

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 communities
4 around the world. Small-scale fisheries face great challenges since they are difficult to monitor,
5 enforce, and manage. Combining territorial use 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 Mexican TURF-reserves gained legal recognition thanks to a regulation passed
10 in 2014; their effectiveness has not been formally evaluated. We combine causal inference
11 techniques and the Social-Ecological Systems framework to provide a holistic evaluation of
12 community-based TURF-reserves in three coastal communities in Mexico. We find that while
13 reserves have not yet achieved their stated goal of increasing the density of lobster and other
14 benthic invertebrates, they continue to receive support from the fishing communities. A lack of
15 clear ecological and socioeconomic effects likely results from a combination of factors. First,
16 some of these reserves might be too young for the effects to show. Second, the reserves are not
17 large enough to protect mobile species, like lobster. Third, variable and extreme oceanographic
18 conditions have impacted harvested populations. Fourth, local fisheries are already well managed,
19 and it is unlikely that reserves might have a detectable effect in catches. However, these reserves
20 may provide a foundation for establishing additional, larger marine reserves needed to effectively
21 conserve 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 (Pauly et al., 2005; Worm et al., 2006; Halpern et al., 2008). 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 approaches taken 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 (*e.g.* fishing cooperatives)
32 have exclusive access to an explicitly delimited portion of the ocean. They promote a sense of stewardship
33 and incentivise resource users to sustainably manage their resources (Gelcich et al., 2008; Costello and
34 Kaffine, 2010; McCay et al., 2014). On the other hand, no-take marine reserves (marine reserves from
35 hereinafter) are areas where all extractive activities are off-limits. These can be implemented to protect
36 biodiversity but also as fishery management tools to aid in the recovery of marine stocks. These instruments
37 can be combined by establishing a marine reserve within a TURF, thus making them TURF-reserves
38 (Afflerbach et al., 2014; Gelcich and Donlan, 2015; Lester et al., 2017).

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

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

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

65 the effectiveness of community-based TURF-reserves in small-scale fisheries where this tool is used to
66 manage and rebuild their coastal fisheries.

2 METHODS

67 2.1 TURF-reserves in Mexico

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

77 In an effort to bridge this normative gap, Mexican Civil Society Organizations (CSOs) served as a link
78 between fishers and government, and created a legal framework that solves this governance issue: Fish
79 Refuges (NOM-049-SAG/PESC, 2014). Fish Refuges can be implemented as temporal or partial reserves,
80 which can protect one, some, or all resources within their boundaries. One of the ways in which fishing
81 communities have taken advantage of this new tool is by implementing temporal marine reserves within
82 their TURFs with a defined expiration date (often 5 years). When the expiration date is reached, fishers can
83 chose to open the reserves to fishing or re-establish them. Our work focuses on Fish Refuges implemented
84 as community-based TURF-reserves that occur in small-scale fisheries.

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

93 2.2 Study areas

94 We evaluate three community-based no-take TURF-reserves implemented in Mexican TURF-managed
95 fisheries, therefore making them TURF-reserves (Fig 1A). The first one was created by the *Buzos y*
96 *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, red sea urchin, snail, and abalone are also an important source of
99 income. In 2006, the community decided to implement two marine reserves within their fishing grounds.
100 The objective of these reserves was “to protect commercially important invertebrate species”; mainly
101 lobster and abalone. These reserves obtained legal recognition in 2018 (DOF, 2018b).

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

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

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

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

132 2.3 Data collection

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

143 The biological data are collected by members from each community and personnel from the Mexican
144 CSO *Comunidad y Biodiversidad* (COBI). Trained divers record species richness and abundances of fish
145 and invertebrate species along replicate transects (30 × 2 m each) at depths 5–20 m in the reserves and
146 control sites (Suman et al., 2010; Fulton et al., 2018, 2019). Size structures are also collected during fish

147 surveys. All sites were surveyed annually, and at least once before implementation of the reserves. A
 148 summary of sampling effort is shown in the supplementary materials (Tables S1-S2).

