Displacement of fishing effort by Large Scale Marine Protected Areas

Updated on 2018-10-16

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Abstract

Large-scale Marine Protected Areas (LSMPAs) have seen a significant increase over the last years. Fishing effort is effectively eliminated within these protected areas upon implementation. The benefits of reducing effort have been largely studied, and include increases in abundance, biomass, and diversity within the bounded regions. These no-take zones may produce spillover effects, which provide fish for outside areas. However, the economic and ecological implications of displacing fishing effort are not yet fully understood. Novel data products that track fishing effort at the vessel-level allow us to identify changes in fleet- and vessel-level behavior upon the implementation of protected areas, as well as how these redistribute. This papers evaluates the implications of implementing LSMPA, by evaluating changes in fishing hours, showing that vessels in the effected region reduce fishing effort after the implementation of PIPA. Our results are robust to a set of specifications. We also track the relative spatial allocation of fishing events thorugh time, and identify that areas closer to PIPA show an increase in relative fishing hourse due to the displacement of PIPA-fishing vessels. Our results not only provide an impact evaluation of the effect of LSMPAs on fishing activity, but provide insights into vessel redistribution dynamics, which may have ecological and economic implications.

1 Introduction

Marine Protected Areas (MPAs) are intended to safeguard parts of the ocean from fishing and other extractive activities. Current international goals aim to protect 10% of the ocean environments by 2020. In an effort to meet this target, the world has seen a rapid increase in MPA coverage (Wood et al., 2008; Sala et al., 2018). A significant part of this rapid increase can be attributed to the designation of a small number of Large Scale Marine Protected Areas (LSMPAs; Boonzaier and Pauly (2016); Singleton and Roberts (2014); Gray et al. (2017)). These are defined as MPAS larger than 250,000 Km^2 in extension (Fig. 1), and are often implemented in the pelagic environment, where the dominant human activity is industrial fishing (Toonen et al., 2013; Gray et al., 2017; Kroodsma et al., 2018).

Given the relatively recent establishment of most LSMPAs, very little is known about their human dimensions and implication for fisheries (Gray et al., 2017). Moreover, the world has seen a widespread increase in the implementation of LSMPAs, likely as a way to rapidly achieve the 10% target (Boonzaier and Pauly, 2016; Alger and Dauvergne, 2017; Sala et al., 2018). However, as with customary MPAs, it is important that we understand the socioeconomic implications of management interventions that result in the spatial displacement of fishing effort. They **key gap in the literature is X**. We adress it here by using new stuff and find results that are of relevance to policy makers and advance our knowledge, which should be incorporated when designing and planning for LSMPAs.

Due to weak property rights, limited habitat transformation, and potentially lower management costs, pelagic MPAs provide an opportunity to safeguard the oceans Game et al. (2009). A growing body of literature has shown that closing the high seas to all fishing could increase fishery yields and profitability of fisheries, with negligible costs to food security (White and Costello, 2014; Sumaila et al., 2015; Sala et al., 2018; Schiller et al., 2018).

The early literature on LSMPAs focused on the inherent difficulties that come with a pelagic (*i.e.* open water) environment, where organisms typically have large home ranges. Game et al. (2009) suggest that most of the challenges can be overcome with the incorporation of technology, in what then became known as Dynamic Ocean Management (Maxwell et al., 2015). **?** claim that very large MPAs would result in excessive opportunity costs and that these would be difficult to enforce. Toonen et al. (2013) show that just a small number of LSMPAs made up 80% of the managed areas in the ocean in 2013, and that these can help achieve the CBD Aichi targets faster. Singleton and Roberts (2014) provide an objective discussion of pros and cons of LSMPAs.

LSMPAs were erroneously assumed to have little social implications due to their remoteness. However, there have been calls to incorporate the human dimensions into LSMPAs management and evaluation (Agardy et al., 2011; Gray et al., 2017). Most research incorporating these human dimensions has focused on governance and enforcement of LSMPAs (*i.e.* Alger and Dauvergne (2017); Christie et al. (2017)), but they are yet to be the focus of economic analyses (Gray et al., 2017). Overall, there has been little empirical work regarding LSMPAs. Recent technological advances in vessel-detection systems allows for the discovery and advancement of many important facets of LSMPAs. For example, (McDermott et al., 2018) show that the anticipation of a LSMPA can lead to preemptive overfishing, which can erode or delay the expected benefits of the intervention. White et al. (2017) combine shark tags and vessel-tracking data to demonstrate that the fairly large Palmyra Atoll National Wildlife Refuge (54,000 Km^2) protectes two thrids of the tagged grey reef sharks by effectively excluding fishing effort. No studies have evaluated the displacement of fishing effort due to LSMPA implementation.

