

# Effectiveness of community-based TURF-reserves in Mexican small-scale fisheries

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## 2 ABSTRACT

3 Coastal marine ecosystems provide livelihoods for small-scale fishers and coastal commu-  
4 nities around the world. Small-scale fisheries face great challenges since they are difficult to  
5 monitor, enforce, and manage. Combining territorial use rights for fisheries (TURF) with no-  
6 take marine reserves to create TURF-reserves can improve the performance of small-scale  
7 fisheries by buffering fisheries from environmental variability and management errors, while  
8 ensuring that fishers reap the benefits of conservation investments. In the last 13 years, 18  
9 old and new community-based Mexican TURF-reserves gained legal recognition thanks to a  
10 regulation passed in 2012; their effectiveness has not been formally evaluated. We combine  
11 causal inference techniques and the Social-Ecological Systems framework to provide a holistic  
12 evaluation of community-based TURF-reserves in three coastal communities in Mexico. We find  
13 that reserves have not yet achieved their stated goal of increasing the density of lobster and  
14 other benthic invertebrates. A lack of clear ecological and socioeconomic effects likely results  
15 from a combination of factors. First, some of these reserves might be too young for the effects to  
16 show. Second, the reserves are not large enough to protect mobile species, like lobster. Third,  
17 variable and extreme oceanographic conditions have impacted harvested populations. Fourth,  
18 local fisheries are already well managed, and it is unlikely that reserves might have a detectable  
19 effect in catches. However, these reserves may provide a foundation for establishing additional,  
20 larger marine reserves needed to effectively conserve mobile species.

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

## 1 INTRODUCTION

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

30 TURFs are a fisheries management tool in which a well-defined group of fishers (*e.g.* fishing cooperatives)  
31 have exclusive access to an explicitly delimited portion of the ocean. They promote a sense of stewardship  
32 and incentivise resource users to sustainably manage their resources (Gelcich et al., 2008; Costello and  
33 Kaffine, 2010; McCay et al., 2014). On the other hand, no-take marine reserves (marine reserves from  
34 hereinafter) are areas where all extractive activities are off-limits. These can be implemented to protect  
35 biodiversity but also as fishery management tools to aid in the recovery of marine stocks. These instruments  
36 can be combined by establishing a marine reserve within a TURF, thus making them TURF-reserves  
37 (Afflerbach et al., 2014; Gelcich and Donlan, 2015; Lester et al., 2017).

38 Conservation science has shown how marine reserves may lead to increased biomass, species richness, and  
39 abundance within the protected regions (Lester et al., 2009), and that these may have a series of additional  
40 benefits such as mitigation and adaptation to climate change effects, protection from environmental  
41 variability, and fisheries benefits (Roberts et al., 2017; Micheli et al., 2012; Krueck et al., 2017). Likewise,  
42 research on TURFs has shown that these areas have higher abundance of targeted species than sites  
43 operating under open access and even similar to that of marine reserves (Gelcich et al., 2008, 2012).  
44 The benefits resulting from reserves established within TURFs (*i.e.* TURF-reserves) should be captured  
45 exclusively by the group of fishers with exclusive access (Gelcich and Donlan, 2015). Although in theory  
46 these systems are expected to be successful (Smallhorn-West et al., 2018), there is little empirical evidence  
47 of their effectiveness and the drivers of their success. Moreover, TURF-reserve systems are inherently  
48 intricate social-ecological systems, and their effectiveness must depend on how environmental and social  
49 factors combine and interact (Ostrom, 2009; Gelcich and Donlan, 2015). It is therefore important to  
50 consider not only the indicators of interest, but also the governance settings under which the reserves  
51 operate.

52 Recent changes in fisheries regulation in Mexico provide a ripe opportunity to study the effectiveness  
53 of community-based TURF-reserves in small-scale fisheries. In Mexico, a legal framework created in  
54 2012 allows fishers to request legal recognition of community-based reserves as “Fish Refuges” (*Zona*  
55 *de Refugio Pesquero*, described in more detail below; NOM-049-SAG/PESC (2014)). Since 2012, 45  
56 old and new marine reserves have gained legal recognition as Fish Refuges. Of these, 18 were originally  
57 implemented within TURFs. However, their effectiveness has not yet been formally evaluated and reported  
58 in the scientific literature.

