

Effectiveness of community-based marine reserves in small-scale fisheries

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2 ABSTRACT

3 Coastal marine ecosystems provide livelihoods for small-scale fishers and coastal communities
4 around the world. Artisanal fisheries face great challenges since they are difficult to monitor,
5 enforce, and manage. Combining territorial user rights for fisheries (TURF) with no-take marine
6 reserves to create TURF-reserves is believed to improve the performance of small-scale fisheries
7 by buffering fisheries from environmental variability and management errors, while ensuring
8 that fishers reap the benefits of conservation investments. In the last six years, and following a
9 2012 regulation, 18 TURF-reserves have been implemented in Mexico; their effectiveness has
10 not been formally evaluated. We combine causal inference techniques and a social-ecological
11 systems framework to provide a holistic evaluation of community-based TURF reserves in three
12 coastal communities in Mexico. We find that while reserves have not yet achieved their stated
13 goal of increasing lobster densities, they continue to receive significant support from the fishing
14 communities. A lack of ecological and socioeconomic effects likely results from a combination of
15 factors. First, the lobster fisheries are already well managed, and it is unlikely that reserves might
16 have a detectable effect. Second, some of the reserves are not large enough to protect lobsters'
17 home ranges. Third, some of these reserves might be too young for the effects to show. However,
18 these reserves have shaped small-scale fishers' way of thinking about marine conservation,
19 which can provide a foundation for establishing additional, larger marine reserves needed to
20 effectively conserve mobile species.

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

1 INTRODUCTION

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

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

TURF-reserves can be created as community-based marine reserves, voluntarily established and enforced by local communities. This bottom-up approach increases compliance and self-enforcement (Gelcich and Donlan, 2015; Espinosa-Romero et al., 2014; Beger et al., 2004). Community-based spatial closures occur in different contexts, like the *kapu* or *ra'ui* areas in the Pacific Islands (Bohnsack et al., 2004; Johannes, 2002). However, without legal recognition no-take regulations are difficult to enforce and fishers rely on the exclusive access granted by the TURF. In an effort to bridge this normative gap, Civil Society Organizations (CSOs) have served as a link between fishers and government, and set out to create a legal framework that solve this governance issue. In Mexico, a new norm was created in 2014 allowing fishers to request the legal recognition of community-based reserves as “Fish Refuge” (*Zona de Refugio Pesquero*; NOM-049-SAG/PESC (2014)). Fish refuges can be implemented as temporal or partial reserves, which can protect one, some, or all resources within their boundaries. Since 2012, 45 of Fish Refuges have been created along the Pacific, Gulf of California, and Mexican Caribbean coastlines, with 18 of them implemented as TURF-reserves. However, their effectiveness has not yet been formally evaluated and reported in the scientific literature.

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

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

2 METHODS

70 2.1 Study area

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

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

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

97 2.2 Data collection

98 We use three main sources of information to evaluate these reserves across the ecological, socioeconomic,
99 and governance dimensions. Ecological data come from the annual ecological monitoring of reserve
100 and control areas, carried out by members from each community and personnel from the Mexican CSO
101 *Comunidad y Biodiversidad* (COBI). Trained divers record richness and abundances of fish and invertebrate
102 species along replicate transects (30x 2 m each) at depths 5-20 m in the reserves and control sites (Fulton
103 et al., 2018, 2019). Size structures are also collected during fish surveys. We define control sites as
104 regions with habitat characteristics similar to the corresponding reserves, and that presumably had a similar
105 probability of being selected as reserves during the design phase. We focus our evaluation on sites where

106 data are available for reserve and control sites, before and after the implementation of the reserve. This
 107 provides us with a Before-After-Control-Impact (*i.e.* BACI) sampling design that allows us to capture and
 108 control for temporal and spatial dynamics (De Palma et al., 2018; Ferraro and Pattanayak, 2006). BACI
 109 designs and causal inference techniques have proven effective to evaluate marine reserves, as they allow us
 110 to causally attribute observed changes to the intervention (Moland et al., 2013; Villaseñor-Derbez et al.,
 111 2018). All sites were surveyed annually, and at least once before implementation of the reserves.

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

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

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

122 Data for the operationalization of the social-ecological system were collected at the community-level
 123 from official documents used in the creation and designation of the marine reserves (DOF, 2012b, 2013,
 124 2018b) and based on the authors' experience and knowledge of the communities. These include information
 125 on the resource system, the resource units, actors, and the governance system itself (Table 2).

126 2.3 Data analysis

127 We evaluate the effect that marine reserves have had on four ecological and two socioeconomic indicators
 128 (Table 1). Recall that reserves were implemented to protect lobster and other benthic invertebrates. However,
 129 we also use the available fish data to test for associated co-benefits.