The exclusion of these LSMPAs is likely to change fisher's behavior. Theoretical models of fishing effort redistribution range from the simplistic assumption that effort inside the bounded region disappears, to spatially explicit models that reallocate fishing effort based on habitat characteristics, presence of other vessels, and expected returns (Smith and Wilen, 2003; Hilborn et al., 2006). However, these focus on the long term optimal equilibrium and redistribution of fishing effort may not be optimal, especially over the first years (Stevenson et al., 2013). The few empirical works suggest that resource users may show idiosyncratic responses. For example, Stevenson et al. (2013) show that a network of MPAs displaced fishing effort farther away from ports resulting in higher *perceived* costs, but that catch per unit effort also increased. Cabral et al. (2017) analyse the redistribution of fishing and non-fishing vessels following the implementation of MPAs in California, and find that commercial dive boats follow a fishing-the-line pattern, while some fishing boats follow an ideal free distribution. The way in which fishers react to a spatial closure can have major implications in its outcome (Smith and Wilen, 2003; Hilborn et al., 2006) highlighting the need to understand how fishers react to the implementation of LSMPAs, and fishing effort changes and is spatially redistributed.

The main objective of this paper is to identify behavioral responses to the implementation of LSMPAs. We combine novel vessel tracking technologies and causal inference techniques to identify behavioral changes of fishing vessels due to the implementation of PIPA. We focus on fishing hours and distance traveled as outcome variables that fishers might adjust following implementation of a LSMPA in an impact-evaluation fashion. Additionally, we evaluate the spatial redistribution of fishing effort that existed within PIPA before its implementation. This work provides novel empirical insights into fisher's responses to the implementation of LSMPAs, and can help guide future interventions. Our work is novel in the sense that it provides empirical evidence of the effect of large-acale Marine Protected Areas in fishing behavior and distribution.

The next sections are as follows: Section 2 provides an overview of the Nauru Agreement and associated countries, a description of the fleet that operates in the region, and a brief history of PIPA. Section 3 describes our data and identification strategy. Section 4 presents our results, section 5 provides an extension of our results to other LSMPAs and discusses our results.

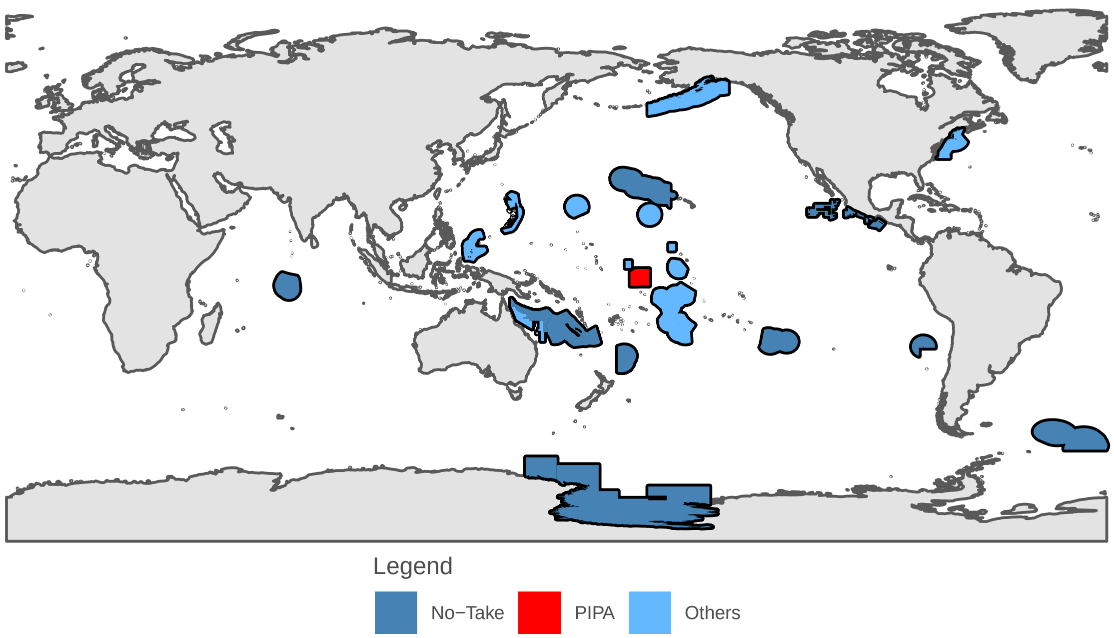


Figure 1: Large Scale Marine Protected Areas. The map shows all areas larger than 250,000 Km^2. Areas in dark blue meet at least one of these conditions: reported no take area is greater than 0, are categorized as IUCN Ia or Ib, their designated english name is 'Protected Area'.

2 Background

The Nauru Agreement was established in 1982 by Pacific island nations to manage their important tuna resources. PNA Members include Federated States of Micronesia, Kiribati, Marshall Islands, Nauru, Palau, Papua New Guinea, Solomon Islands, and Tuvalu.The Nauru Agreement regulated access of foreign vessels (*i.e.* those from non-PNA countries). Holding ~80% of the historical purse seining grounds within their Exclusive Economic Zones, PNA countries gained bargaining power when providing access to foreign fleets (Havice, 2010).