59 Here, we combine causal inference techniques and the Social-Ecological Systems (SES) framework  
60 to evaluate community-based TURF-reserves in three coastal communities in Mexico. The objective  
61 of this work is twofold. First, to provide a holistic evaluation of the effectiveness of community-based  
62 TURF-reserves in terms of the changes in biological and socioeconomic indicators and the governance  
63 settings under which these develop, which may inform similar processes in other countries. Second, to  
64 identify opportunities where improvement or adjustment might lead to increased effectiveness. We draw

65 from lessons learned in these three case studies and provide management recommendations to maximize  
66 the effectiveness of community-based TURF-reserves in small-scale fisheries where this tool is used to  
67 manage and rebuild coastal fisheries.

## 2 METHODS

### 68 2.1 TURF-reserves in Mexico

69 Community-based marine reserves that are implemented within TURFs are a form of TURF-reserve,  
70 voluntarily established and enforced by local communities. This bottom-up approach can increase complia-  
71 nce and self-enforcement, and reserves can yield benefits similar to systematically-designed reserves (Beger  
72 et al., 2004; Smallhorn-West et al., 2018). Community-based spatial closures occur in different contexts,  
73 like the *kapu* or *ra’ui* areas in the Pacific Islands (Johannes, 2002; Bohnsack et al., 2004). However,  
74 community-based reserves can be hard to enforce if they are not legally recognized. In such conditions,  
75 TURF fishers must rely on the exclusive access of the TURF to maintain high levels of compliance.

76 In an effort to bridge this normative gap, Mexican Civil Society Organizations (CSOs) served as a link  
77 between fishers and government, and helped create a legal framework that solves this governance issue:  
78 Fish Refuges (NOM-049-SAG/PESC, 2014). Fish Refuges can be implemented as permanent, temporary  
79 or partial reserves, which can protect one, some, or all resources within their boundaries. One of the ways  
80 in which fishing communities have taken advantage of this new tool is by implementing temporary marine  
81 reserves within their TURFs with a defined expiration date (often five years). When the expiration date is  
82 reached, fishers can chose to open the reserves to fishing or re-establish them. Our work focuses on Fish  
83 Refuges implemented as community-based TURF-reserves in small-scale fisheries.

84 The most common setup of community-based TURF-reserves in Mexico is the following. Fishers from a  
85 given community are assembled in fishing cooperatives which have exclusive fishing rights over a spatially  
86 delimited area (*i.e.* TURFs shown as blue polygons in Fig 1A). Each TURF is exclusively fished by one  
87 cooperative, and each community usually hosts no more than one cooperative. The profits from each  
88 TURF are shared amongst all fishers from the cooperative. Fishing cooperatives interested in implementing  
89 marine reserves work with CSOs to implement marine reserves within their TURFs (*i.e.* TURF-reserves).  
90 Fishers then ask the government to grant legal recognition to their TURF-reserves as Fish Refuges after  
91 conducting a series of studies outlined in the regulation, namely the creation of a baseline of ecological  
92 data and historical performance of the fishery (NOM-049-SAG/PESC, 2014).

### 93 2.2 Study areas

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

102 The other two TURF-reserve systems are located in Maria Elena and Punta Herrero, in the Yucatan  
103 Peninsula (Fig 1C). In contrast with Isla Nativdad, which hosts a well-established fishing community,  
104 Maria Elena is a fishing camp visited intermittently during the fishing season that belongs to the *Cozumel*

105 fishing cooperative. Punta Herrero is home to the *José María Azcorra* fishing cooperative, and similar to  
106 Isla Natividad hosts a small community. Their main fishery is the Caribbean spiny lobster (*Panulirus argus*),  
107 but they also target finfish in the off-season. Maria Elena and Punta Herrero established eight and four  
108 marine reserves in 2012 and 2013, respectively. These reserves have been legally recognized as Fishing  
109 Refuges since their original implementation (DOF, 2012b, 2013) and subsequent re-establishments (DOF,  
110 2017).

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

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

### 133 2.3 Data collection

134 We use three main sources of information to evaluate these reserves across ecological, socioeconomic,  
135 and governance dimensions. Ecological data come from the annual ecological monitoring of reserve and  
136 control sites. Reserve sites are areas where no fishing occurs. Control sites are areas that meet the following  
137 criteria: i) habitat characteristics are similar to the corresponding reserves, ii) presumably had a similar  
138 probability of being selected as reserves during the design phase, iii) are located within the TURF, where  
139 fishing occurs, and iv) are not directly adjacent to the reserves. We focus our evaluation on sites where data  
140 are available for reserve and control sites, before and after the implementation of the reserve. This provides  
141 us with a Before-After-Control-Impact (*i.e.* BACI) sampling design that allows us to capture and control  
142 for temporal and spatial dynamics (Stewart-Oaten et al., 1986; De Palma et al., 2018) and causally attribute  
143 the changes to the reserve (Francini-Filho and Moura, 2008; Villaseñor-Derbez et al., 2018).