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

159 Data for the evaluation of the SES were collected at the community-level from official documents used
 160 in the design, creation, and designation of the marine reserves. These include the technical studies that
 161 the cooperatives submit when they request recognition of their reserves, as well as the official enactments
 162 (DOF, 2012b, 2013, 2018b). We also complimented information based on the authors' experience and
 163 knowledge of the communities. We collected information on the Resource Systems, Resource Units, Actors,
 164 and Governance System (Table 2). The next section further describe the variables and indicators chosen for
 165 each of these.

166 2.4 Data analysis

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

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

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

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

184 Socioeconomic indicators are evaluated with a similar approach. Due to data constraints, we only
 185 evaluate socioeconomic data for Isla Natividad (2000 - 2014) and Maria Elena (2006 - 2013). Neighboring

186 communities are used as counterfactuals that allow us to control for unobserved time-invariants. Each focal
 187 community (*i.e.* Isla Natividad and Maria Elena) has three counterfactual communities.

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

188 The coefficient interpretations remains as for Eq. 1, but in this case the *Treated* dummy variable indicates
 189 if the community has a reserve (*Treated* = 1) or not (*Treated* = 0). These regression models allow us to
 190 establish a causal link between the implementation of marine reserves and the observed trends by accounting
 191 for temporal and site-specific dynamics (De Palma et al., 2018). We fit one model per community and
 192 indicators (*e.g.* there are three models for lobster density, one for each community) for a total of 12 biological
 193 model fits and four socioeconomic model fits. Model coefficients were estimated via ordinary least-squares
 194 and used heteroskedastic-robust standard errors (Zeileis, 2004). All analyses were performed in R version
 195 3.5.1 and R Studio version 1.1.456 (R Core Team, 2018). All data and code needed to reproduce our analyses
 196 are available in a GitHub repository at: <https://github.com/jcvdav/ReserveEffect>.

197 We use the SES framework to evaluate each community and create a narrative that provides context for
 198 each community. The use of this framework standardizes our analysis and allows us to communicate our
 199 results in a common language across fields by using a set of previously defined variables and indicators. We
 200 based our variable selection primarily on Leslie et al. (2015) and Basurto et al. (2013), who operationalized
 201 and analyzed Mexican fishing cooperatives using this framework, and identified the key variables relevant
 202 to fishing cooperatives in Mexico. We also incorporate other relevant variables known to influence reserve
 203 performance following Di Franco et al. (2016) and Edgar et al. (2014). Table 2 shows the selected variables,
 204 along with definitions and values.

3 RESULTS

205 The following sections present the effect that marine reserves had on each of the biological and socioeco-
 206 nomic indicators for each coastal community. Results are presented in terms of the difference through time
 207 and across sites, relative to the control site on the year of implementation (*i.e.* the difference-in-differences
 208 estimate or effect size λ_t from Eqs. 1 and 2). We also provide an overview of the governance settings
 209 of each community, and discuss how these might be related to the effectiveness and performance of the
 210 reserves.

211 3.1 Biological effects

212 Indicators showed ambiguous responses through time for each reserve. Figure 2A shows positive effect
 213 sizes for lobster densities in Isla Natividad and Punta Herrero during the first years, but the effect is eroded
 214 through time. In the case of Maria Elena, positive changes were observed in the third and fourth year.
 215 These effects are in the order of 0.2 extra organisms m^{-2} for Isla Natividad and Punta Herrero, and 0.01
 216 organisms m^{-2} for Maria Elena, but are not significantly different from zero ($p > 0.05$). Likewise, no
 217 significant changes were detected in fish biomass or invertebrate and fish densities (Fig. 2B-D), where
 218 effect sizes oscillated around zero without clear trends. Figures and tables with time series of indicators
 219 and model coefficients are presented in the supplementary materials (Figures S1-S4, Tables S4-S6).

