
Effectiveness of community-based TURF-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 and
15 socioeconomic effects likely results from a combination of factors. First, some of these reserves
16 might be too young for the effects to show. Second, the reserves are not large enough to protect
17 mobile species, like lobster. Third, variable and extreme oceanographic conditions have impacted
18 harvested populations. Fourth, local fisheries are already well managed, and it is unlikely that
19 reserves might have a detectable effect in landings. However, these reserves may provide a
20 foundation for establishing additional, larger marine reserves needed to effectively conserve
21 mobile species.

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

1 INTRODUCTION

24 Marine ecosystems around the world sustain significant impacts due to overfishing and unsustainable
25 fishing practices (Halpern et al., 2008; Worm et al., 2006; Pauly et al., 2005). In particular, small-scale
26 fisheries face great challenges since they tend to be hard to monitor and enforce (Costello et al., 2012).
27 One of the many important ways to improve the performance of coastal fisheries and health of the local
28 resources is through the implementation of Territorial Use Rights for Fisheries (TURFs) that contain
29 no-take marine reserves within them, thus creating TURF-reserve systems (Afflerbach et al., 2014; Gelcich
30 and Donlan, 2015; Lester et al., 2017).

31 TURFs are a fisheries management tool in which a well defined group of fishers have exclusive access to
32 an explicitly delimited portion of the ocean. They promote a sense of stewardship and incentivise resource
33 users to sustainably manage their resources (Gelcich et al., 2008; Costello and Kaffine, 2010; McCay et al.,
34 2014). On the other hand, no-take marine reserves (marine reserves from hereinafter) are areas where all
35 extractive activities are off-limits. These can be implemented to protect biodiversity but also as fishery
36 management tools that restrict fishing effort and gears and therefore aid in the recovery of marine stocks.
37 Commonly known as TURF-reserves, the combination of these tools should in theory increase the benefits
38 of spatial access rights allowing the maintenance of healthy resources (Afflerbach et al., 2014; Gelcich and
39 Donlan, 2015; Lester et al., 2017).

40 Conservation science has shown how marine reserves lead to increased biomass, species richness, and
41 abundance within the protected regions (Lester et al., 2009), and that these may have a series of additional
42 benefits like climate change mitigation, protection from environmental variability, and fisheries benefits
43 Roberts et al. (2017); Micheli et al. (2012); Krueck et al. (2017). Likewise, research on TURFs has shown
44 that these areas have higher abundance of targeted species than sites operating under open access and
45 even similar to that of marine reserves (Gelcich et al., 2008, 2012). The benefits resulting from reserves
46 established within TURFs (*i.e.* TURF-reserves) should be captured exclusively by the group of fishers with
47 exclusive access (Gelcich and Donlan, 2015). Although in theory these systems are successful (Gelcich
48 and Donlan, 2015; Smallhorn-West et al., 2018), there is little empirical evidence of their effectiveness and
49 the drivers of their success (Afflerbach et al., 2014; Lester et al., 2017).

50 TURF-reserve systems are inherently intricate social-ecological systems, and their effectiveness must
51 depend on how environmental and social factors combine and interact (Gelcich and Donlan, 2015). This is
52 especially important in social-ecological coastal systems dominated by close interactions and feedbacks
53 between people and natural resources (Ostrom, 2009). There is a growing body of literature focusing on
54 the relations between socioeconomic and governance structures and reserve effectiveness (Halpern et al.,
55 2013; López-Angarita et al., 2014; Mascia et al., 2017; Bergseth et al., 2018). However, to our knowledge,
56 no studies exist that evaluate TURF-reserves from both a social and ecological perspective.

57 Moreover, a new Mexican norm was created in 2014 allowing fishers to request the legal recognition of
58 community-based reserves as “Fish Refuges” (*Zona de Refugio Pesquero*; NOM-049-SAG/PESC (2014)).
59 Since 2012, old and new marine reserves have gained legal recognition as Fishing Refuges. Of these, 18
60 were originally implemented as community-based TURF-reserves. However, their effectiveness has not yet
61 been formally evaluated and reported in the scientific literature.

