

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 that
8 fishers reap the benefits of conservation investments. In the last 4 years, 45 TURF reserves
9 have been implemented in Mexico, but their effectiveness has not been formally evaluated. We
10 combine causal inference techniques and a social-ecological systems framework to provide a
11 holistic evaluation of community-based TURF reserves in three coastal communities in Mexico.
12 We find that while reserves have not yet achieved their stated goal of increasing lobster densities,
13 they continue to receive significant support from the fishing communities. A lack of ecological
14 and socioeconomic effects or reserves likely results from a combination of factors. First, the
15 lobster fisheries are already well managed, and it is unlikely that reserves might have a detectable
16 effect. Second, reserves are not large enough to protect lobster's home ranges. Third, some of
17 these reserves might be too young for the effects to show. However, these reserves have shaped
18 small-scale fishers' way of thinking about marine management, which can provide a foundation
19 for establishing additional, larger marine reserves needed to effectively conserve mobile species.

20 **Keywords:** Marine Protected Areas, Marine Conservation, Small-Scale Fisheries, Citizen Science, TURF-reserves, Social-Ecological
21 Systems

1 INTRODUCTION

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

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

45 TURF-reserves can be created as community-based marine reserves, voluntarily established and enforced
46 by local communities. This bottom-up approach increases compliance and self-enforcement (Gelcich and
47 Donlan, 2015; Espinosa-Romero et al., 2014; Beger et al., 2004). Community-based spatial closures occur
48 in different contexts, like the *kapu* or *ra'ui* areas in the Pacific Islands (Bohnsack et al., 2004; Johannes,
49 2002). However, without legal recognition no-take regulations are difficult to enforce and fishers rely
50 on the exclusive access granted by the TURF. In an effort to bridge this normative gap, Civil Society
51 Organizations (CSOs) have served as a link between fishers and government, and set out to create a legal
52 framework that solve this governance issue. In Mexico, a new norm was created in 2014 allowing fishers to
53 request the legal recognition of community-based reserves as “Fish Refuge” (*Zona de Refugio Pesquero*
54 ;NOM-049-SAG/PESC (2014)). Fish refuges can be implemented as temporal or partial reserves, which
55 can protect one, some, or all resources within their boundaries. Since 2014, 45 of community-based
56 marine reserves along the Pacific, Gulf of California, and Mexican Caribbean coastlines have gained legal
57 recognition. However, their effectiveness has not yet been formally evaluated and reported in the scientific
58 literature.

59 Here, we combine causal inference techniques and a social-ecological systems framework to provide a
60 holistic evaluation of community-based marine reserves in three coastal communities in Mexico. These
61 three case studies span a range of ecological and social conditions representative of different regions of
62 Mexico. The objective of this work is twofold. First, to provide a triple bottom line evaluation of the
63 effectiveness of community-based marine reserves that can inform similar processes in other countries.
64 Second, to evaluate the effectiveness of TURF-reserves established as Fish Refuges in Mexico to identify

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

2 MATERIALS AND METHODS

69 2.1 Study area

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

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

85 These communities are representative of their region in terms of ecology, socioeconomic, and governance
86 aspects. Isla Natividad, for example, is part of a greater group of fishing cooperatives that assamble
87 into a Federation of Cooperatives (**FEDECOOP**). This group of X communities has been identified as a
88 cohesive group that often cooperate to better manage their resources (McCay, 2017; McCay et al., 2014)
89 (**Aceves, 2018?**). All these communities occur in the sourthenmost raange distribution of Giant Kelp, and
90 are still under the influence of the California Current System. Likewise, Maria Elena and Punta Herrero
91 are representative of fishing cooperatives in the Mexican Caribbean. Together, these three communities
92 provide an accurate representation of all other fishing communities in Mexico. While each region has more
93 communities that have established community-based TURF-reserves, available data would not allow us to
94 perform the in-depth analysis that we undertake. Yet, given the similarities among communities and the
95 socioeconomic and governance setting under which they opperate, it is sage to generalize our results to
96 other communities in Mexico and other regions around the world.