144 The biological data are collected by members from each community and personnel from the Mexican  
145 CSO *Comunidad y Biodiversidad* (COBI). Trained divers record species richness and abundances of fish  
146 and invertebrate species along replicate transects (30 × 2 m each) at depths 5–20 m in the reserves and  
147 control sites, where a minimum of 4 transects per site are performed (Suman et al., 2010; Fulton et al., 2018,

148 2019). Size structures are also collected during fish surveys. All sites were surveyed annually, and at least  
 149 once before implementation of the reserves. A summary of sampling effort is shown in the supplementary  
 150 materials (Tables S1-S2).

151 Socioeconomic data come from landing receipts reported to the National Commission for Aquaculture  
 152 and Fisheries (*Comisión Nacional de Acuacultura y Pesca*; CONAPESCA). Data contain monthly lobster  
 153 landings (Kg) and revenues (MXP) for TURF-managed cooperatives with and without marine reserves.  
 154 In this case our treated unit are the cooperatives (*i.e.* communities) that have implemented a reserve  
 155 within their TURF, and the controls are nearby communities that have a TURF but did not implement a  
 156 reserve. These communities have TURFs adjacent to the TURFs of our focal communities. Cooperatives  
 157 incorporated in this analysis have similar number of members, belong to larger regional-level Cooperative  
 158 Federations, and are exposed to the same markets and institutional frameworks, making them plausible  
 159 controls (McCay et al., 2014; McCay, 2017; Ayer et al., 2018). Landings and revenues were aggregated  
 160 at the cooperative-year level, and revenues were adjusted to represent 2014 values by the Consumer  
 161 Price Index for Mexico (OECD, 2017). A table with summary statistics for this data is provided in the  
 162 supplementary materials (Table S3, Figure S5).

163 Data for the evaluation of the SES were collected at the community-level from official documents used  
 164 in the design, creation, and designation of the marine reserves. These include the technical studies that  
 165 the cooperatives submit when they request recognition of their reserves, as well as the official enactments  
 166 (DOF, 2012b, 2013, 2018b). We also complimented information based on the authors' experience and  
 167 knowledge of the communities. We collected information on the Resource Systems, Resource Units, Actors,  
 168 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 since reserve implementation (Moland et al., 2013; Villaseñor-  
 178 Derbez et al., 2018). To perform difference-in-differences, we regress the indicator of interest on a dummy  
 179 variable for treatment, a dummy variable for years, and the interaction term between these with a multiple  
 180 linear regression of the form:

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

181 Where year-level fixed effects capturing a temporal trend are represented by  $\gamma_t Year_t$ , and  $\beta Zone_i$   
 182 captures the difference between reserve ( $Zone = 1$ ) and control ( $Zone = 0$ ) sites. The effect of the reserve  
 183 is captured by the  $\lambda_t$  coefficient, and represents the difference observed between the control site before  
 184 the implementation of the reserve and the treated sites at time  $t$  after controlling for other time and space  
 185 variations (*i.e.*  $\gamma_t$  and  $\beta$  respectively). Therefore, we would expect this term to be positive if the indicator  
 186 increases because of the reserve. Finally,  $\epsilon_{i,t}$  represents the error term of the regression.

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

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

191 The coefficient interpretations remains as for Eq. 1, but in this case the *Treated* dummy variable  
 192 indicates if the community has a reserve (*Treated* = 1) or not (*Treated* = 0). These regression models  
 193 allow us to establish a causal link between the implementation of marine reserves and the observed  
 194 trends by accounting for temporal and site-specific dynamics (De Palma et al., 2018). For each indicator  
 195 in each community, we fit one model (*e.g.* there are three models for lobster density, one for each  
 196 community) for a total of 12 biological model fits and four socioeconomic model fits. Model coefficients  
 197 were estimated via ordinary least-squares and used heteroskedastic-robust standard errors (Zeileis, 2004).  
 198 All analyses were performed in R version 3.5.2 and R Studio version 1.1.456 (R Core Team, 2018).  
 199 All data and code needed to reproduce our analyses are available in a GitHub repository at: <https://github.com/jcvdav/ReserveEffect>.  
 200