220 3.2 Socioeconomic effects

221 Lobster landings and revenue were only available for Isla Natividad and Maria Elena (Fig 3). For all years
 222 before implementation, the effect sizes are close to zero, indicating that the control and treatment sites

have similar pre-treatment trends, suggesting that these are plausible controls. However, effect sizes do not change after the implementation of the reserve. Interestingly, the negative effect observed for Isla Natividad on year 5 correspond to the 2011 hypoxia events. The only positive change observed in lobster landings is for Isla Natividad in 2014 ($p < 0.1$). The three years of post-implementation data for Maria Elena do not show a significant effect of the reserve. Isla Natividad shows higher revenues after the implementation of the reserve, as compared to the control communities. However, these changes are only significant for the third year ($p < 0.05$). Full tables with model coefficients are presented in the supplementary materials (Tables S4-S5).

3.3 Governance

Our analysis of the SES (Table 2) shows that all analyzed communities share similarities known to foster sustainable resource management and increase reserve effectiveness. For example, fishers operate within clearly outlined TURFs (RS2, GS6.1.4.3) that provide exclusive access to resources and reserves. Along with their relatively small groups (A1 - Number of relevant actors), Isolation (A3), Operational rules (GS6.2), Social monitoring (GS9.1), and Graduated sanctions (GS10.1), these fisheries have solid governance structures that enable them to monitor their resources and enforce rules to ensure sustainable management. In general, success of conservation initiatives depends on the incentives of local communities to maintain a healthy status of the resources upon which they depend (Jupiter et al., 2017). Due to the clarity of access rights and isolation, the benefits of conservation directly benefit the members of the fishing cooperatives, which have favored the development of efficient community-based enforcement systems. However, our SES analysis also highlights factors that might hinder reserve performance or mask outcomes. While total reserve size ranges from 0.2% to 3.7% of the TURF area, individual reserves are often small (RS3); the largest reserve is only 4.37 km², and the smallest one is 0.09 km². Reserves are also relatively young (RS5). Additionally, fishers harvest healthy stocks (RS4.1), and it's unlikely that marine reserves will result in increased catches.

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 aside from 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. Lastly, the communities exhibit all the social enabling conditions for effective reserve and resource management. Here we discuss possible shortcomings in our analyses as well as possible explanations for the observed patterns.

While many ecology studies have used BACI sampling designs and respective analyses (e.g. Stewart-Oaten et al. (1986)), few conservation studies have done so to evaluate the effect of an intervention (e.g. Francini-Filho and Moura (2008); Lester et al. (2009); Moland et al. (2013)) which has resulted in a call for more robust analyses in conservation science (Guidetti, 2002; Ferraro and Pattanayak, 2006). Our approach to evaluate the temporal and spatial changes provides a more robust measure of reserve effectiveness, and captures previously described patterns. For example, the rapid increase observed for lobster densities in Isla Natividad on the sixth year (*i.e.* 2012; Fig. 2A), occurs 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. The use of causal inference techniques may help us support evidence-based conservation.

Our analyses of socioeconomic indicators has two limitations. First, we only look at landings and revenues by landings for communities with and without TURF-reserves. There are a number of other possible indicators that could show a change due to the implementation of the reserve. Notably, one often cited in the literature is additional benefits, such as tourism (Viana et al., 2017). However, it is unlikely that the evaluated communities will experience tourism benefits due to their remoteness and the lack of proper infrastructure to sustain tourism. A second limitation of our socioeconomic analysis is that we do not observe effort data, which may mask the effect of the reserve. For example, if catches remain relatively unchanged but fishing effort decreased, that would imply a larger catch per unit effort and thus higher profitability.

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

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

Even if reserves had appropriate sizes and were placed in optimal locations, there are other plausible explanations for the observed patterns. For instance, marine reserves are only likely to provide fisheries benefits if initial population sizes are low and the fishery is poorly managed (Hilborn et al., 2004, 2006). Both lobster fisheries were certified by the Marine Stewardship Council and are managed via species-specific minimum catch sizes, seasonal closures, protection of “berried” females, and escapement windows where traps are allowed (DOF, 1993). It is uncertain whether such a well-managed fishery will experience additional benefits from marine reserves; reserves implemented in TURFs where fishing pressure is already optimally managed will still show a trade-off between fisheries and conservation objectives (Lester et al., 2017). Furthermore, Gelcich et al. (2008) have shown that TURFs alone can have greater biomass and richness than areas operating under open access. This might reduce the difference between indicators from the TURF and reserve sites, making it difficult to detect such a small change. Further research should focus on evaluating sites in the reserve, TURF, and open access areas or similar Fish Refuges established without the presence of TURFs where the impact of the reserves might be greater.

