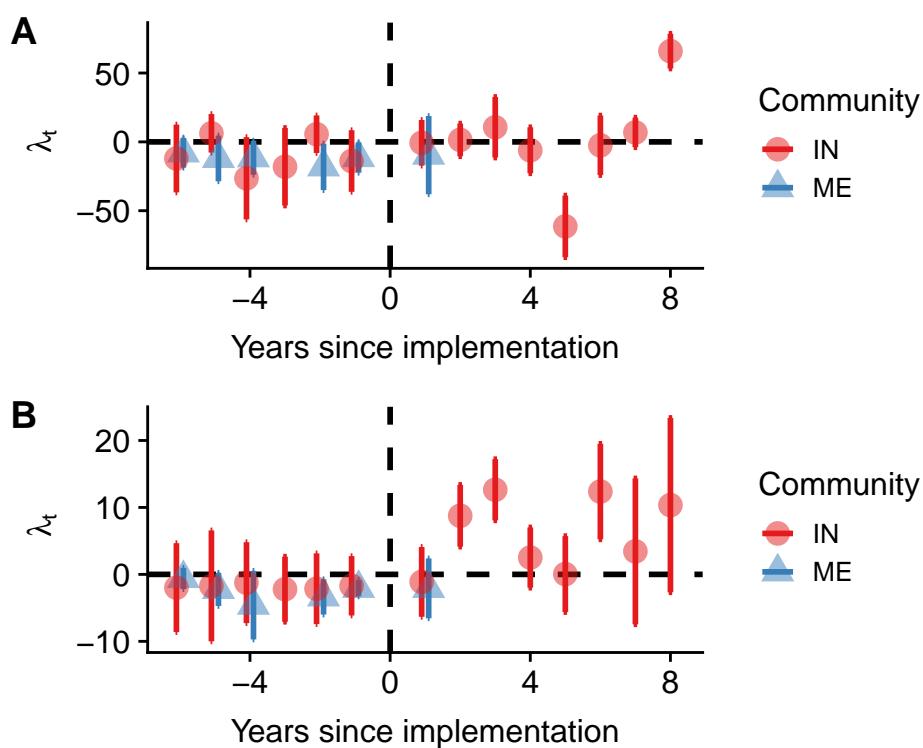


Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points indicate the effect size and standard errors. Years have been centered to year of implementation.

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

Indicator	Units
Biological	
Lobster density	org m ⁻²
Invertebrate density	org m ⁻²
Fish density	org m ⁻²
Fish biomass	Kg m ⁻²
Socioeconomic	
Income from target species	M MXP
Landings from target species	Metric Tonnes

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

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