201 We use the SES framework to evaluate each community and create a narrative that provides context for  
 202 each community. The use of this framework standardizes our analysis and allows us to communicate our  
 203 results in a common language across fields by using a set of previously defined variables and indicators.  
 204 Due to the lack of sufficient information to quantitatively operationalize the social-ecological systems  
 205 framework for these case studies (as in Leslie et al. (2015)), we followed a similar approach to Basurto et al.  
 206 (2013), who used the SES framework as a classification system of the available information to qualitatively  
 207 analyze fisheries systems. We based our variable selection primarily on Leslie et al. (2015) and Basurto  
 208 et al. (2013), who operationalized and analyzed Mexican fishing cooperatives using this framework, and  
 209 identified the key variables relevant to fishing cooperatives in Mexico. We also incorporate other relevant  
 210 variables known to influence reserve performance following Di Franco et al. (2016) and Edgar et al. (2014).  
 211 Table 2 shows the selected variables, along with definitions and values.

### 3 RESULTS

212 The following sections present the effect that marine reserves had on the biological and socioeconomic  
 213 indicators for each coastal community. Results are presented in terms of difference through time and across  
 214 sites, relative to the control site on the year of implementation (*i.e.* the difference-in-differences estimate  
 215 or effect size  $\lambda_t$  from Eqs. 1 and 2). We also provide an overview of the governance settings of each  
 216 community, and discuss how these might be related to the effectiveness and performance of the reserves.

#### 217 3.1 Biological effects

218 Indicators showed ambiguous responses through time for each reserve. Figure 2A shows positive effect  
 219 sizes for lobster densities in Isla Natividad and Punta Herrero during the first years, but the effect is eroded  
 220 through time. In the case of Maria Elena, positive changes were observed in the third and fourth year.  
 221 These effects are in the order of 0.2 extra organisms  $m^{-2}$  for Isla Natividad and Punta Herrero, and 0.01  
 222 organisms  $m^{-2}$  for Maria Elena, but are not significantly different from zero ( $p > 0.05$ ). Likewise, no  
 223 significant changes were detected in fish biomass or invertebrate and fish densities (Fig. 2B-D), where

224 effect sizes oscillated around zero without clear trends. Figures and tables with time series of indicators  
225 and model coefficients are presented in the supplementary materials (Figures S1-S4, Tables S4-S6).

### 226 **3.2 Socioeconomic effects**

227 Lobster landings and revenue were only available for Isla Natividad and Maria Elena (Fig 3). For all  
228 years before implementation, the effect sizes are close to zero, indicating that the control and treatment  
229 sites have similar pre-treatment trends, suggesting that these are plausible controls. However, effect sizes  
230 do not change after the implementation of the reserve. Interestingly, the negative effect observed for Isla  
231 Natividad on year 5 corresponds to the 2011 hypoxia events (Micheli et al., 2012). The only positive change  
232 observed in lobster landings is for Isla Natividad in 2014 ( $p < 0.1$ ). The year of post-implementation data  
233 for Maria Elena does not show a significant effect of the reserve. Isla Natividad shows higher revenues  
234 after the implementation of the reserve, as compared to the control communities. However, these changes  
235 are only significant for the third year ( $p < 0.05$ ). Full tables with model coefficients are presented in the  
236 supplementary materials (Tables S4-S5).

### 237 **3.3 Governance**

238 Our qualitative implementation of the social-ecological systems framework allowed us to systematically  
239 identify important differences between the case studies' governance systems and incorporate other char-  
240 acteristics of these fisheries neglected during the process of data collection (Table 2). We find that the  
241 analyzed communities share similarities known to foster sustainable resource management and increase  
242 reserve effectiveness. For example, fishers operate within clearly outlined TURFs (RS2, GS6.1.4.3) that  
243 provide exclusive access to resources and reserves. Along with their relatively small groups (A1 - Number  
244 of relevant actors), Isolation (A3), Operational rules (GS6.2), Social monitoring (GS9.1), and Graduated  
245 sanctions (GS10.1), these fisheries have solid governance structures that enable them to monitor their  
246 resources and enforce rules to ensure sustainable management. In general, success of conservation initia-  
247 tives depends on the incentives of local communities to maintain a healthy status of the resources upon  
248 which they depend (Jupiter et al., 2017). Due to the clarity of access rights and isolation, the benefits of  
249 conservation directly benefit the members of the fishing cooperatives, which have favored the development  
250 of efficient community-based enforcement systems. However, our SES analysis also highlights factors that  
251 might hinder reserve performance or mask outcomes. While total reserve size ranges from 0.2% to 3.7% of  
252 the TURF area, individual reserves are often small (RS3); the largest reserve is only 4.37 km<sup>2</sup>, and the  
253 smallest one is 0.09 km<sup>2</sup>. Reserves are also relatively young (RS5). Additionally, fishers harvest healthy  
254 stocks (RS4.1), and it is unlikely that marine reserves will result in increased catches.