The cooperation that emerged thanks to the PNA allowed for subsequent agreements that strengthened fisheries management, like the Palau Agreement, which limited the number of purse seiners at 205 vessels from 1995-2007[1]. However, the most notable regulation is their approach to manage fishing effort: a Vessel Day Scheme (VDS) implemented in 2007 (Havice, 2013). This effectively modified how fishing effort was managed, from number of vessels under the Palau Agreement to fishing hours. The VDS works as follows: Each year, scientific advisors recomend a total number of fishing vessel-days per year. Hours are allocated to each PNA country based on catch history, and they then sell fishing rights to other non-PNA countries (Aqorau et al., 2018). While the effectiveness of this scheme has been debated in terms of meeting their fishery management and conservation objectives, the licensing significantly contributes to the economy of these island nations (Havice, 2010).

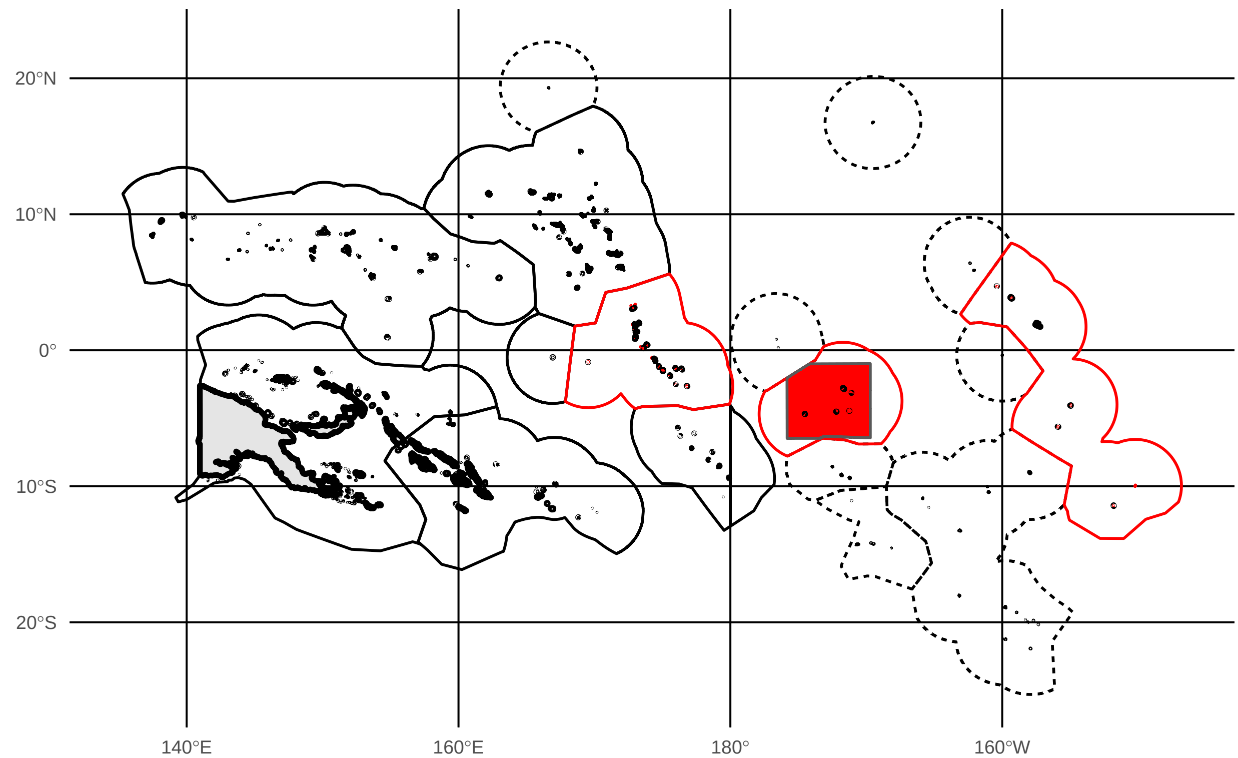


Figure 2: Map of the Exclusive Economic Zones (EEZs) of the region of interest. A solid line indicates countries that belong to the PNA, while a dashed line indicates all others. A red line indicates the Kiribati EEZ, and a solid red polygon delineates PIPA.

[1]See Havice (2010) for a detailed description of the Nauru, Palau, and Federal States of Micronesia agreements, their objectives, and outcomes.

The main tuna species caught in the region are skipjack (*Katsuwonus pelamis*), yellowfin (*Thunnus albacares*), albacore (*Thunnus alalunga*) and bigeye (*Thunnus obesus*). From these, the first two are amongst the top-10 species represented in global fisheries production statistics, with 2016 catches increasing relative to the 2005-2014 average (Food and of the United Nations, 2018). The pacific region has historically accounted for a large portion of tuna species production (Aqorau and Bergin, 1997). Today, the PNA controls close to 50% of the global skipjack tuna production (PNA, 2018). A large portion of these catches derive from purse seine vessels licensed under the VDS. Fishing vessels from Australia, New Zeland, China, France, Korea, Japan, the Philippines, Taiwan, and the United States participate in the purse-seining VDS.