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

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

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

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

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

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

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

153 We use the social-ecological system (SES) framework to evaluate each community as a means of providing
 154 an explanation to the biological and socioeconomic results. We use the SES framework standardizes
 155 our analysis and allows us to communicate our results in a common language across fields. We based our
 156 variable selection primarily on Leslie et al. (2015); Basurto et al. (2013), who have previously analyzed
 157 Mexican fishing cooperatives using this framework. We also incorporate other relevant information following
 158 Di Franco et al. (2016) and Edgar et al. (2014). Table 2 shows the selected variables, their definition
 159 and selected indicators.

3 RESULTS

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

165 3.1 Biological effects

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

174 3.2 Socioeconomic effects

175 Lobster landings and revenue were only available for Isla Natividad and Maria Elena (Fig 3). For all years
 176 before implementation, the effect sizes are close to zero, indicating that the control and treatment sites
 177 have similar pre-treatment trends, suggesting that these are plausible controls. However, effect sizes do not

178 change after the implementation of the reserve. Interestingly, the negative effect observed for Isla Natividad
179 on year 5 correspond to the 2011 hypoxia events. The only positive change observed in lobster landings is
180 for Isla Natividad in 2014 ($p < 0.1$). The three years of post-implementation data for Maria Elena do not
181 show a significant effect of the reserve. Isla Natividad shows higher revenues after the implementation of
182 the reserve, as compared to the control communities. However, these changes are not significant and are
183 associated to increased variation. Full tables with model coefficients are presented in the supplementary
184 materials (S4 Table, S5 Table).

185 3.3 Governance

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

4 DISCUSSION

200 Our results indicate that these TURF-reserves have not increased lobster densities. Additionally, no
201 co-benefits were identified when using other ecological indicators other than the previously reported
202 buffering effect that reserves can have to environmental variability in Isla Natividad (Micheli et al., 2012).
203 The socioeconomic indicators pertaining landings and revenues showed little to no change after reserve
204 implementation. The coastal ecosystems where these reserves are located have been profoundly affected by
205 climatic and oceanographic extremes, including warming events, extreme storms and prolonged hypoxia
206 (Micheli et al., 2012; Woodson et al., in press). Despite the lack of evidence of the effectiveness of these
207 reserves, most of the communities show a positive perception about their performance and continue to
208 support their presence (Ayer et al., 2018). Understanding the social-ecological context in which these
209 communities and their reserves operate might provide insights to this.

210 Some works evaluate marine reserves by performing inside-outside (Guidetti et al., 2014; Friedlander
211 et al., 2017; Rodriguez and Fanning, 2017) or before-after comparisons (Betti et al., 2017). The first
212 approach does not address temporal variability, and the second can not distinguish between the temporal
213 trends in a reserve and the entire system (De Palma et al., 2018). Our approach to evaluate the temporal
214 and spatial changes provides a more robust measure of reserve effectiveness. For example, we capture
215 previously described patterns like the rapid increase observed for lobster densities in Isla Natividad on the
216 sixth year (*i.e.* 2012; Fig. 2A) -a year after the hypoxia events described by Micheli et al. (2012)- which
217 caused mass mortality of sedentary organisms such as abalone and sea urchins, but not lobster and finfish.
218 Yet, our empirical approach assumes control sites are a plausible counterfactual for treated sites. This

219 implies that treated sites would have followed the same trend as control sites, had the reserves not been
220 implemented. Nonetheless, temporal trends for each site don't show any significant increases (S1 Table, S2
221 Table, S3 Table), supporting our findings of lack of change in the indicators used.

222 A first possible explanation for the lack of effectiveness may be the young age of the reserves. Literature
223 shows that age and enforcement are important factors that influence reserve effectiveness (Edgar et al.,
224 2014; Babcock et al., 2010). Isla Natividad has the oldest reserves, and our SES analysis suggests that all
225 communities have a well-established community-based enforcement system. With these characteristics,
226 one would expect the reserves to be effective. Maria Elena and Punta Herrero are relatively young reserves
227 (*i.e.* < 6 years old) and effects may not yet be evident due to the short duration of protection, relative to the
228 life histories of the protected species; other community-based marine reserves in tropical ecosystems may
229 take up to six years to show a spillover effect (da Silva et al., 2015).

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

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

253 While the evaluated reserves have failed to provide fishery benefits up to now, there are a number of
254 additional ecological, fisheries, and social benefits. Marine reserves provide protection to a wider range of
255 species and vulnerable habitat, like coral reefs. These sites can serve as an insurance against uncertainty
256 and errors in fisheries management, as well as environmental shocks (Hilborn et al., 2004, 2006; Micheli
257 et al., 2012; Aalto et al., in press). Self-regulation of fishing effort (*i.e.* reduction in harvest) can serve as
258 a way to compensate for future declines associated to environmental variation (Finkbeiner et al., 2018).
259 Embarking in a marine conservation project can bring the community together, which promotes social
260 cohesion and builds social capital (Fulton et al., 2019). Showing commitment to marine conservation and
261 sustainable fishing practices allows fishers to have greater bargaining power and leverage over fisheries
262 management (Pérez-Ramírez et al., 2012). Furthermore, the lack of effectiveness observed in these reserves

263 may not be generalizable to other reserves established under the same legal frameowrk (*i.e.* Fish Refges)
264 in Mexico, and future research should aim at evaluating other areas that have also been established as
265 bottom-up processes bt without the presence of TURFs (*e.g.* DOF (2012a)), or others established through a
266 top-down process (*i.e.* DOF (2018a)).