## 4 DISCUSSION

255 Our results indicate that these TURF-reserves have not increased lobster densities. Additionally, no  
256 co-benefits were identified when using other ecological indicators aside from the previously reported  
257 buffering effect that reserves can have to environmental variability in Isla Natividad (Micheli et al., 2012).  
258 The socioeconomic indicators pertaining landings and revenues showed little to no change after reserve  
259 implementation. Lastly, the communities exhibit all the social enabling conditions for effective reserve  
260 and resource management. Here we discuss possible shortcomings in our analyses as well as possible  
261 explanations for the observed patterns.

262 While many ecology studies have used BACI sampling designs and respective analyses (e.g. Stewart-  
263 Oaten et al. (1986)), few conservation studies have done so to evaluate the effect of an intervention (e.g.

264 Francini-Filho and Moura (2008); Lester et al. (2009); Moland et al. (2013)) which has resulted in a call for  
265 more robust analyses in conservation science (Guidetti, 2002; Ferraro and Pattanayak, 2006). Our approach  
266 to evaluate the temporal and spatial changes provides a more robust measure of reserve effectiveness, and  
267 captures previously described patterns. For example, the rapid increase observed for lobster densities in  
268 Isla Natividad on the sixth year (*i.e.* 2012; Fig. 2A), occurs a year after the hypoxia events described by  
269 Micheli et al. (2012), which caused mass mortality of sedentary organisms such as abalone and sea urchins,  
270 but not lobster and finfish. The use of causal inference techniques may help us support evidence-based  
271 conservation.

272 Our analyses of socioeconomic indicators has three limitations. First, we only look at landings and  
273 revenues by landings for communities with and without TURF-reserves. There are a number of other  
274 possible indicators that could show a change due to the implementation of the reserve. Notably, one often  
275 cited in the literature is additional benefits, such as tourism (Viana et al., 2017). However, it is unlikely  
276 that the evaluated communities will experience tourism benefits due to their remoteness and the lack of  
277 proper infrastructure to sustain tourism. A second limitation of our socioeconomic analysis is that we do  
278 not observe effort data, which may mask the effect of the reserve. For example, if catches remain relatively  
279 unchanged but fishing effort decreased, that would imply a larger catch per unit effort and thus higher  
280 profitability, provided that cost per unit effort does not increase. Likewise, it is possible that fishing effort  
281 increased around reserves to maintain the historical levels of landings. A final limitation applies to Maria  
282 Elena, where we only observe landings and income for one year after reserve implementation. While one  
283 would not expect to observe increased landings or income in such a short period, a spatial closure might  
284 cause total catches to decline, especially if effort is held constant.

285 A first possible explanation for the lack of effectiveness may be the young age of the reserves. Literature  
286 shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al.,  
287 2014; Babcock et al., 2010). Isla Natividad has the oldest reserves, and our SES analysis suggests that all  
288 communities have a well-established community-based enforcement system. With these characteristics,  
289 one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves  
290 (*i.e.* < 6 years old; RS5 in Table 2) and effects may not yet be evident due to the short duration of  
291 protection, relative to the life histories of the protected species; community-based marine reserves in  
292 tropical ecosystems may take six years or more to show a spillover effect (da Silva et al., 2015).

293 Another key condition for effectiveness is reserve size (Edgar et al., 2014), and the lack of effectiveness  
294 can perhaps be attributed to poor ecological coherence in reserve design (*sensu* Rees et al. (2018)). Previous  
295 research has shown that reserves in Isla Natividad yield fishery benefits for the abalone fishery (Rossetto  
296 et al., 2015), however, abalone are less mobile than lobsters, and perhaps the reserves provide enough  
297 protection to these sedentary invertebrates, but not lobsters. Design principles developed by Green et al.  
298 (2017) for marine reserves in the Caribbean state that reserves “should be more than twice the size of the  
299 home range of adults and juveniles”, and suggest that reserves seeking to protect spiny lobsters should have  
300 at least 14 km across. As shown through the SES analysis, the size of the marine reserves appears small  
301 compared to the movement capacity of the main targeted species (RU1, RS3; Table 2). Furthermore, fishers  
302 may favor implementation of reserves that pose low fishing costs due to their small size or location. Our  
303 analysis of economic data supports this hypothesis, as neither landings nor revenues showed the expected  
304 short-term reductions associated to the first years of reserve implementation (Ovando et al., 2016).