62 Here, we combine causal inference techniques and the Social-Ecological Systems (SES) framework to
63 evaluate community-based TURF-reserves in three coastal communities in Mexico. These three case studies
64 span a range of ecological and social conditions representative of different regions of Mexico. The objective
65 of this work is twofold. First, to provide a holistic evaluation of the effectiveness of community-based

66 TURF-reserves in terms of the changes in biological and socioeconomic indicators and the governance
67 settings under which these develop, which may inform similar processes in other countries. Second, to
68 evaluate the effectiveness of TURF-reserves established as Fish Refuges in Mexico to identify opportunities
69 where improvement or adjustment might lead to increased effectiveness. We draw from lessons learned
70 in these three case studies and provide management recommendations to maximize the effectiveness of
71 community-based TURF-reserves in small-scale fisheries where this tool is used to manage and rebuild
72 their coastal fisheries.

2 METHODS

73 2.1 TURF-reserves in Mexico

74 Before discussing our data collection methods and describing our analyses, our case studies warrant
75 some more background. Community-based marine reserves that are implemented within TURFs are a form
76 of TURF-reserves, voluntarily established and enforced by local communities. This bottom-up approach
77 increases compliance and self-enforcement, and reserves can yield benefits similar to systematically-
78 designed reserves (Beger et al., 2004; Gelcich and Donlan, 2015; Smallhorn-West et al., 2018). Community-
79 based spatial closures occur in different contexts, like the *kapu* or *ra'ui* areas in the Pacific Islands (Bohnsack
80 et al., 2004; Johannes, 2002). However, marine reserves are difficult to enforce if they are not legally
81 recognized, and fishers rely on the exclusive access granted by the TURF.

82 In an effort to bridge this normative gap, Mexican Civil Society Organizations (CSOs) served as a link
83 between fishers and government, and created a legal framework that solves this governance issue (*i.e.* Fish
84 Refuges; NOM-049-SAG/PESC (2014)). Fish refuges can be implemented as temporal or partial reserves,
85 which can protect one, some, or all resources within their boundaries. One of the ways in which fishing
86 communities have taken advantage of this new tool is by implementing marine reserves within their
87 TURFs. Our work focuses on some of these community-based TURF-reserves that occur in small-scale
88 fisheries.

89 The common setup of community-based TURF-reserves in Mexico is the following. Fishers from a
90 given community are assembled in fishing cooperatives which have exclusive fishing rights over a spatially
91 delimited area (*i.e.* TURFs shown as blue polygons in Fig 1A). Each TURF is exclusively fished by one
92 cooperative, and each community usually hosts no more than one cooperative. Fishing cooperatives
93 interested in implementing marine reserves work with CSOs to implement marine reserves within their
94 TURFs (*i.e.* TURF-reserves). Fishers then ask the government to grant legal recognition to their TURF-
95 reserves under the name of Fish refuges following a series of studies outlined in NOM-049-SAG/PESC
96 (2014).

97 2.2 Study areas

98 We evaluate three community-based no-take marine reserves implemented in Mexican TURF-managed
99 fisheries, therefore making them TURF-reserves (Fig 1A). The first one was created by the *Buzos y*
100 *Pescadores de la Baja California* fishing cooperative, located in Isla Natividad in the Baja California
101 Peninsula (Fig 1B). The main fishery in the island is the spiny lobster (*Panulirus interruptus*), but other
102 resources like finfish, sea cucumber, red sea urchin, snail, and abalone are also an important source of
103 income. In 2006, the community decided to implement two marine reserves within their fishing grounds to
104 protect commercially important invertebrate species; mainly lobster and abalone. These reserves obtained
105 legal recognition only in 2018 (DOF, 2018b).

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

114 These communities are representative of their region in terms of ecology, socioeconomic, and governance
115 aspects. Isla Natividad, for example, is part of a greater group of fishing cooperatives belonging to
116 a Federation of Fishing Cooperatives. This group has been identified as a cohesive group that often
117 cooperates to better manage their resources (McCay et al., 2014; McCay, 2017; Aceves-Bueno et al.,
118 2017). Likewise, Maria Elena and Punta Herrero are representative of fishing cooperatives in the Mexican
119 Caribbean, which are also part of a regional Federation. Together, these three communities provide an
120 accurate representation of other fishing communities that have been historically manged with TURFs in
121 each of their regions. While each region has additional communities that have established community-based
122 TURF-reserves, available data would not allow us to perform the in-depth causal inference analysis that we
123 undertake. Yet, given the similarities among communities and the socioeconomic and governance setting
124 under which they operate, it is safe to cautiously generalize our insights to other similar reserves in Mexico
125 and elsewhere around the world.