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

CONFLICT OF INTEREST STATEMENT

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

AUTHOR CONTRIBUTIONS

279 JC and EA analyzed and interpreted data, discussed the results, and wrote the first draft. AHV coordinated
280 fieldwork and collected the data. AS, AHV, SF, JT, and FM discussed the results and edited the manuscript.

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REFERENCES

- 287 Aalto, E., Micheli, F., Boch, C., Espinoza-Montes, A., Woodson, C., and De Leo, G. (in press). Marine
288 protected areas lower risk of abalone fishery collapse following widespread catastrophic mortality events.
289 *American Naturalist*
- 290 Aceves-Bueno, E., Cornejo-Donoso, J., Miller, S. J., and Gaines, S. D. (2017). Are territorial use rights in
291 fisheries (TURFs) sufficiently large? *Marine Policy* 78, 189–195. doi:10.1016/j.marpol.2017.01.024
- 292 Afflerbach, J. C., Lester, S. E., Dougherty, D. T., and Poon, S. E. (2014). A global survey of turf-reserves,
293 territorial use rights for fisheries coupled with marine reserves. *Global Ecology and Conservation* 2,
294 97–106. doi:10.1016/j.gecco.2014.08.001
- 295 Arafeh-Dalmau, N., Torres-Moye, G., Seingier, G., Montaño-Moctezuma, G., and Micheli, F. (2017).
296 Marine spatial planning in a transboundary context: Linking baja california with califonia's network of
297 marine protected areas. *Front Mar Sci* 4. doi:10.3389/fmars.2017.00150
- 298 Ayer, A., Fulton, S., Caamal-Madrigal, J. A., and Espinoza-Tenorio, A. (2018). Halfway to sustainability:
299 Management lessons from community-based, marine no-take zones in the mexican caribbean. *Marine
300 Policy* 93, 22–30. doi:10.1016/j.marpol.2018.03.008
- 301 Babcock, R. C., Shears, N. T., Alcalá, A. C., Barrett, N. S., Edgar, G. J., Lafferty, K. D., et al. (2010).
302 Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proc
303 Natl Acad Sci USA* 107, 18256–18261. doi:10.1073/pnas.0908012107
- 304 Basurto, X., Gelcich, S., and Ostrom, E. (2013). The social–ecological system framework as a knowledge
305 classificatory system for benthic small-scale fisheries. *Global Environmental Change* 23, 1366–1380.
306 doi:10.1016/j.gloenvcha.2013.08.001
- 307 Beger, M., Harborne, A. R., Dacles, T. P., Solandt, J.-L., and Ledesma, G. L. (2004). A framework of
308 lessons learned from community-based marine reserves and its effectiveness in guiding a new coastal
309 management initiative in the philippines. *Environ Manage* 34, 786–801. doi:10.1007/s00267-004-0149-z
- 310 Betti, F., Bavestrello, G., Bo, M., Asnaghi, V., Chiantore, M., Bava, S., et al. (2017). Over 10 years of
311 variation in mediterranean reef benthic communities. *Marine Ecology* 38, e12439. doi:10.1111/maec.
312 12439
- 313 Bohnsack, J. A., Ault, J. S., and Causey, B. (2004). Why have no-take marine protected areas? In *American
314 Fisheries Society Symposium*. vol. 42, 185–193
- 315 Costello, C. and Kaffine, D. T. (2010). Marine protected areas in spatial property-rights fisheries*.
316 *Australian Journal of Agricultural and Resource Economics* 54, 321–341. doi:10.1111/j.1467-8489.
317 2010.00495.x
- 318 Costello, C., Ovando, D., Hilborn, R., Gaines, S. D., Deschenes, O., and Lester, S. E. (2012). Status and
319 solutions for the world's unassessed fisheries. *Science* 338, 517–520. doi:10.1126/science.1223389
- 320 da Silva, I. M., Hill, N., Shimadzu, H., Soares, A. M. V. M., and Dornelas, M. (2015). Spillover effects of
321 a community-managed marine reserve. *PLoS ONE* 10, e0111774. doi:10.1371/journal.pone.0111774
- 322 De Leo, G. A. and Micheli, F. (2015). The good, the bad and the ugly of marine reserves for fishery yields.
323 *Philos Trans R Soc Lond, B, Biol Sci* 370. doi:10.1098/rstb.2014.0276
- 324 De Palma, A., Sanchez Ortiz, K., Martin, P. A., Chadwick, A., Gilbert, G., Bates, A. E., et al. (2018).
325 Challenges with inferring how land-use affects terrestrial biodiversity: Study design, time, space and
326 synthesis. *Advances in ecological research* doi:10.1016/bs.aecr.2017.12.004
- 327 Di Franco, A., Thiriet, P., Di Carlo, G., Dimitriadis, C., Francour, P., Gutiérrez, N. L., et al. (2016). Five
328 key attributes can increase marine protected areas performance for small-scale fisheries management.
329 *Sci Rep* 6, 38135. doi:10.1038/srep38135