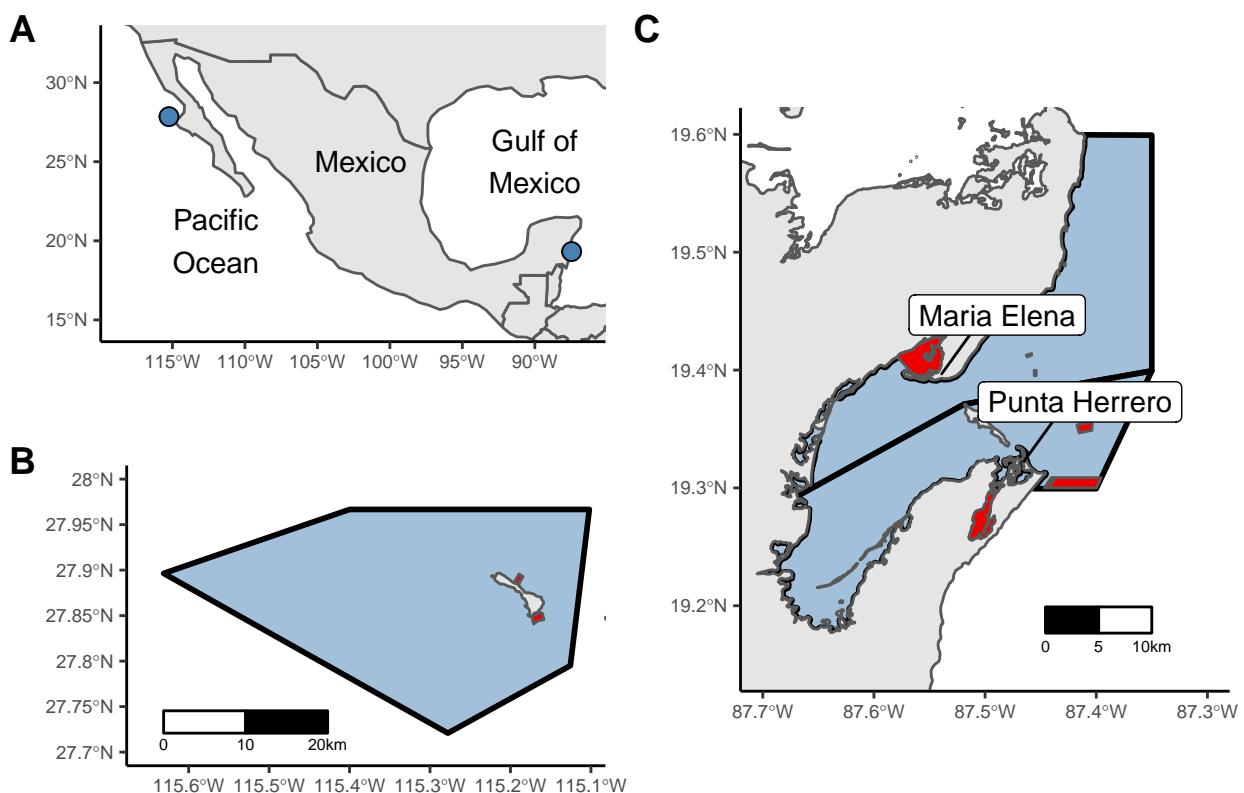


Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

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). 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. Table 1 shows a summary of the TURF-reserves included in this study.

Table 1. Summary of community-based marine reserves by community.

Community	TURF area (km ⁻²)	Reserve area (km ⁻²)	Percent as reserves	Year of implementation
Isla Natividad	889.5	1.53	0.1720067	2006
Maria Elena	353.1	0.10	0.0283206	2012
Punta Herrero	299.7	0.43	0.1434768	2013

113 Socioeconomic data come from landing receipts reported to the National Commission for Aquaculture
 114 and Fisheries (*Comisión Nacional de Acuacultura y Pesca*; CONAPESCA). Data contain monthly lobster
 115 landings (Kg) and revenues (MXP) for cooperatives with and without marine reserves (**Fig S1**). Cooperatives
 116 incorporated in this analysis belong to larger regional-level Cooperative Federations, and are exposed to
 117 the same markets and institutional frameworks, making them plausible controls (McCay, 2017; Ayer et al.,
 118 2018). Landings and revenues were aggregated at the cooperative-year level, and revenues were adjusted to
 119 represent 2014 values by the Consumer Price Index for Mexico (OECD, 2017) as:

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

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

123 Data for the qualitative analysis of the social-ecological system were collected at the community-level
 124 from official documents used in the creation and designation of the marine reserves (DOF, 2012, 2013,
 125 2018) and based on the authors' experience and knowledge of the communities. These include information
 126 on the resource system, the resource units, actors, and the governance system itself (**S1 Table**).