305 Even if reserves had appropriate sizes and were placed in optimal locations, there are other plausible  
306 explanations for the observed patterns. For instance, marine reserves are only likely to provide fisheries  
307 benefits if initial population sizes are low and the fishery is poorly managed (Hilborn et al., 2004, 2006).

308 Both lobster fisheries were certified by the Marine Stewardship Council and are managed via species-  
309 specific minimum catch sizes, seasonal closures, protection of “berried” females, and escapement windows  
310 where traps are allowed (DOF, 1993). It is uncertain whether such a well-managed fishery will experience  
311 additional benefits from marine reserves; reserves implemented in TURFs where fishing pressure is already  
312 optimally managed will still show a trade-off between fisheries and conservation objectives (Lester et al.,  
313 2017). Furthermore, Gelcich et al. (2008) have shown that TURFs alone can have greater biomass and  
314 richness than areas operating under open access. This might reduce the difference between indicators from  
315 the TURF and reserve sites, making it difficult to detect such a small change. Further research should focus  
316 on evaluating sites in the reserve, TURF, and open access areas or similar Fish Refuges established without  
317 the presence of TURFs where the impact of the reserves might be greater.

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

323 While the evaluated reserves have failed to provide fishery benefits to date, there are a number of  
324 additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range  
325 of species and vulnerable habitat. Previous research focusing on these specific sites has shown that they  
326 serve as an insurance mechanism against uncertainty and errors in fisheries management, as well as mild  
327 environmental shocks (Micheli et al., 2012; De Leo and Micheli, 2015; Roberts et al., 2017; Aalto et al.,  
328 0). Self-regulation of fishing effort can serve as a way to compensate for future declines associated to  
329 environmental variation (Finkbeiner et al., 2018). Furthermore, embarking on a marine conservation project  
330 can bring the community together, which promotes social cohesion and builds social capital (Fulton et al.,  
331 2019). Showing commitment to marine conservation and sustainable fishing practices has allowed fishers to  
332 have greater bargaining power and leverage over fisheries management (Pérez-Ramírez et al., 2012). These  
333 additional benefits might explain why communities show a positive perception about their performance  
334 and continue to support their presence by re-establishing the reserves (Ayer et al., 2018).

335 Community-based TURF-reserves in small-scale fisheries may be helpful conservation and fishery  
336 management tools when appropriately implemented (Gelcich and Donlan, 2015). We must promote  
337 bottom-up design and implementation processes like the ones in the evaluated reserves, but without setting  
338 design principles aside. Having full community support surely represents an advantage, but it is important  
339 that community-based TURF-reserves meet essential design principles such as size and placement so  
340 as to maximize their effectiveness. Furthermore, conservation and advocacy groups should consider the  
341 opportunity costs of such interventions (*sensu* Smith et al. (2010)) and evaluate the potential of other  
342 approaches that may yield similar benefits.

343 In terms of fisheries regulation in Mexico, our work only evaluates Fish Refuges established within  
344 TURFs. Future research should aim at evaluating other Fish Refuges established as bottom-up processes  
345 but without the presence of TURFs (e.g. DOF (2012a)), others established through top-down processes (*i.e.*  
346 DOF (2018a)), as well as the relationship between governance and effectiveness across this gradient of  
347 approaches. For the particular case of the reserves that we evaluate, the possibility of expanding reserves or  
348 merging existing polygons into larger areas should be evaluated and proposed to the communities.