Finally, extreme conditions, including prolonged hypoxia, heat waves, and storms have affected both the Pacific and Caribbean regions, with large negative impacts of coastal marine species and ecosystems (Cavole et al., 2016; Hughes et al., 2018; Breitburg et al., 2018). The coastal ecosystems where these

308 reserves are located have been profoundly affected by these events (Micheli et al., 2012; Woodson et al.,
309 2018). Effects of protection might be eliminated by the mortalities associated with these extreme conditions.

310 While the evaluated reserves have failed to provide fishery benefits up to now, there are a number of
311 additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range
312 of species and vulnerable habitat. Previous research focusing on these specific sites has shown that they
313 serve as an insurance mechanism against uncertainty and errors in fisheries management, as well as mild
314 environmental shocks (Micheli et al., 2012; De Leo and Micheli, 2015; Roberts et al., 2017; Aalto et al., in
315 press). Self-regulation of fishing effort can serve as a way to compensate for future declines associated
316 to environmental variation (Finkbeiner et al., 2018). Furthermore, embarking in a marine conservation
317 project can bring the community together, which promotes social cohesion and builds social capital (Fulton
318 et al., 2019). Showing commitment to marine conservation and sustainable fishing practices has allowed
319 fishers to have greater bargaining power and leverage over fisheries management (Pérez-Ramírez et al.,
320 2012). These additional benefits might explain why communities show a positive perception about their
321 performance and continue to support their presence by re-implementing the reserves (Ayer et al., 2018).

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

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

CONFLICT OF INTEREST STATEMENT

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

AUTHOR CONTRIBUTIONS

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

FUNDING

341 JCVD received funding from UCMexus - CONACyT Doctoral Fellowship (CVU 669403) and the Latin
342 American Fisheries Fellowship Program. AS, AHV, SF and JT received funding from Marisla Foundation,

343 Packard Foundation, Walton Family Foundation, Summit Foundation, and Oak Foundation. FM was
344 supported by NSF-CNH and NSF BioOce (grants DEB-1212124 and 1736830).

ACKNOWLEDGMENTS

345 The authors wish to acknowledge Imelda Amador for contributions on the governance data, as well as
346 pre-processing biological data. This study would have not been possible without the effort by members
347 of the fishing communities here mentioned, who participated in the data-collection process. The authors
348 wish to acknowledge comments by the reviewrs and editor, which significantly improved the quality of this
349 work.