- 330 [Dataset] DOF, D. (1993). Norma oficial mexicana 006-pesc-1993, para regular el aprovechamiento de
331 todas las especies de langosta en las aguas de jurisdiccion federal del golfo de mexico y mar caribe, asi
332 como del oceano pacifico incluyendo el golfo de california
- 333 DOF, D. (2012a). Acuerdo por el que se establece una red de zonas de refugio en aguas marinas de
334 jurisdiccion federal frente a la costa oriental del estado de baja california sur, en el corredor marino de
335 san cosme a punta coyote. *Diario Oficial de la Federación*
- 336 DOF, D. (2012b). Acuerdo por el que se establece una red de zonas de refugio pesquero en aguas marinas de
337 jurisdiccion federal ubicadas en el área de sian ka an, dentro de la bahía espíritu santo en el estado de
338 quintana roo. *Diario Oficial de la Federación*
- 339 DOF, D. (2013). Acuerdo por el que se establece una red de zonas de refugio pesquero en aguas marinas de
340 jurisdiccion federal ubicadas en las áreas de banco chinchorro y punta herrero en el estado de quintana
341 roo. *Diario Oficial de la Federación*
- 342 DOF, D. (2018a). Acuerdo por el que se establece el área de refugio para la tortuga amarilla (caretta
343 caretta) en el golfo de ulloa, en baja california sur. *Diario Oficial de la Federación*
- 344 DOF, D. (2018b). Acuerdo por el que se establece una red de dos zonas de refugio pesquero parciales
345 permanentes en aguas marinas de jurisdiccion federal adyacentes a isla natividad, ubicada en el municipio
346 de mulegé, en el estado de baja california sur. *Diario Oficial de la Federación*
- 347 Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., et al. (2014). Global
348 conservation outcomes depend on marine protected areas with five key features. *Nature* 506, 216–220.
349 doi:10.1038/nature13022
- 350 Espinosa-Romero, M. J., Rodriguez, L. F., Weaver, A. H., Villanueva-Aznar, C., and Torre, J. (2014). The
351 changing role of ngos in mexican small-scale fisheries: From environmental conservation to multi-scale
352 governance. *Marine Policy* 50, 290–299. doi:10.1016/j.marpol.2014.07.005
- 353 Ferraro, P. J. and Pattanayak, S. K. (2006). Money for nothing? a call for empirical evaluation of biodiversity
354 conservation investments. *PLoS Biol* 4, e105. doi:10.1371/journal.pbio.0040105
- 355 Finkbeiner, E., Micheli, F., Saenz-Arroyo, A., Vazquez-Vera, L., Perafan, C., and Cárdenas, J. (2018).
356 Local response to global uncertainty: Insights from experimental economics in small-scale fisheries.
357 *Global Environmental Change* 48, 151–157. doi:10.1016/j.gloenvcha.2017.11.010
- 358 Friedlander, A. M., Golbuu, Y., Ballesteros, E., Caselle, J. E., Gouezo, M., Olsudong, D., et al. (2017). Size,
359 age, and habitat determine effectiveness of palau's marine protected areas. *PLoS ONE* 12, e0174787.
360 doi:10.1371/journal.pone.0174787
- 361 Fulton, S., Caamal-Madrigal, J., Aguilar-Perera, A., Bourillón, L., and Heyman, W. D. (2018). Marine
362 conservation outcomes are more likely when fishers participate as citizen scientists: Case studies from
363 the mexican mesoamerican reef. *CSTP* 3. doi:10.5334/cstp.118
- 364 Fulton, S., Hernandez-Velasco, A., Suarez-Castillo, A., Fernandez-Rivera Melo, F., Rojo, M., Saenz-
365 Arroyo, A., et al. (2019). From fishing fish to fishing data: the role of artisanal fishers in conservation
366 and resource management in mexico. In *Viability and Sustainability of Small-Scale Fisheries in*
367 *Latin America and The Caribbean*, eds. S. Salas, M. J. Barragán-Paladines, and R. Chuenpagdee
368 (Cham: Springer International Publishing), vol. 19 of *MARE Publication Series*. 151–175. doi:10.1007/
369 978-3-319-76078-0_7
- 370 Gelcich, S. and Donlan, C. J. (2015). Incentivizing biodiversity conservation in artisanal fishing com-
371 munities through territorial user rights and business model innovation. *Conserv Biol* 29, 1076–1085.
372 doi:10.1111/cobi.12477
- 373 Gelcich, S., Godoy, N., Prado, L., and Castilla, J. C. (2008). Add-on conservation benefits of marine
374 territorial user rights fishery policies in central chile. *Ecol Appl* 18, 273–281. doi:10.1890/06-1896.1