One of the most notable and recent management interventions in the region is the implementation of the Phoenix Island Protected Area (PIPA) by the government of Kiribati. PIPA was first declared in 2006, and established in 2008 with only 4% of it was declared as no-take. In January 1st, 2015, the no-take area within PIPA was expanded to a total area of 397,447 Km^2, roughly 1.5 times the size of Ecuador. The implementation of such a large conservation area in one of the most important fishing regions in the world provides a great opportunity to evaluate the behavioral responses and redistribution of fishing effort by vessels that used to fish there. McDermott et al. (2018) showed that fishing effort within the Phoenix Islands Marine Protected Area (PIPA) increased between the anouncement (September 1st, 2014) and its implementation in Janyary 1st, 2015. Likewise, they demonstrate that fishing effort is effectively reduced after implementation. To this, we pose two questions: How do individual vessels respond to the sudden exclusion of such a big area? And where did all the vessels go? In the next sections we describe the data and methods used to answer this questions.

3 Methods

This section is divided into two main parts. First, we provide a general description of AIS data and the process of identification of vessel-level fishing events done by Global Fishing Watch[2]. Alongside, we describe the subset of data that we use for these analyses. When relevant, we also point out possible shortcomings in the data, or factors that must be considered in the later analyses. We then move on to explain our empirical strategy for the identification of the behavioral changes and redistribution of fishing effort.

3.1 Data

Automatic Identification Systems are on-board devices intended to provide at-sea safety and prevent ship collisions by broadcasting vessel position, course, and activities to surrounding vessels. These broadcasted messages can be received by satellites and land-based antennas. GFW uses a neural network to infer vessel characteristics and whether each broadcasted position represents a fishing event, thus allowing us to estimate near real-time fishing events globally since 2012 (Kroodsma et al., 2018). Our data contain information for 2012 - 2017. The recent addition of satellites that can receive AIS signals causes an apparent increase in the number of broadcasted AIS messages (*i.e.* points), and therefore number of vessels and fishing hours. The variability in AIS data and ocean conditions require that temporal trends be taken into account. We do that by obtaining a subset of

[2]Global Fishing Watch: globalfishingwatch.org

data that meet a BACI design, which gives us the full tracks for vessels affected and unaffected by the implementation of PIPA.

Our data contain over 45 million individual AIS messages (*i.e.* positions) for 371 fishing vessels that at some point fished in PNA waters. A total of 233 vessels have fished within PIPA waters; 217 did so at least once before 2015. However, not all vessels continued to fish elsewhere after PIPA implementation: 34 vessels have no recorded AIS messages after 2015, leaving us with 176 vessels that fished inside PIPA before its implementation, and continued to fish elsewhere afterward[3]. From these, we focus on purse seiners -the most important for PNA countries- and are left with 61 vessels[4]. New vessels might have also entered the fishery after PIPA closure, and were likely not exposed to the policy intervention in the pre-treatment period. To account for this, we identify a subset of vessels which we track since before the implementation of PIPA, and categorize them as treated or control vessels. Our treatment and control groups are defined as follows.

The treatment group contains all purse seiners (n = 61) that fished within PIPA at least once before the anouncement (09/01/2014), and that continued to fish elsewhere after the January 2015 implementation. Vessels in the control group meet the following two conditions: i) vessels never fished within PIPA waters, and ii) vessels have fished in surrounding areas (*i.e.* PNA-countries' EEZ) before and after PIPA closure. This definition of control vessels provides us with 26 purse seiners that never fished inside PIPA but fished in other PNA waters before and after PIPA implementation. During our causal identification, we also use two more strict definitions of control groups: one with only vessels that belong to PNA countries (n = 7), and one that excludes Chinese vessels (n = 21). Our definition of treatment and control groups leaves us with 61 and 21 treated and control vessels, which have just over 22 million observations where about 22% are identified as fishing events. For each of the remainig vessels, we calculate their total daily fishing hours and obtain a pannel data with 38,200 observations. Table 1 shows the number of vessels following a BACI design, as well as the fishing hours, before and after PIPA. Figure S1 provides a visual representation of the vessel-level fishing events that make up each group through time.

Table 1: Number of fishing vessels and mean daily fishing hours by group before and after PIPA implementation.

Group	n	Before	After	Change (A / B)
Control	26	11.74	20.13	1.72
Treatment	61	10.67	18.36	1.72

3.2 Analyses

The first analysis focuses on identifying the response of fishing vessels to PIPA closure. Our variables of interests are daily fishing hours, and daily distance traveled (Km). We compare our variable of interest before and after the implementation of PIPA using a Difference-in-Differences approach, where we track the variable of interest for vessels that used to fish inside PIPA and vessels that never fished inside PIPA, before and after PIPA implementation. Our specification is the following:

[3]The 34 missing vessels might have exited the fishery, been decommissioned or sold (therefore changing their AIS and mmsi), or turned off their AIS transmitters. In either case, we are not able to observe these.