126 The regulation governing the implementation of Fish refuges states that these are fishery management
127 tools intended to have biological or socioeconomic benefits (NOM-049-SAG/PESC, 2014). For this reason,
128 the main portion of our analyses focuses on a series of biological and socioeconomic indicators that may
129 respond to reserve implementation. However, the effectiveness of conservation and fisheries management
130 interventions also depends on the social and governance structures in place. We therefore incorporate a
131 reduced version of the Social Ecological Systems framework (Ostrom, 2009) and evaluate variables and
132 indicators known to aid and hinder the effectiveness of management interventions in conservation and
133 fisheries. The incorporation of the SES is not intended to relate different levels of governance with reserve
134 effectiveness, but rather provide a discussion of potential causes of individual reserve performance. The
135 following two sections describe our data collection methods and analyses of biological and socioeconomic
136 indicators as well as the SES analysis.

137 2.3 Data collection

138 We use three main sources of information to evaluate these reserves across the ecological, socioeconomic,
139 and governance dimensions. Ecological data come from the annual ecological monitoring of reserve and
140 control sites. Reserve sites are those withing the reserves, and thus no fishing takes place. We define control
141 sites as areas where: i) habitat characteristics are similar to the corresponding reserves, ii) presumably had
142 a similar probability of being selected as reserves during the design phase, and iii) are located within the
143 TURF, where fishing occurs. We focus our evaluation on sites where data are available for reserve and
144 control sites, before and after the implementation of the reserve. This provides us with a Before-After-
145 Control-Impact (*i.e.* BACI) sampling design that allows us to capture and control for temporal and spatial
146 dynamics (Stewart-Oaten et al., 1986; De Palma et al., 2018) and causally attribute the changes to the
147 reserve (Francini-Filho and Moura, 2008; Moland et al., 2013; Villaseñor-Derbez et al., 2018).

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

155 Socioeconomic data come from landing receipts reported to the National Commission for Aquaculture
 156 and Fisheries (*Comisión Nacional de Acuacultura y Pesca*; CONAPESCA). Data contain monthly lobster
 157 landings (Kg) and revenues (MXP) for cooperatives with and without marine reserves. In this case our
 158 treated unit are the cooperatives (*i.e.* communities) that have implemented a reserve within their TURF,
 159 and the controls are nearby communities that have a TURF but did not implement a reserve. Cooperatives
 160 incorporated in this analysis belong to larger regional-level Cooperative Federations, and are exposed to
 161 the same markets and institutional frameworks, making them plausible controls (McCay, 2017; Ayer et al.,
 162 2018). Landings and revenues were aggregated at the cooperative-year level, and revenues were adjusted
 163 to represent 2014 values by the Consumer Price Index for Mexico (OECD, 2017). A table with summary
 164 statistics for this data is provided in the supplementary materials (**S3 Table**).

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

169 2.4 Data analysis

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

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

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

181 Where year-level fixed effects are represented by $\gamma_t Year_t$, and $\beta Zone_i$ captures the difference between
 182 reserve ($Zone = 1$) and control ($Zone = 0$) sites. The interaction term $\lambda_t Year_t \times Zone_i$ represents the
 183 mean change in the indicator inside the reserve, for year t , with respect to the year of implementation
 184 in the control site after accounting for the temporal and spatial differences. Therefore, we would expect
 185 this term to be positive if the indicator increases because of the reserve implementation. When evaluating
 186 biomass and densities of the invertebrate or fish communities, we include σ_j to control for species-level
 187 fixed effects. $\epsilon_{i,t,j}$ represents the error term of the regression.