- 375 Giakoumi, S., Scianna, C., Plass-Johnson, J., Micheli, F., Grorud-Colvert, K., Thiriet, P., et al. (2017).
376 Ecological effects of full and partial protection in the crowded mediterranean sea: a regional meta-
377 analysis. *Sci Rep* 7, 8940. doi:10.1038/s41598-017-08850-w
- 378 Green, A., Chollett, I., Suarez, A., Dahlgren, C., Cruz, S., Zepeda, C., et al. (2017). *Biophysical Principles*
379 *for Designing a Network of Replenishment Zones for the Mesoamerican Reef System*. Technical report
- 380 Guidetti, P., Baiata, P., Ballesteros, E., Di Franco, A., Hereu, B., Macpherson, E., et al. (2014). Large-scale
381 assessment of mediterranean marine protected areas effects on fish assemblages. *PLoS ONE* 9, e91841.
382 doi:10.1371/journal.pone.0091841
- 383 Halpern, B. S., Klein, C. J., Brown, C. J., Beger, M., Grantham, H. S., Mangubhai, S., et al. (2013).
384 Achieving the triple bottom line in the face of inherent trade-offs among social equity, economic return,
385 and conservation. *Proc Natl Acad Sci USA* 110, 6229–6234. doi:10.1073/pnas.1217689110
- 386 Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., et al. (2008). A global
387 map of human impact on marine ecosystems. *Science* 319, 948–952. doi:10.1126/science.1149345
- 388 Hilborn, R., Micheli, F., and De Leo, G. A. (2006). Integrating marine protected areas with catch regulation.
389 *Can. J. Fish. Aquat. Sci.* 63, 642–649. doi:10.1139/f05-243
- 390 Hilborn, R., Stokes, K., Maguire, J.-J., Smith, T., Botsford, L. W., Mangel, M., et al. (2004). When
391 can marine reserves improve fisheries management? *Ocean and Coastal Management* 47, 197 – 205.
392 doi:<https://doi.org/10.1016/j.ocecoaman.2004.04.001>
- 393 Johannes, R. E. (2002). The renaissance of community-based marine resource management in oceania.
394 *Annual Review of Ecology and Systematics* 33, 317–340
- 395 Jupiter, S. D., Epstein, G., Ban, N. C., Mangubhai, S., Fox, M., and Cox, M. (2017). A social–ecological
396 systems approach to assessing conservation and fisheries outcomes in fijian locally managed marine
397 areas. *Soc Nat Resour* 30, 1096–1111. doi:10.1080/08941920.2017.1315654
- 398 Krueck, N. C., Ahmadi, G. N., Possingham, H. P., Riginos, C., Treml, E. A., and Mumby, P. J. (2017).
399 Marine reserve targets to sustain and rebuild unregulated fisheries. *PLoS Biol* 15, e2000537. doi:10.
400 1371/journal.pbio.2000537
- 401 Leslie, H. M., Basurto, X., Nenadovic, M., Sievanen, L., Cavanaugh, K. C., Cota-Nieto, J. J., et al. (2015).
402 Operationalizing the social-ecological systems framework to assess sustainability. *Proc Natl Acad Sci U
403 SA* 112, 5979–5984. doi:10.1073/pnas.1414640112
- 404 Lester, S., Halpern, B., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B., Gaines, S., et al. (2009).
405 Biological effects within no-take marine reserves: a global synthesis. *Mar. Ecol. Prog. Ser.* 384, 33–46.
406 doi:10.3354/meps08029
- 407 Lester, S., McDonald, G., Clemence, M., Dougherty, D., and Szewalski, C. (2017). Impacts of TURFs and
408 marine reserves on fisheries and conservation goals: theory, empirical evidence, and modeling. *BMS* 93,
409 173–198. doi:10.5343/bms.2015.1083
- 410 López-Angarita, J., Moreno-Sánchez, R., Maldonado, J. H., and Sánchez, J. A. (2014). Evaluating linked
411 social-ecological systems in marine protected areas. *Conserv Lett* 7, 241–252. doi:10.1111/conl.12063
- 412 Mascia, M. B., Fox, H. E., Glew, L., Ahmadi, G. N., Agrawal, A., Barnes, M., et al. (2017). A novel
413 framework for analyzing conservation impacts: evaluation, theory, and marine protected areas. *Ann NY
414 Acad Sci* 1399, 93–115. doi:10.1111/nyas.13428
- 415 McCay, B. (2017). Territorial use rights in fisheries of the northern pacific coast of mexico. *BMS* 93,
416 69–81. doi:10.5343/bms.2015.1091
- 417 McCay, B. J., Micheli, F., Ponce-Díaz, G., Murray, G., Shester, G., Ramirez-Sánchez, S., et al. (2014).
418 Cooperatives, concessions, and co-management on the pacific coast of mexico. *Marine Policy* 44, 49–59.
419 doi:10.1016/j.marpol.2013.08.001