## CONFLICT OF INTEREST STATEMENT

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

## AUTHOR CONTRIBUTIONS

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

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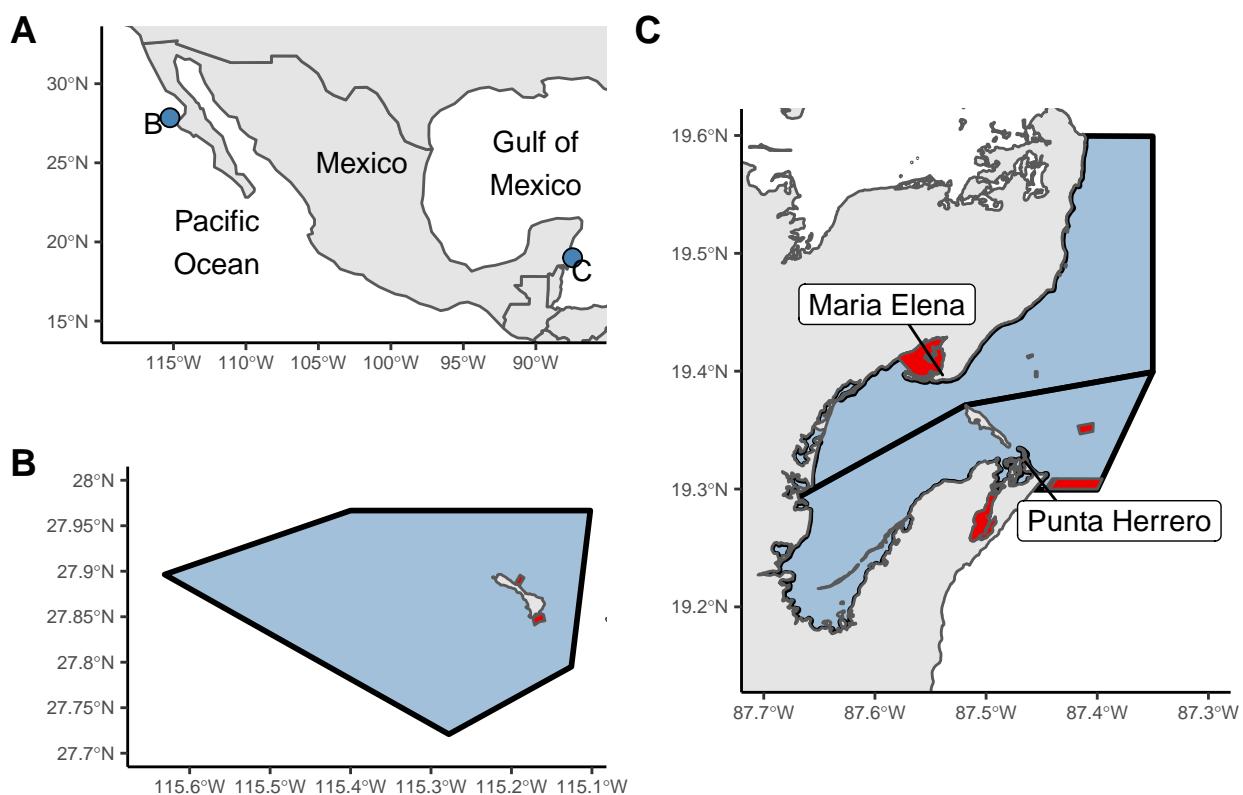
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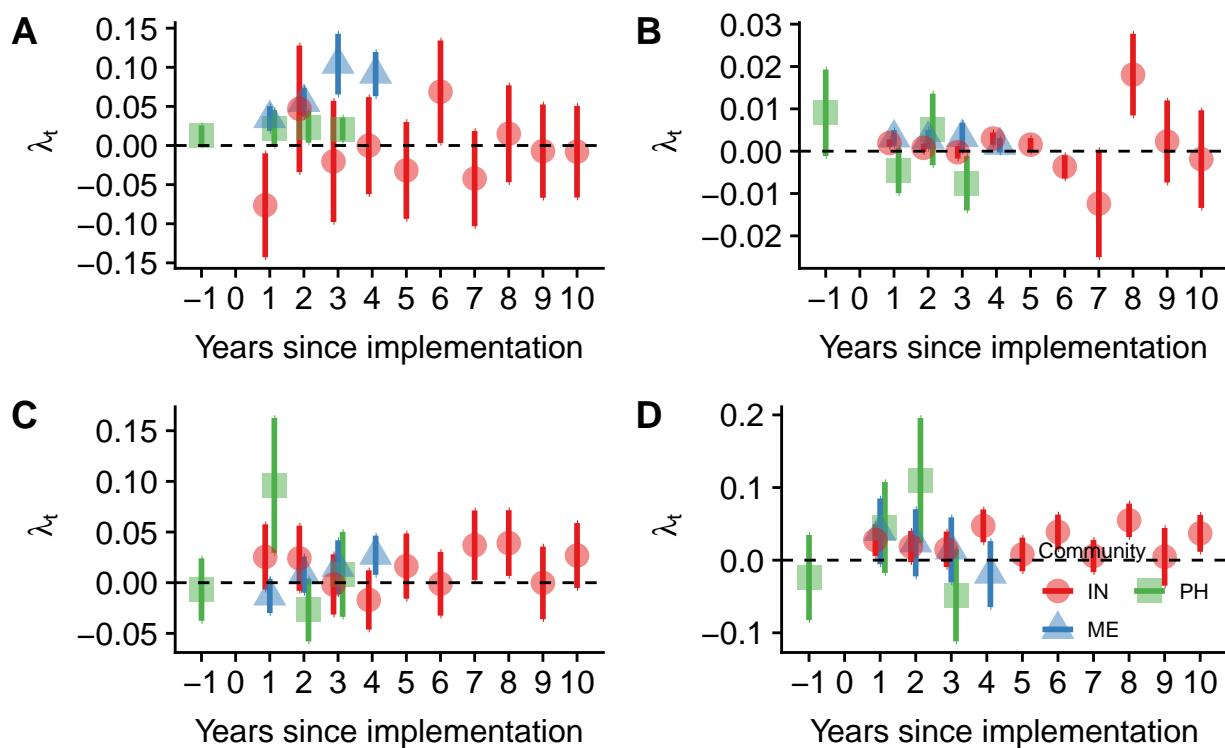
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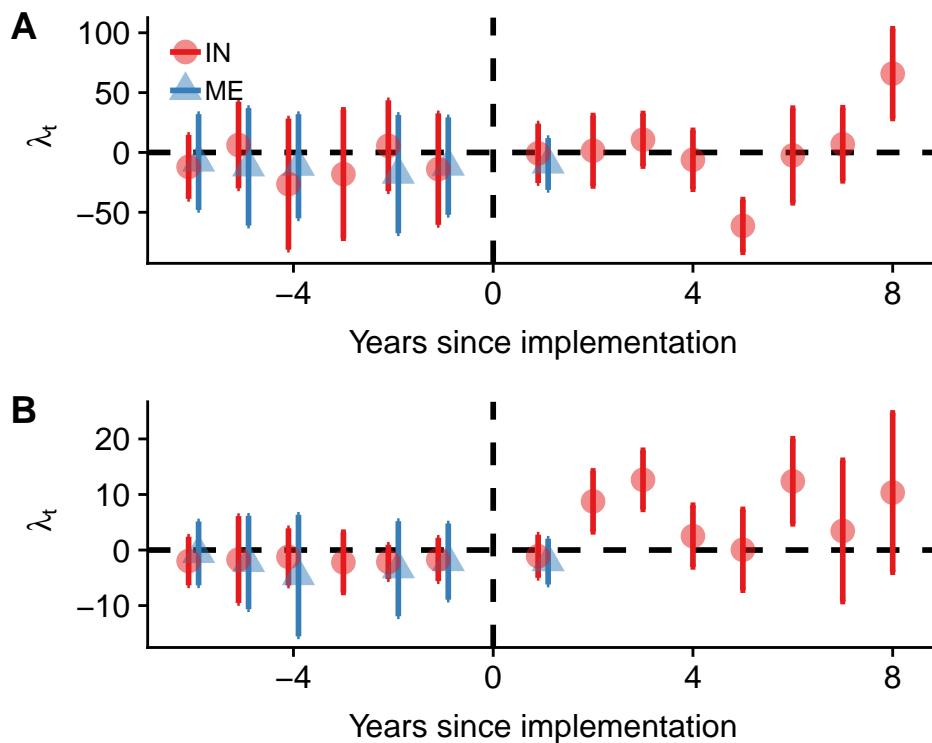
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**FIGURE CAPTIONS**