[4]We perform some of the same analyses for longliners and include them in the Appendix

$$y_{i,t} = \alpha + \beta_1 Post_t + \beta_2 Treat_i + \beta_3 Post_t \times Treat_i + \phi_t + \gamma_i + \epsilon_{i,t}$$

Where $y_{i,t}$ is the variable of interest for vessel i in time period t. A dummy variable $Post_t$ takes the value of 0 for all dates prior to PIPA implementation and a value of 1 for all dates including and following PIPA implementation. $Treat_i$ is a dummy variable indicating whether a vessel belongs to the control ($Treat_i = 0$) or treatment ($Treat_i = 1$) group. α is the standard intercept, β_1 captures the temporal trend change, β_2 captures the difference between treated and control groups, and β_3 is our parameter of interest: de DiD estimate capturing the treatment effect. Finally, ϕ_t and γ_i represent month-, and flag-level dummies that account for seasonality or country-level management interventions[5]

Our second part of the analyses focuses on the redistribution of fishing effort. In other words, identifying where do vessels that used to fish inside PIPA go after its establishment. We discretize spatial units by using a polygon for PIPA [6] and distinct spatial units for each EEZ of each country. Some vessels might shift from EEZs into the High Seas, but we are interested in knowing *where* in the High Seas, so we incorporate additional regions by using a 1 degree buffer of the High Seas arround each of the EEZ regions. The rest of the high seas are merged into a single spatial unit. For example, if we were to do this only for Kiribati, we would have 8 spatial units: PIPA, three EEZs, three 1-degree buffers of High Seas arround each EEZs, and the rest of the High Seas. Whenever the buffers overlapped between themselves, we randocmply clipped one on to the other. EEZs that had sporadic fishing events were pooled into a group of "others".

To evaluate this change in effort allocation, we regress our variable of interest (*i.e.* fishing hours) on the interaction between a dummy variable indicating the policy intervention and a dummy variable for countries. This gives us the by-country change in proportional allocation of fishing effort:

$$y_{i,t} = \alpha + \beta_1 Post + \beta_{2,i} Country + \beta_{3,i} Post_t \times Country_i + \epsilon_{i,t}$$

Our variable of interest, $y_{i,t}$ represents the proportion of fishing hours that country i receives at time t. $Post$ also represents a policy dummy that takes the value of 0 for all dates before implementation of PIPA, and 1 otherwise. $Country$ is a dummy variable for countries for the spatial units defined above. Our parameter of interest is $\beta_{1,i}$, which captures the country-level change in proportional fishing effort.

All regression coefficients were estimated via ordinary least squares, and heteroskedastic-robust standard errors were calculated. All analyses were performed in R version 3.5.1 (R Core Team, 2018). Raw data and code used in this work are available on github.

[5] We test for other more complex specifications that interact a quarterly or year-month variable with the treatment group and find qualitativel the same results.

[6] we would expect to see a decrease here

Table 2: Changes in the relative allocation of fishing effort by region (EEZ, PIPA, high seas) and gear.

country	change
EEZ COK 1	0.53
EEZ FSM 1	0.81
EEZ KIR 1	-0.16
EEZ KIR 2	4.50
EEZ KIR 3	-2.79
EEZ MHL 1	-0.44
EEZ NRU 1	0.07
EEZ PNG 2	-9.07
EEZ SLB 1	3.18
EEZ TUV 1	1.27
HS	3.93
HS COK 1	0.06
HS FSM 1	0.01
HS KIR 1	2.96
HS KIR 2	1.07
HS KIR 3	2.53
HS MHL 1	0.11
HS MUS 1	0.00
HS NRU 1	-0.08
HS PNG 2	-0.20
HS SLB 1	-0.02
HS TUV 1	0.01
PIPA PIPA 1	-8.25

4 Results

Our data suggest that purse seiners and longliners have different responses to the implementation of a Large-Scale Marine Protected Areas. Fig. 4 shows that mean fishing hours for purse seiners have an abrupt increase, just before January 1st, 2015. This trend is observed for both treated and control groups. The pattern closely corresponds with the increase in total hours, but the total number of vessels doesn't entirely follow this pattern. The increase in fishing hours might be caused by the increased number of satellites[7]. Longliners, however, show no apparent trend with a clear seasonality (Ortuño-Crespo et al., 2018). The number of mmsi codes also increases slightly through time, but becomes stable after 2015. For both gears and across all measures, the treatment and control vessels follow similar patterns, confirming our claim that the control group provides a plausible counterfactual.