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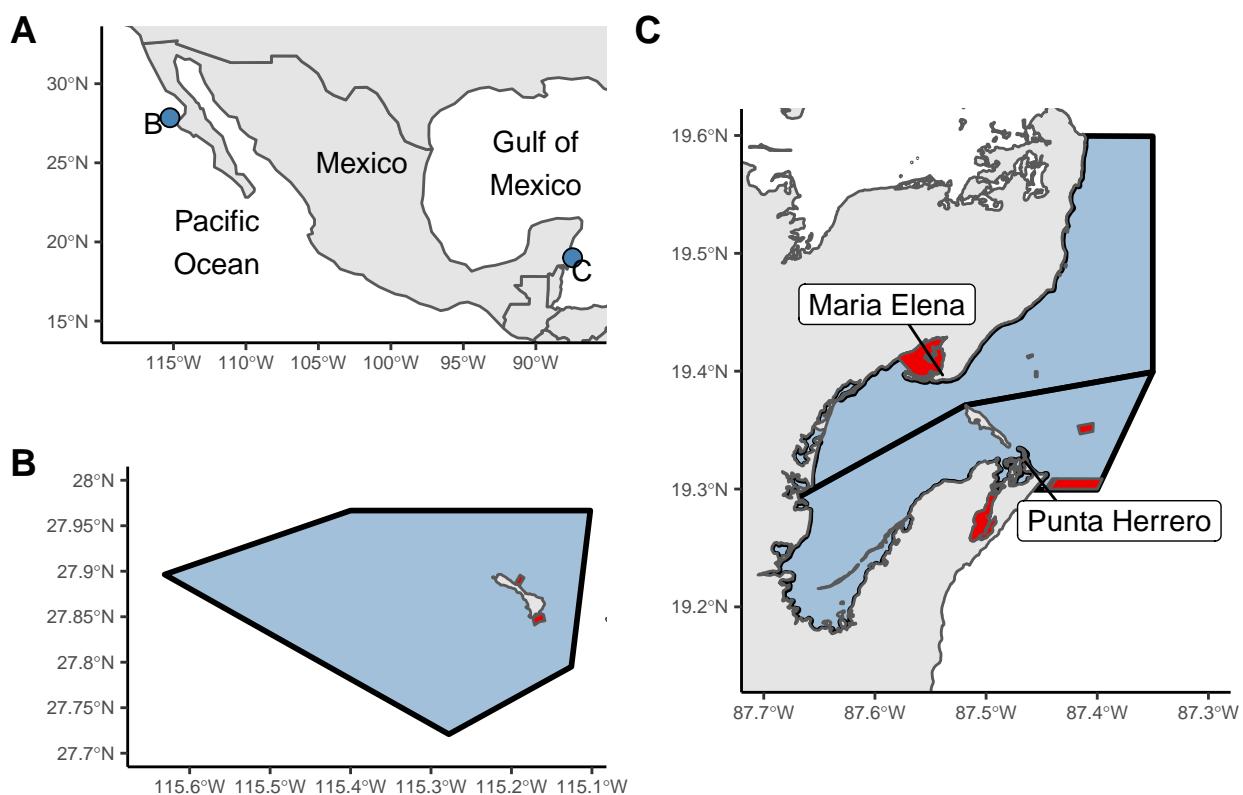
FIGURE CAPTIONS

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.