- 420 Micheli, F., Saenz-Arroyo, A., Greenley, A., Vazquez, L., Espinoza Montes, J. A., Rossetto, M., et al.
421 (2012). Evidence that marine reserves enhance resilience to climatic impacts. *PLoS ONE* 7, e40832.
422 doi:10.1371/journal.pone.0040832
- 423 Moland, E., Olsen, E. M., Knutsen, H., Garrigou, P., Espeland, S. H., Kleiven, A. R., et al. (2013). Lobster
424 and cod benefit from small-scale northern marine protected areas: inference from an empirical before-
425 after control-impact study. *Proceedings of the Royal Society B: Biological Sciences* 280, 20122679–
426 20122679. doi:10.1098/rspb.2012.2679
- 427 NOM-049-SAG/PESC (2014). Norma oficial mexicana nom-049-sag/pesc-2014, que determina el procedi-
428 miento para establecer zonas de refugio para los recursos pesqueros en aguas de jurisdicción federal de
429 los estados unidos mexicanos. *DOF*
- 430 [Dataset] OECD (2017). Inflation CPI
- 431 Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science*
432 325, 419–422. doi:10.1126/science.1172133
- 433 Ovando, D., Dougherty, D., and Wilson, J. R. (2016). Market and design solutions to the short-term
434 economic impacts of marine reserves. *Fish Fish* 17, 939–954. doi:10.1111/faf.12153
- 435 Pauly, D., Watson, R., and Alder, J. (2005). Global trends in world fisheries: impacts on marine ecosystems
436 and food security. *Philosophical Transactions of the Royal Society B: Biological Sciences* 360, 5–12.
437 doi:10.1098/rstb.2004.1574
- 438 Pérez-Ramírez, M., Castrejón, M., Gutiérrez, N. L., and Defeo, O. (2016). The marine stewardship council
439 certification in latin america and the caribbean: A review of experiences, potentials and pitfalls. *Fisheries*
440 Research 182, 50–58. doi:10.1016/j.fishres.2015.11.007
- 441 Pérez-Ramírez, M., Ponce-Díaz, G., and Lluch-Cota, S. (2012). The role of msc certification in the
442 empowerment of fishing cooperatives in mexico: The case of red rock lobster co-managed fishery. *Ocean*
443 *Coast Manag* 63, 24–29. doi:10.1016/j.ocecoaman.2012.03.009
- 444 R Core Team (2018). *R: A Language and Environment for Statistical Computing*. R Foundation for
445 Statistical Computing, Vienna, Austria
- 446 Rees, S. E., Pittman, S. J., Foster, N., Langmead, O., Griffiths, C., Fletcher, S., et al. (2018). Bridging the
447 divide: Social–ecological coherence in marine protected area network design. *Aquatic Conservation: Marine and Freshwater Ecosystems*
- 449 Roberts, C. M., OLeary, B. C., McCauley, D. J., Cury, P. M., Duarte, C. M., Lubchenco, J., et al. (2017).
450 Marine reserves can mitigate and promote adaptation to climate change. *Proc Natl Acad Sci USA* 114,
451 6167–6175. doi:10.1073/pnas.1701262114
- 452 Rodriguez, A. G. and Fanning, L. M. (2017). Assessing marine protected areas effectiveness: A case study
453 with the tobago cays marine park. *OJMS* 07, 379–408. doi:10.4236/ojms.2017.73027
- 454 Rossetto, M., Micheli, F., Saenz-Arroyo, A., Montes, J. A. E., and De Leo, G. A. (2015). No-take marine
455 reserves can enhance population persistence and support the fishery of abalone. *Can. J. Fish. Aquat. Sci.*
456 72, 1503–1517. doi:10.1139/cjfas-2013-0623
- 457 Sala, E. and Giakoumi, S. (2017). No-take marine reserves are the most effective protected areas in the
458 ocean. *ICES Journal of Marine Science* doi:10.1093/icesjms/fsx059
- 459 Smallhorn-West, P. F., Bridge, T. C. L., Malimali, S., Pressey, R. L., and Jones, G. P. (2018). Predicting
460 impact to assess the efficacy of community-based marine reserve design. *Conserv Lett*, e12602doi:10.
461 1111/conl.12602
- 462 Villaseñor-Derbez, J. C., Faro, C., Wright, M., Martínez, J., Fitzgerald, S., Fulton, S., et al. (2018).
463 A user-friendly tool to evaluate the effectiveness of no-take marine reserves. *PLOS ONE* 13, 1–21.
464 doi:10.1371/journal.pone.0191821