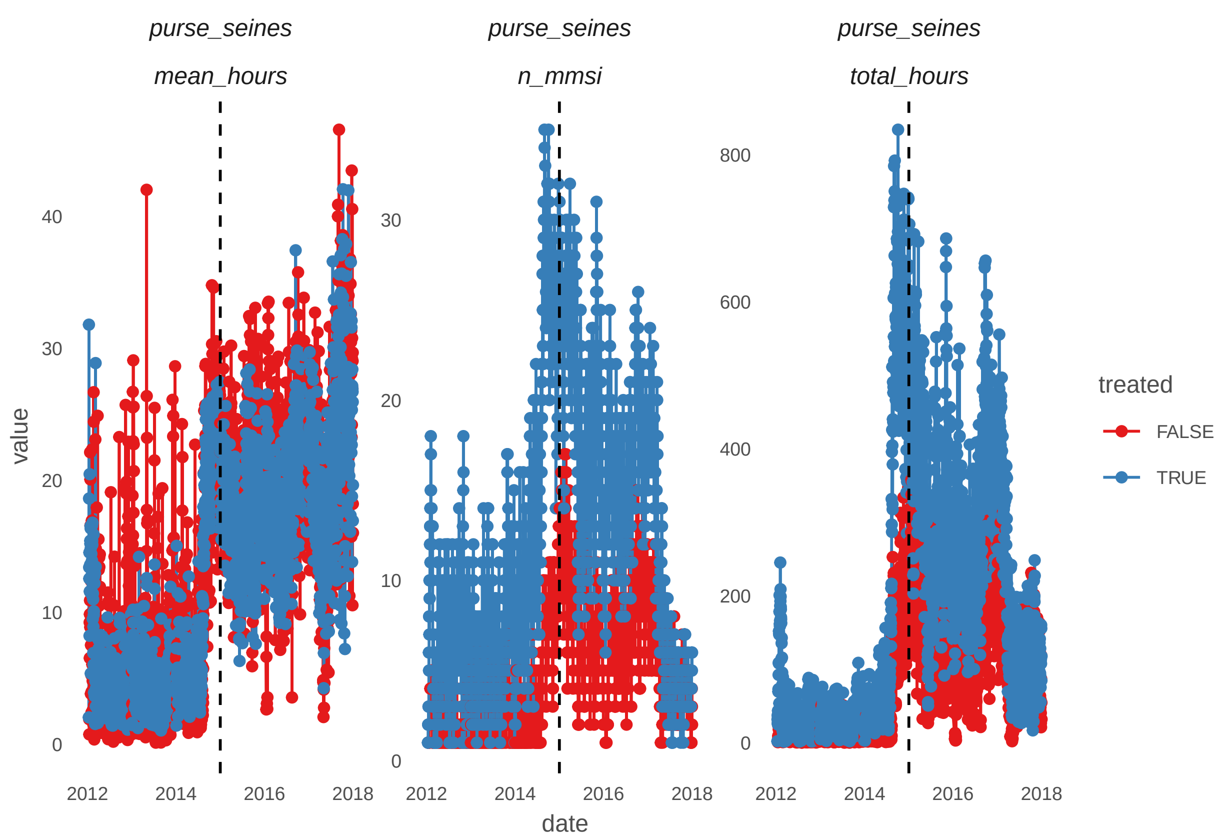


Figure 3: Fishing hours and number of vessels by month for all vessels. Vertical dashed line indicates PIPA closure.

[7]Need to check this. Not sure any satellites were incorporated during 2014. It is also possible that PNA countries started enforcing the requirement of having an AIS unit. On either case, both treatment and control groups seem to be affected equally.