127 2.3 Data analysis

128 We evaluate the effect that marine reserves have had on four ecological and two socioeconomic indicators
 129 (Table 2). Recall that reserves were implemented to protect lobster and other benthic invertebrates. However,
 130 we also use the available fish data to test for associated co-benefits.

Table 2. 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

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

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

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

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

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

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

3 RESULTS

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

159 3.1 Biological effects

160 Indicators showed ambiguous responses through time for each reserve. Figure 2A shows positive effect
 161 sizes for lobster densities in Isla Natividad and Punta Herrero during the first years, but the effect is eroded
 162 through time. In the case of Maria Elena, positive changes were observed in the third and forth year. These
 163 effects are in the order of 0.2 extra organisms m⁻² for Isla Natividad and Punta Herrero, and 0.01 organisms
 164 m⁻² for Maria Elena, but are not significantly different from zero ($p > 0.05$). Likewise, no changes were

165 detected in fish biomass or invertebrate and fish densities (2B-D), where effect sizes oscillated around zero
166 without clear trends. Full tables with model coefficients are presented in the supplementary materials (**S2**
167 **Table**, **S3 Table**, **S4 Table**).

168 **3.2 Socioeconomic effects**

169 Lobster landings and revenue were only available for Isla Natividad and Maria Elena (Fig 3). For all years
170 before implementation, the effect sizes are close to zero, indicating that the control and treatment sites
171 have similar pre-treatment trends, suggesting that these are plausible controls. However, effect sizes do not
172 change after the implementation of the reserve. Interestingly, the negative effect observed for Isla Natividad
173 on year 5 correspond to the 2011 hypoxia events. The only positive change observed in lobster landings
174 is for Isla Natividad in 2014 ($p < 0.1$). The three years of post-implementation data for Maria Elena do
175 not show a significant effect of the reserve. Isla Natividad shows higher revenues after the implementation
176 of the reserve, as compared to the control communities. However, these changes are not significant and
177 are associated to increased variation. All regression coefficients for each community and indicator are
178 presented in **S5 Table**.

179 **3.3 Governance**

180 Although we have little information on the social dimension of these fisheries, we can use the social-
181 ecological systems framework (**S1 Table**) to analyze the performance of each governance system (**S6**
182 **Table**). Our analysis shows that all communities analyzed share similarities in their Governance system
183 which is based on cooperatives (GS5.2.3.2), with strong rules in use that include Operational rules (GS6.2),
184 Collective-choice rules (GS6.3), Constitutional rules (GS6.3), and even Territorial use communal rights
185 (GS6.1.4.3). However, we identified important differences in terms of the actors, resource systems, and
186 resource units. Although all communities show a high level of leadership (A5), the level of trust (A6.1) is
187 lower in Punta Herrero. In general, the presence and success of conservation initiatives depends on the
188 incentives of local communities to maintain a healthy status of the resources they depend upon (Jupiter
189 et al., 2017). The enabling conditions for conservation seem to be strongly present in all communities.
190 Due to the clarity of access rights and isolation, the benefits of conservation directly benefit the members
191 of the fishing cooperative. These conditions have favored the development of efficient community-based
192 enforcement systems.