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

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

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

202 We use the SES framework to evaluate each community. The use of this framework standardizes our
 203 analysis and allows us to communicate our results in a common language across fields by using a set
 204 of previously defined variables and indicators. We based our variable selection primarily on Leslie et al.
 205 (2015) and Basurto et al. (2013), who operationalized and analyzed Mexican fishing cooperatives using this
 206 framework. We also incorporate other relevant variables known to influence reserve performance following
 207 Di Franco et al. (2016) and Edgar et al. (2014). Table 2 shows the selected variables, their definition and
 208 values.

3 RESULTS

209 The following sections present the effect that marine reserves had on each of the biological and socioe-
 210 conomic indicators for each coastal community. Results are presented in terms of the difference through
 211 time and across sites, relative to the control site on the year of implementation (i.e. the difference-in-
 212 differences estimate or effect size λ_t from 1). We also provide an overview of the governance settings
 213 of each community, and discuss how these might be related to the effectiveness and performance of the
 214 reserves.

215 3.1 Biological effects

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

224 3.2 Socioeconomic effects

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

235 3.3 Governance

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

250 The fact that these reserves were re-established and just enacted shows a commitment of support (issue #16
251 on GH). the SES provides a static picture of governance and resource conditions, whereas the biological
252 assessment reflects changes through time (Issue #31).

4 DISCUSSION

253 Our results indicate that these TURF-reserves have not increased lobster densities. Additionally, no
254 co-benefits were identified when using other ecological indicators aside from the previously reported
255 buffering effect that reserves can have to environmental variability in Isla Natividad (Micheli et al., 2012).
256 The socioeconomic indicators pertaining landings and revenues showed little to no change after reserve
257 implementation. Despite the lack of evidence of the effectiveness of these reserves, most of the communities
258 show a positive perception about their performance and continue to support their presence (Ayer et al.,
259 2018). Understanding the social-ecological context in which these communities and their reserves operate
260 might provide insights as to why this happens.

261 Some works evaluate marine reserves by performing inside-outside (Guidetti et al., 2014; Friedlander
262 et al., 2017; Rodriguez and Fanning, 2017) or before-after comparisons (Betti et al., 2017). The first
263 approach does not address temporal variability, and the second can not distinguish between the temporal

264 trends in a reserve and the entire system (De Palma et al., 2018). While many ecology studies have used
265 BACI sampling designs and respective analyses (*e.g.* (Stewart-Oaten et al., 1986)), few conservations
266 studies have done so to evaluate the effect of an intervention (*i.e.* Francini-Filho and Moura (2008); Lester
267 et al. (2009); Moland et al. (2013)) which has resulted in a call for more robust analyses in conservation
268 science (Guidetti, 2002; Ferraro and Pattanayak, 2006).

269 Our approach to evaluate the temporal and spatial changes provides a more robust measure of reserve
270 effectiveness. For example, we capture previously described patterns like the rapid increase observed for
271 lobster densities in Isla Natividad on the sixth year (*i.e.* 2012; Fig. 2A), a year after the hypoxia events
272 described by Micheli et al. (2012), which caused mass mortality of sedentary organisms such as abalone
273 and sea urchins, but not lobster and finfish. Yet, our empirical approach assumes control sites are a plausible
274 counterfactual for treated sites. This implies that treated sites would have followed the same trend as
275 control sites, had the reserves not been implemented. Nonetheless, temporal trends for each site don't show
276 any significant increases (S1 Table, S2 Table, S3 Table), supporting our findings of lack of change in the
277 indicators used.

278 Our analyses of socioeconomic indicators has two limitations. First, we only look at landings and
279 revenues by landings for communities with and without TURF-reserves. There are a number of other
280 possible indicators that could show a change due to the implementation of the reserve. Notably, one often
281 cited in the literature is additional benefits, such as tourism (Viana et al., 2017). However, it is unlikely that
282 the evaluated communities will experience tourism benefits due to their remoteness and the lack of proper
283 infrastructure to sustain tourism.