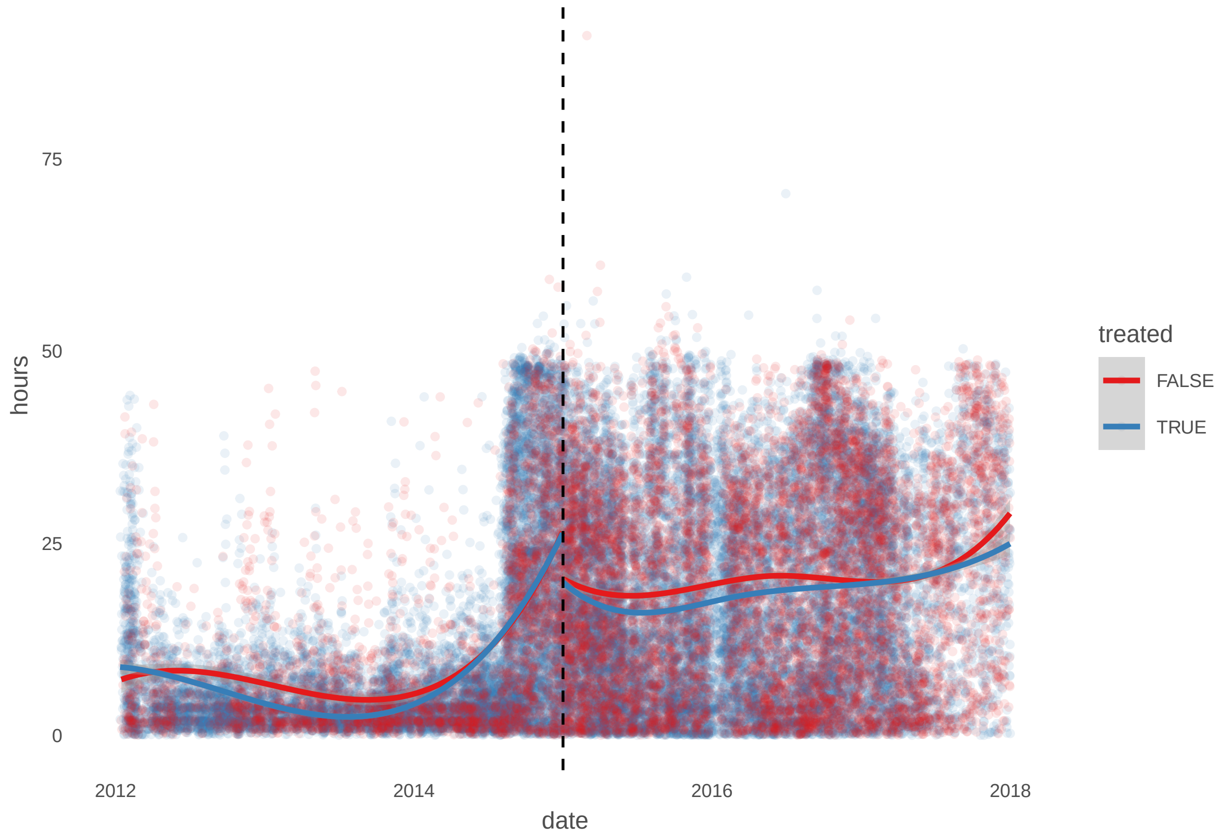


Figure 4: Fishing hours and number of vessels by month for all vessels. Vertical dashed line indicates PIPA closure.

Our DiD analysis shows that treated purse seiners reduce their fishing effort after PIPA implementation in the order of 16 hours per month. This result is robust and significant ($p < 0.05$) for all model specifications, with the effect varying between $\beta_3 = -16.457$ and $\beta_3 = -18.709$. Model specifications that include the year polynomial show lower values for the β_1 coefficient associated to the $Post_t$ policy dummy, and show positive and negative values for μ_1 and μ_2, the linear and quadratic terms for Y_t, respectively. These effectively represent the patterns observed in Figure 4.

Regressions coefficients for each gear type are shown in Tables **??** and S2. Column (1) presents the DiD regression with no fixed effects, column (2) includes month fixed effects, column (3) includes month and the second degree polynomial for years, and column (4) includes all of the above and country-level fixed effects.

Recall that to evaluate the redistribution of fishing effort we only track fishing vessels that belong to the treated group. In this case, we calculate the proportion of fishing effort allocated every month to each spatially explicit region outlined by EEZs and the high seas. For purse seiners, these represent 9 main EEZs, PIPA, the high seas, and a group of other EEZs. Figure 5 shows the monthly relative fishing hours that each region received by all 176 treated vessels. The top-left panel shows the change in fishing effort inside PIPA, including the preemptive fishing and immediate reduction previously reported (McDermott et al., 2018).

The change in the relative allocation of fishing effort by purse seiners increases in eight of the 12 regions after PIPA implementation (Table **??**). The largest increase is observed for the I-Kiribati EEZ, with an average increase of 0.11 ($p < 0.001$). In other words, the redistribution of treated vessels caused a 10% increase in the *relative* allocation of fishing effort within I-Kiribati waters. The only decrease is observed for Papua New Guinea, but the coefficient is not significant. Figure 6 provides a spatial representation of these changes. It is evident that the increase in relative fishing effort is greater for for regions closer to PIPA.