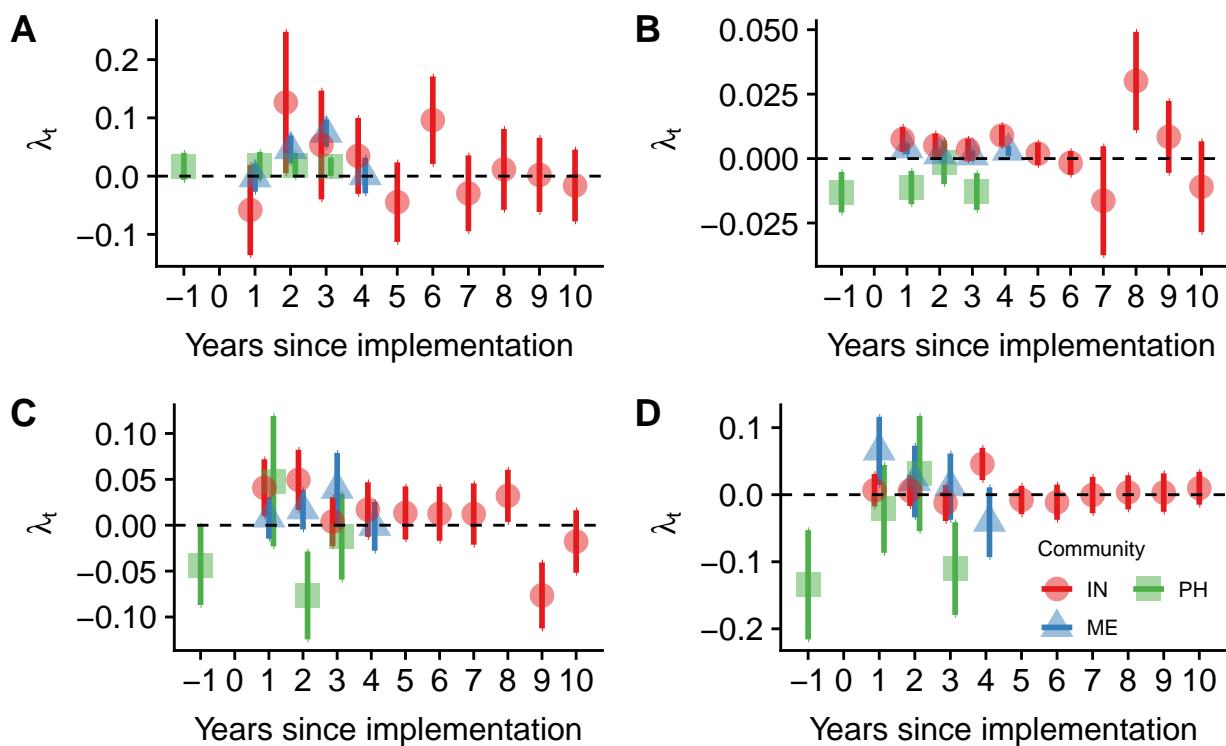


Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red circles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

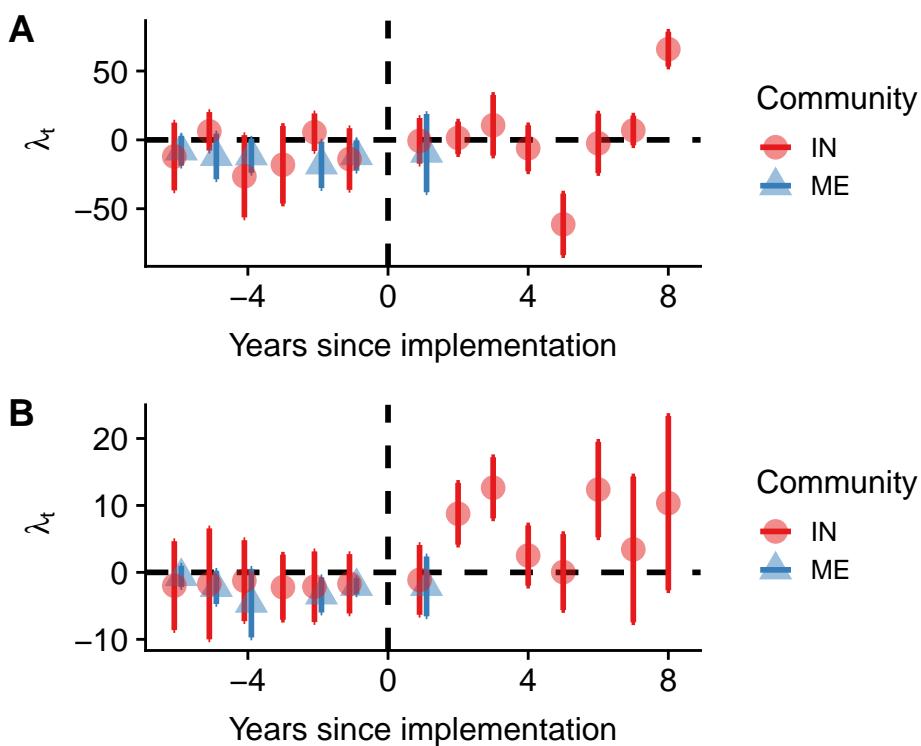


Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points indicate the effect size and standard errors. Years have been centered to year of implementation.