- 465 Woodson, C., Micheli, F., Boch, C., M, A.-N., Hernandez, A., Vera, L., et al. (in press). Harnessing
466 environmental variability as a climate change adaptation for small-scale fisheries. *Conservation Letters*
467 Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., et al. (2006). Impacts of
468 biodiversity loss on ocean ecosystem services. *Science* 314, 787–790. doi:10.1126/science.1132294
469 Zeileis, A. (2004). Econometric computing with hc and hac covariance matrix estimators. *J Stat Softw* 11.
470 doi:10.18637/jss.v011.i10

FIGURE CAPTIONS

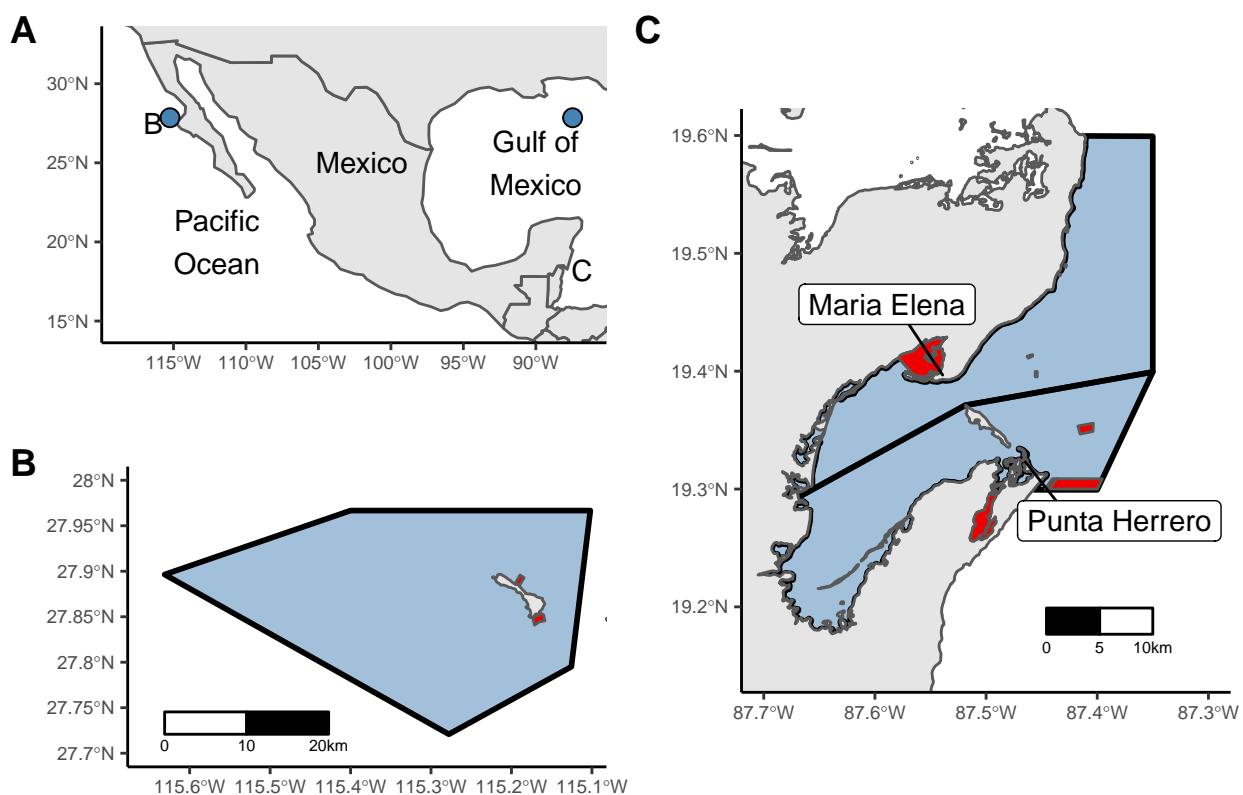


Figure 1. Location of the three coastal communities studied (A). Isla Natividad (B) is located off the Baja California Peninsula, Maria Elena and Punta Herrero (C) are located in the Yucatan Peninsula. Blue polygons represent the TURFs, and red polygons the marine reserves.