284 A second limitation of our socioeconomic analysis is that we do not observe effort data, which may mask
285 the effect of the reserve. For example, if catches remain relatively unchanged but fishing effort decreased,
286 that would imply a larger catch per unit effort. Even without changes to revenues, a lower input of effort
287 (*e.g.* less fishing hours, less trips, less fuel) would imply that the fishery is more profitable. However, fishers
288 are assembled into fishing cooperatives, meaning that profits are shared amongst all members (McCay
289 et al., 2014; McCay, 2017). Therefore, the same revenue is still shared by the same amount of fishers.

290 A first possible explanation for the lack of effectiveness may be the young age of the reserves. Literature
291 shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al.,
292 2014; Babcock et al., 2010). Isla Natividad has the oldest reserves, and our SES analysis suggests that all
293 communities have a well-established community-based enforcement system. With these characteristics,
294 one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves
295 (*i.e.* < 6 years old) and effects may not yet be evident due to the short duration of protection, relative to the
296 life histories of the protected species; community-based marine reserves in tropical ecosystems may take
297 six years or more to show a spillover effect (da Silva et al., 2015).

298 Another key condition for effectiveness is reserve size (Edgar et al., 2014), and the lack of effectiveness
299 can perhaps be attributed to poor ecological coherence in reserve design (*sensu* Rees et al. (2018)). Previous
300 research has shown that reserves in Isla Natividad yield fishery benefits for the abalone fishery (Rossetto
301 et al., 2015). Abalone are less mobile than lobsters, and perhaps the reserves provide enough protection
302 to these sedentary invertebrates, but not lobsters. Design principles developed by Green et al. (2017) for
303 marine reserves in the Caribbean state that reserves "should be more than twice the size of the home range
304 of adults and juveniles", and suggest that reserves seeking to protect spiny lobsters should have at least 14
305 km across. Furthermore, fishers may favor implementation of reserves that pose low fishing costs due to
306 their small size or location. Our analysis of economic data supports this hypothesis, as neither landings

307 nor revenues showed the expected short-term costs associated to the first years of reserve implementation
308 (Ovando et al., 2016).

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

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

327 While the evaluated reserves have failed to provide fishery benefits up to now, there are a number of
328 additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range
329 of species and vulnerable habitat. These sites can serve as an insurance against uncertainty and errors in
330 fisheries management, as well as mild environmental shocks (Micheli et al., 2012; De Leo and Micheli,
331 2015; Roberts et al., 2017; Aalto et al., in press). Self-regulation of fishing effort (*i.e.* reduction in harvest)
332 can serve as a way to compensate for future declines associated to environmental variation (Finkbeiner et al.,
333 2018). Embarking in a marine conservation project can bring the community together, which promotes
334 social cohesion and builds social capital (Fulton et al., 2019). Showing commitment to marine conservation
335 and sustainable fishing practices allows fishers to have greater bargaining power and leverage over fisheries
336 management (Pérez-Ramírez et al., 2012). Furthermore, the lack of effectiveness observed in these reserves
337 should not be generalizable to other reserves established under the same legal framework (*i.e.* Fish Refuges)
338 in Mexico, and future research should aim at evaluating other areas that have also been established as
339 bottom-up processes but without the presence of TURFs (*e.g.* DOF (2012a)), or others established through
340 a top-down process (*i.e.* DOF (2018a)).

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

CONFLICT OF INTEREST STATEMENT

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

AUTHOR CONTRIBUTIONS

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

FUNDING

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

ACKNOWLEDGMENTS

360 The authors wish to acknowledge Imelda Amador for contributions on the governance data, as well as
361 pre-processing biological data. This study would have not been possible without the effort by members of
362 the fishing communities here mentioned, who participated in the data-collection process.