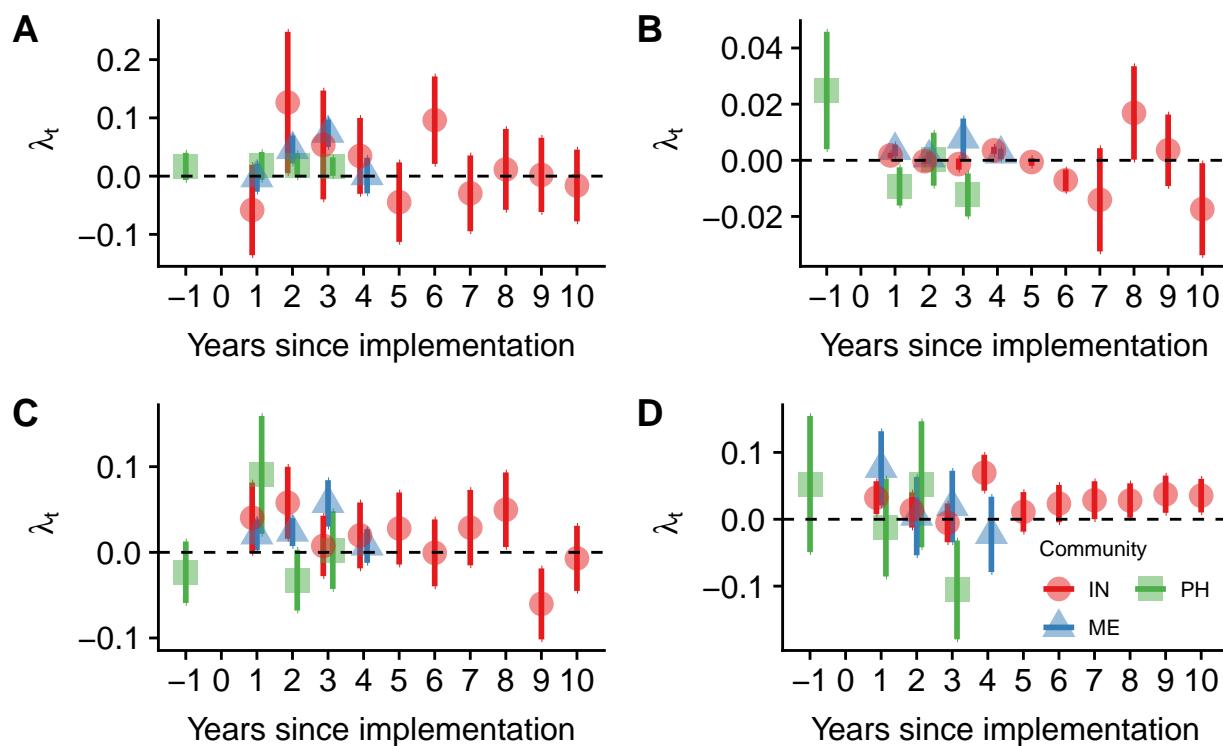


Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red circles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

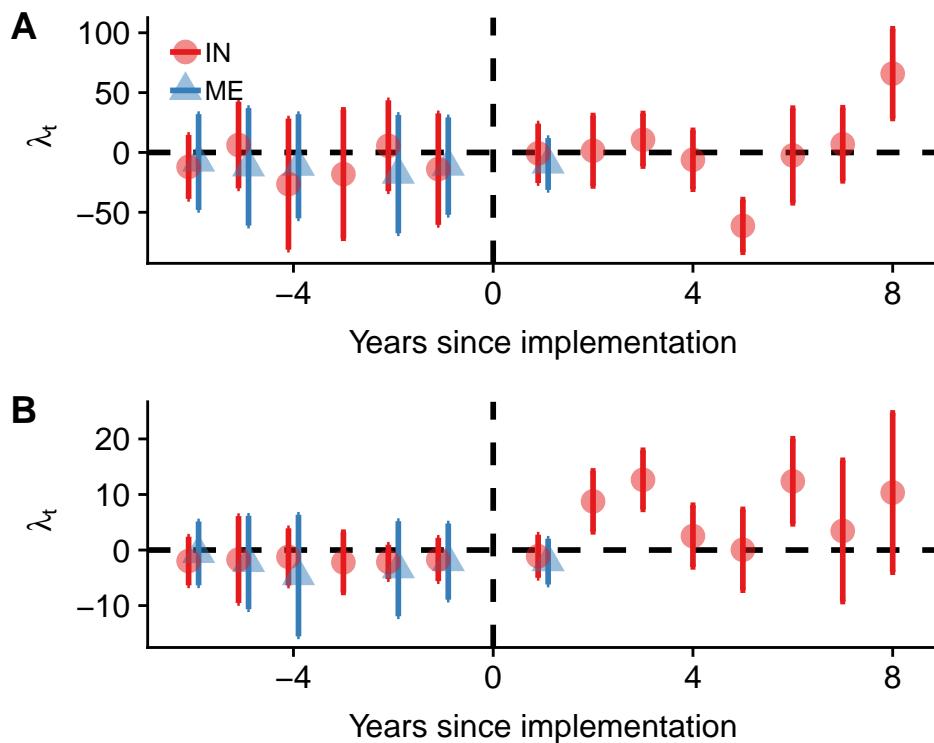


Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points are jittered horizontally to avoid overplotting. Points indicate the effect size and error bars are heteroskedastic-robust standard errors. Years have been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

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

Indicator	Units
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 = 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
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 (2 spp each). Additionally, fishers from Isla Natividad target other sedentary benthic invertebrates (4 spp).
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. IN lies 545 Km south from Tijuana, and ME and PH 230 Km south from Cancun, where the nearest international airports are located.
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 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.