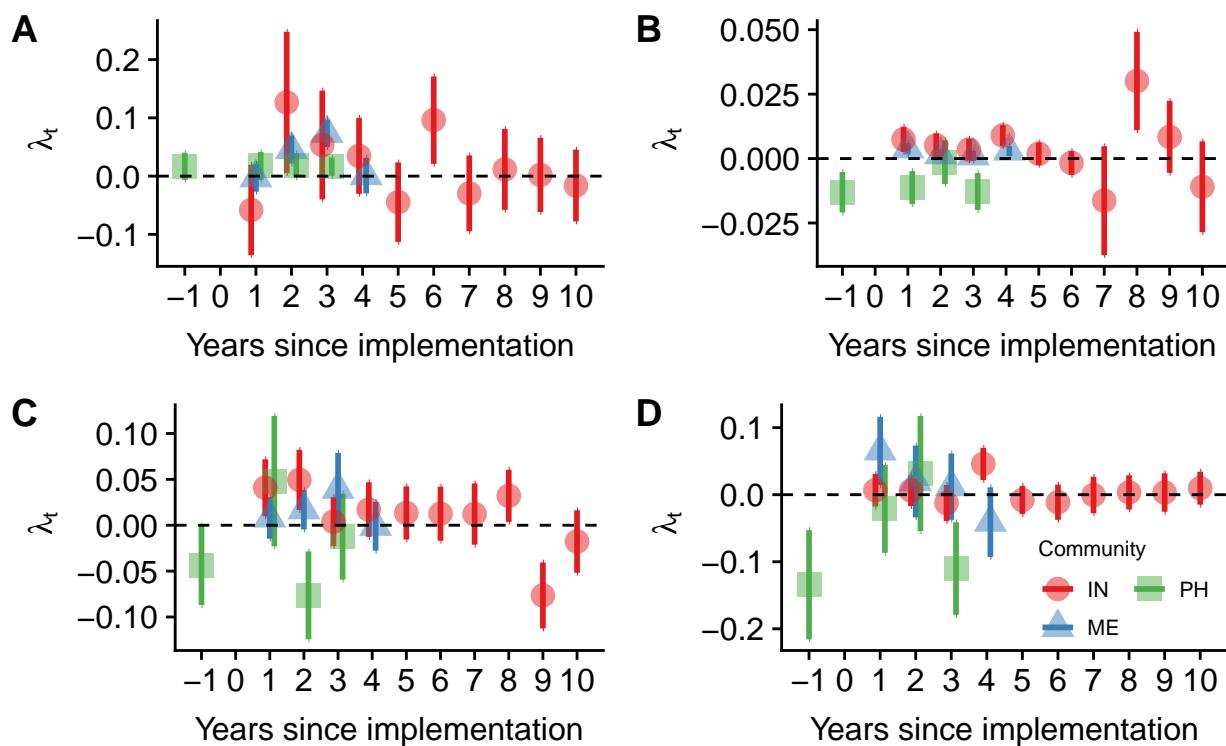


Figure 2. Effect sizes for marine reserves from Isla Natividad (IN; red circles), Maria Elena (ME; blue triangles), and Punta Herrero (PH; green squares) for lobster densities (*Panulirus spp.*; A), fish biomass (B), invertebrate densities (C), and fish densities (D). Plots are ordered by survey type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

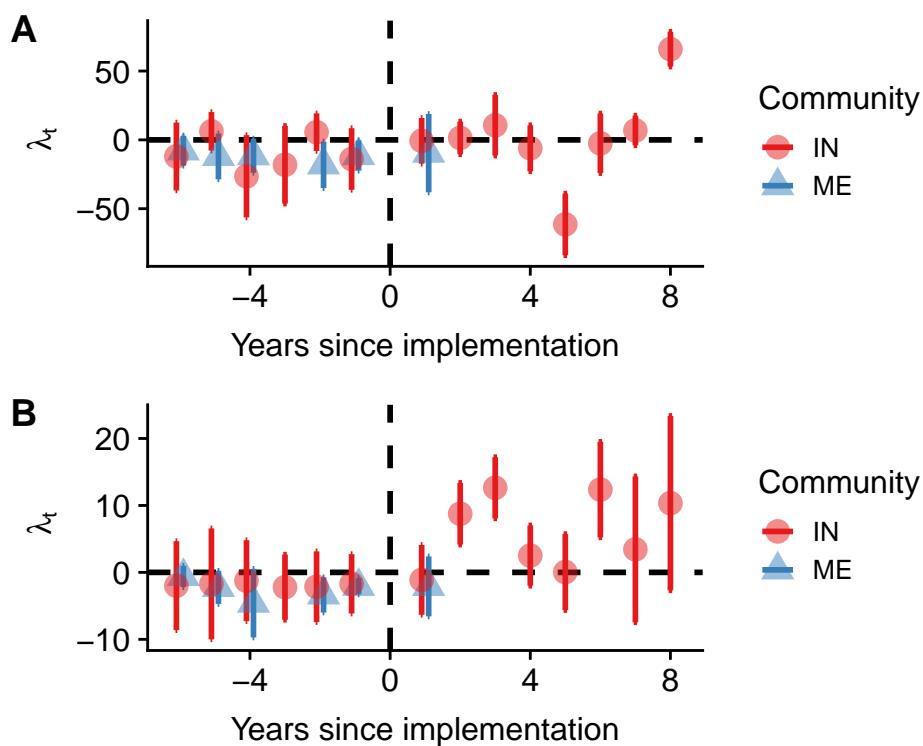


Figure 3. Effect sizes for lobster catches (A) and revenues (B) in at Isla Natividad (IN; red circles) and Maria Elena (ME; blue triangles). Points indicate the effect size and standard errors. Years have been centered to year of implementation.

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

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

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).

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 mandate. 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.