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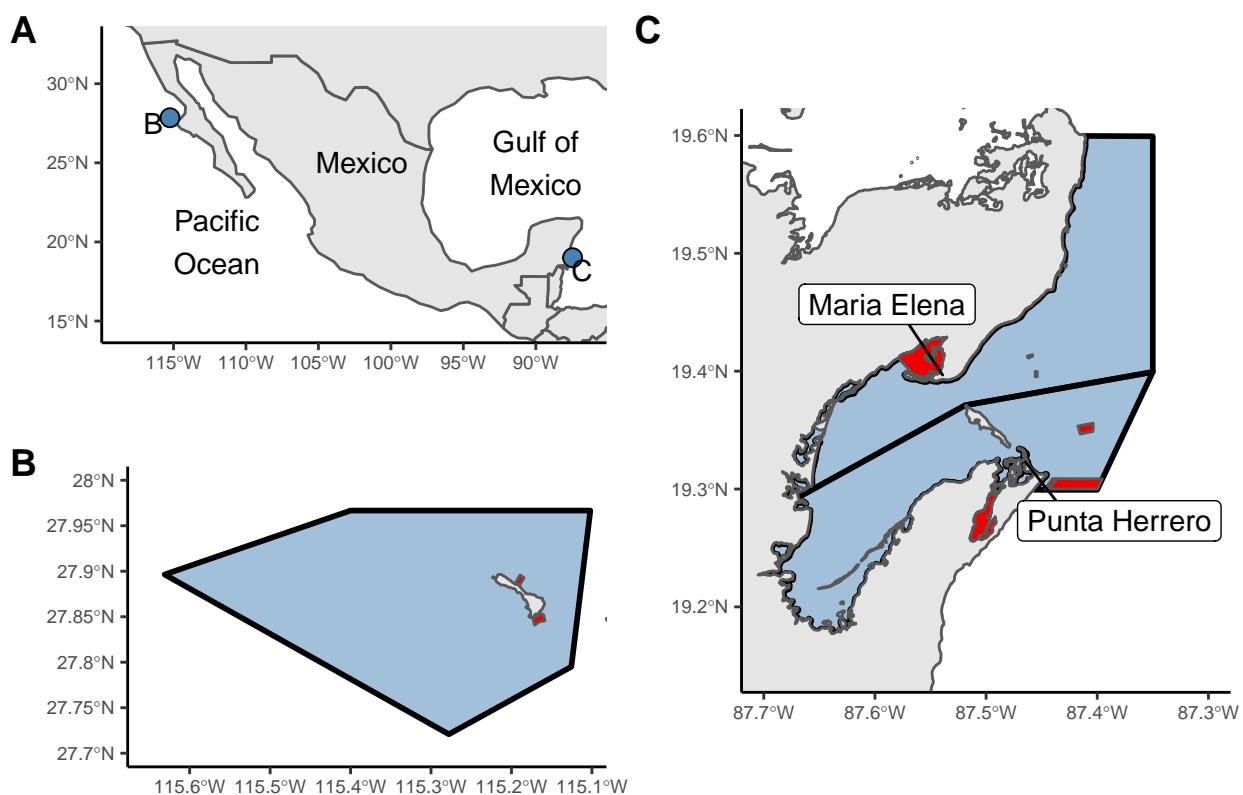
FIGURE CAPTIONS

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.