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 changes in lobster densities for Isla Natividad on the sixth year (*i.e.* 2012; Fig. 2A) occur 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

212 for treated sites. This implies that treated sites would have followed the same trend as control sites, had
213 the reserves not been implemented. Nonetheless, overall trends for each site don't show any significant
214 increases, supporting our findings of lack of change in the indicators used (**S2 Figure**, **S3 Figure**, **S4**
215 **Figure**, **S5 Figure**, **S6 Figure**).

216 A first possible explanation for the lack of effectiveness may be the young age of the reserves. Literature
217 shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al.,
218 2014) **Babcock PNAS 2010**. Isla Natividad has the oldest reserves, and our SES analysis suggests that all
219 communities have a well-established community-based enforcement system. With these characteristics,
220 one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves
221 (*i.e.* < 5 years old) and effects may not yet be evident due to the short duration of protection, relative to the
222 life histories of the protected species; other community-based marine reserves in tropical ecosystems may
223 take up to six years to show a spillover effect (da Silva et al., 2015).

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

234 Even if reserves had appropriate sizes and were placed in optimal locations, there are other plausible
235 explanations for the observed patterns. For instance, marine reserves are only likely to provide fisheries
236 benefits if initial population sizes are low and the fishery is poorly managed (Hilborn et al., 2006).
237 Both lobster fisheries were certified by the Marine Stewardship Council (Pérez-Ramírez et al., 2016).
238 Additionally, lobster fisheries are managed via species-specific minimum catch sizes, seasonal closures,
239 protection of "berried" females, and escapement windows where traps are allowed (DOF, 1993). It is
240 uncertain whether such a well-managed fishery will experience additional benefits from marine reserves.
241 Additionally, Gelcich et al. (2008) has shown that TURFs alone can have greater biomass and richness
242 than areas operating under open access. These increased attributes perhaps minimize the difference between
243 TURF and reserve. Further research should focus on evaluating sites in the reserve, TURF, and open access
244 areas.

245 While reserves have failed to provide fishery benefits up to now, there are a number of additional
246 ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range of species
247 and vulnerable habitat, like coral reefs. These sites can serve as an insurance against uncertainty and
248 errors in fisheries management, as well as environmental shocks (Hilborn et al., 2004, 2006; Micheli et al.,
249 2012) (, **Aalto et al, in press Am Nat**). Self-regulation of fishing effort (*i.e.* reduction in harvest) can
250 serve as a way to compensate for future declines associated to environmental variation (Finkbeiner et al.,
251 2018). Embarking in a marine conservation project can bring the community together, which promotes
252 social cohesion and builds social capital. Furthermore, showing commitment to marine conservation and
253 sustainable fishing practices allows fishers to have greater bargaining power and leverage over fisheries
254 management (Pérez-Ramírez et al., 2012).

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

CONFLICT OF INTEREST STATEMENT

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

AUTHOR CONTRIBUTIONS

267 JC and EA analyzed and interpreted data, discussed the results, and wrote the first draft. AS, SF and JT
268 discussed the results and edited the manuscript.

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274 by members of the communities here mentioned, who collected the biological data.

SUPPLEMENTAL DATA

275 Supplementary Material should be uploaded separately on submission, if there are Supplementary Figures,
276 please include the caption in the same file as the figure. LaTeX Supplementary Material templates can be
277 found in the Frontiers LaTeX folder

278 **S1 Figure**

279 Map of control and treated sites in A and control and treated landings in B

280 **S2 Figure**

281 Time series of biological indicators for IN

282 **S3 Figure**

283 Time series of biological indicators for ME

284 **S4 Figure**

285 Time series of biological indicators for PH

286 **S5 Figure**

287 Time series of economic indicators for ME

288 **S6 Figure**

289 Time series of economic indicators for PH

290 **S1 Table**

291 Coefficient estimates for biological indicators in Isla Natividad

292 **S2 Table**

293 Coefficient estimates for biological indicators in Maria Elena

294 **S3 Table**

295 Coefficient estimates for biological indicators in Punta Herrero

296 **S4 Table**

297 Coefficient estimates for economic indicators

FIGURE CAPTIONS