**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 at 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
<b>Biological</b>	
Lobster density	org m <sup>-2</sup>
Invertebrate density	org m <sup>-2</sup>
Fish density	org m <sup>-2</sup>
Fish biomass	Kg m <sup>-2</sup>
<b>Socioeconomic</b>	
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 = Maria Elena, PH = Punta Herrero). Alphanumeric codes follow Basurto et al. (2013); an asterisk (\*) denotes variables incorporated based on Di Franco et al. (2016) and Edgar et al. (2014). The presented narrative applies equally for all communities unless otherwise noted.

Variable	Narrative
<b>Resource System (RS)</b>	
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 <sup>2</sup> )	IN = 889.5; ME = 353.1; PH = 299.7
RS3 - Size of resource system: Reserve area (Evaluated reserve area; Km <sup>2</sup> )	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
<b>Resource Unit (RU)</b>	
RU1 - Resource unit mobility	Adult spiny lobsters can move between 1 and 10 Km, while larvae can have displacements in the order of hundreds of Km (Aceves-Bueno et al., 2017; Green et al., 2017).
RU5 - Number of units (catch diversity): Number of targeted species	Lobster is their main fishery of these three communities, but they also target finfish (2 spp each). Additionally, fishers from Isla Natividad target other sedentary benthic invertebrates (4 spp).
<b>Actors (A)</b>	
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. IN lies 545 Km south from Tijuana, and ME and PH 230 Km south from Cancun, where the nearest international airports are located.
<b>Governance system (G)</b>	
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 are: 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 (Consejo de vigilancia) 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.