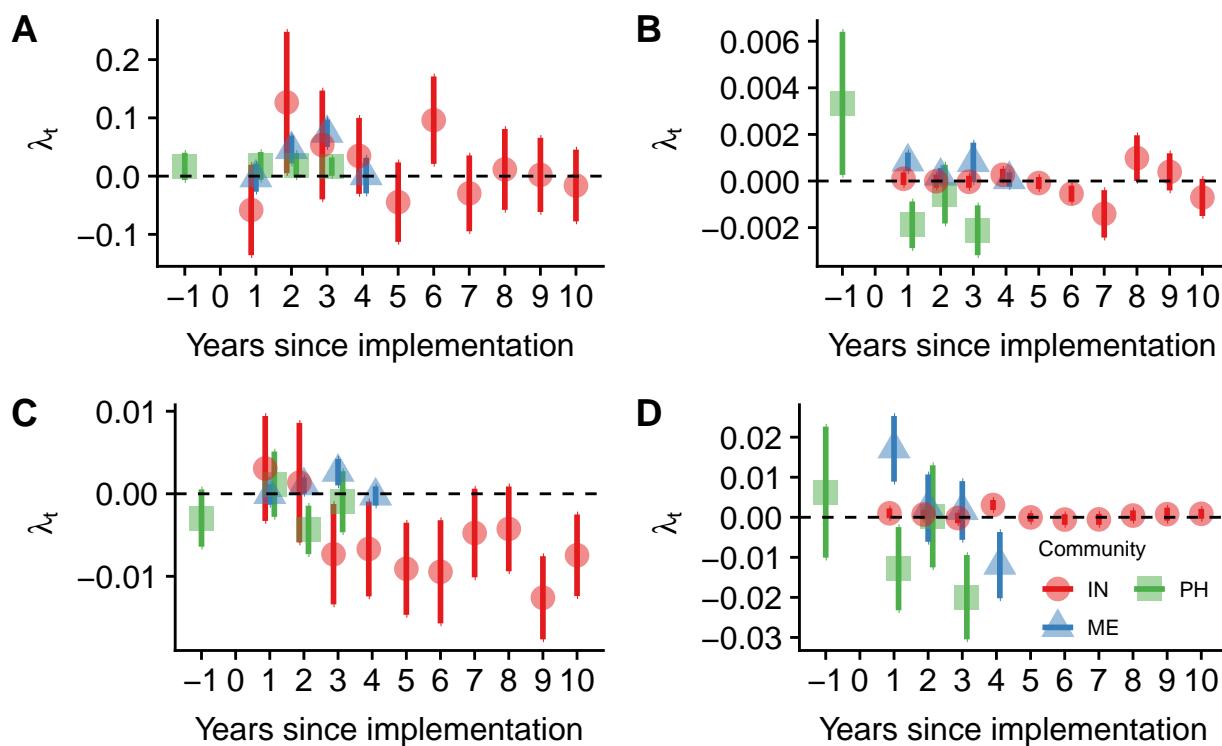


Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red circles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

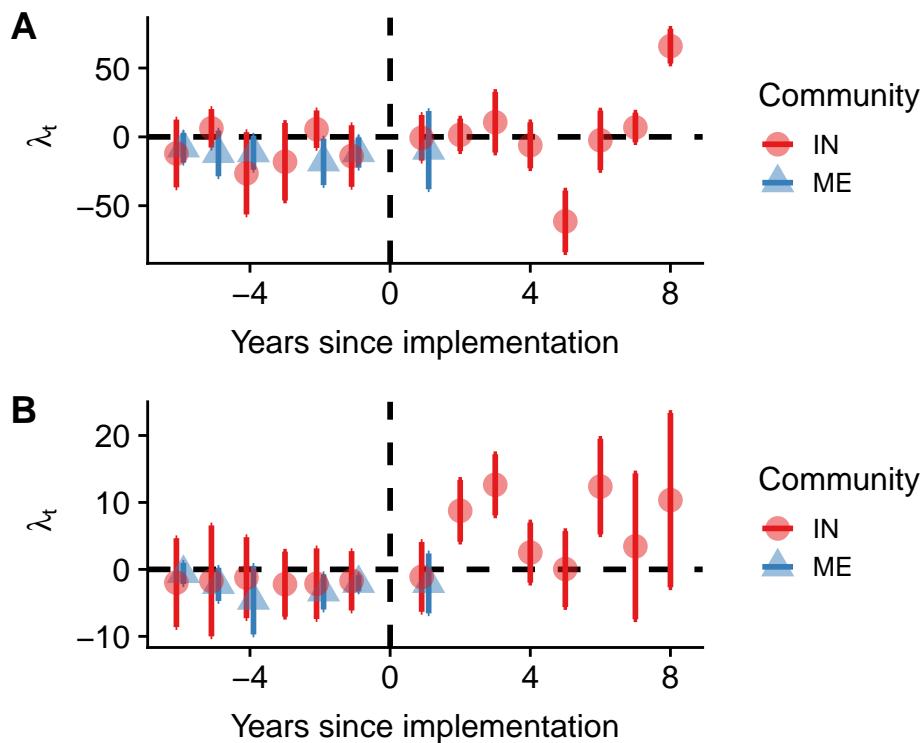


Figure 3. Effect sizes for lobster catches (A) and revenues (B) in Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points are jittered horizontally to avoid overplotting. Points indicate the effect size and error bars are heteroskedastic-robust standard errors. Years have been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

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

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

Table 2. Variables for the Social-Ecological System analysis (IN = Isla Natividad, ME = 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.