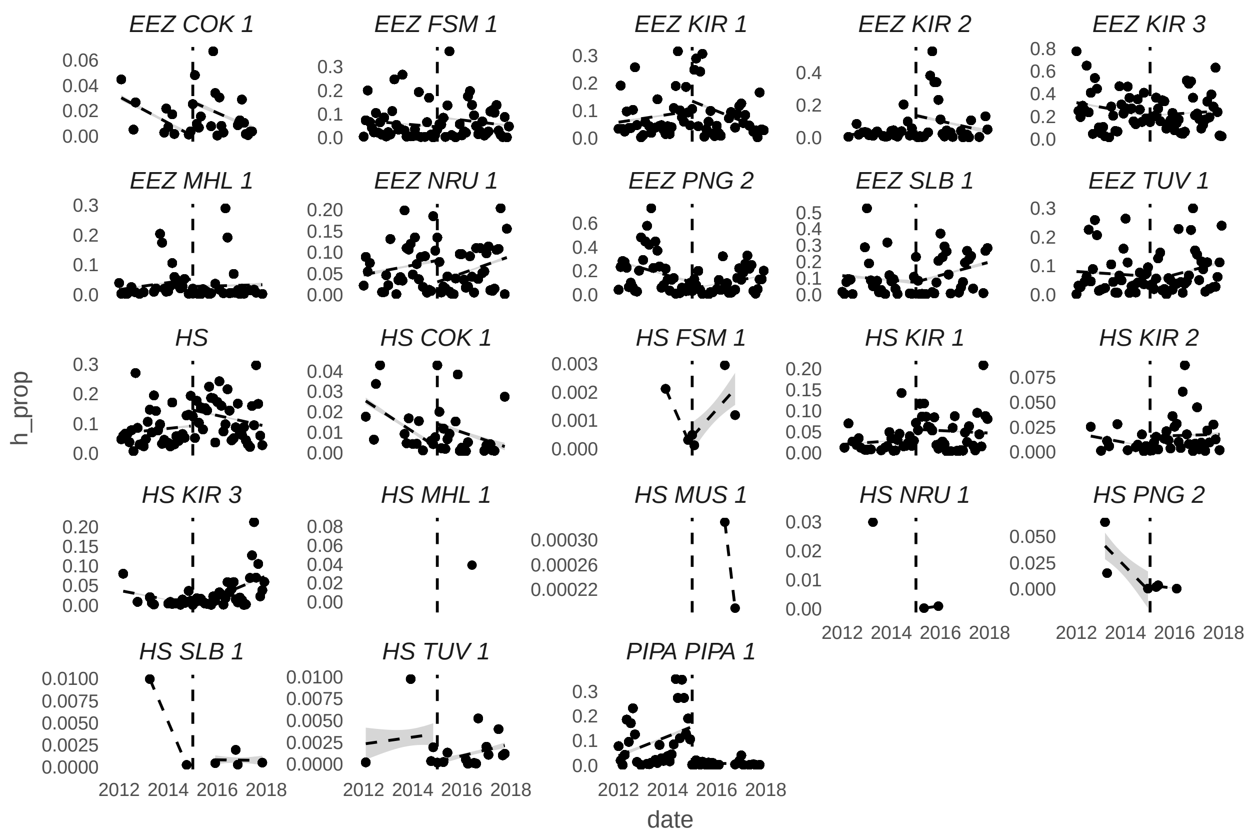


Figure 5: Monthly relative allocation of fishing effort by PIPA-fishing vessels before and after PIPA for 9 EEZs, PIPA, the high seas and 'other EEZs'.

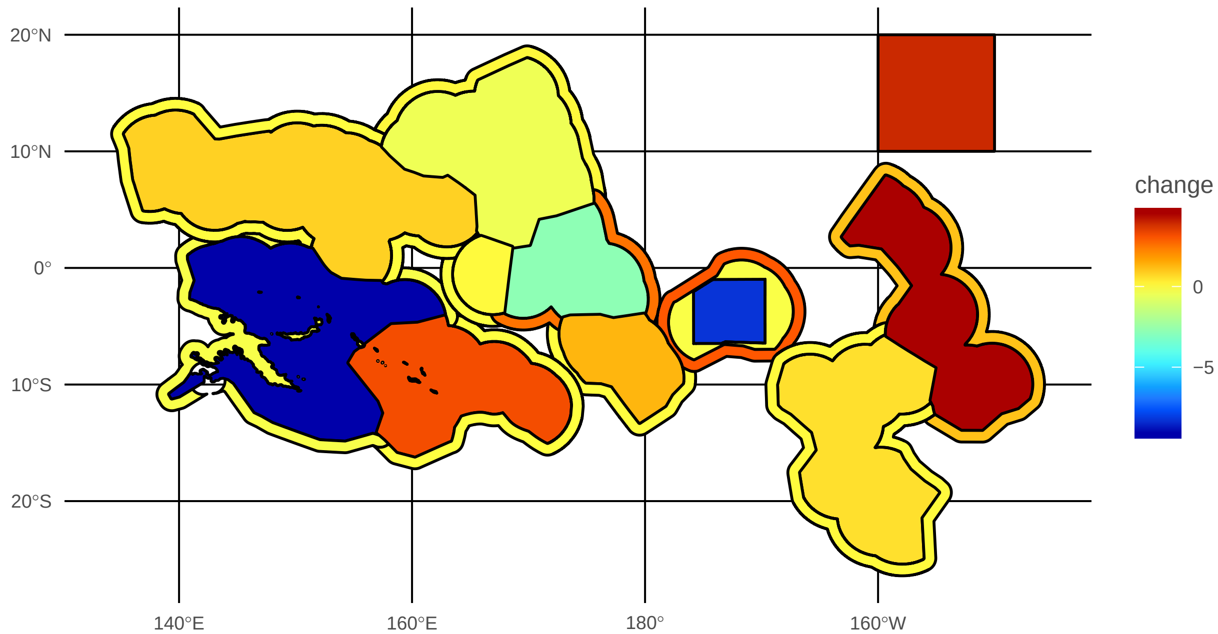


Figure 6: Spatial representation of the mean change in the monthly allocation of fishing effort for purse seiners.

Table 3: Change in the relative allocation of fishing hours by purse seiners ()

term	h_prop

5 Discussion

Our findings provide interesting insights into the effect that LSMPAs can have on vessel behavior and the redistribution of fishing effort. These collection of results shows that the implementation of PIPA caused treated vessels to reduce their fishing hours, and that this effect is greater for purse seiners than longliners. Even though treated vessels fish less, their relative allocation of fishing hours increased for all other fishing grounds. This does not imply that there is more fishing effort exerted by treated vessels, but rather that each region receives a greater portion of the post-PIPA fishing effort of these same vessels, which is lower than pre-PIPA levels. In this section we discuss the implications of vessel-level reductions in fishing effort and the increase in relative allocation of the remaining effort through space. We also provide plausible explanations as to why purse seiners seem to be more reactive to the spatial closure.

A major shortcoming of our analyses is that we do not observe catches or revenues, which ultimately are the factors that guide the decision making process of profit-maximizing agents. Therefore, it is difficult to know whether the reduction in fishing effort represents a positive or negative impact. A decrease in fishing effort is associated to an increase in catches (and therefore greater CPUE) only when the entire fleet does it, and if previous levels of effort were greater than F_{MEY} (*i.e.* the effort that would yield the maximum economic yield). Therefore, it is plausible that the reduction of fishing hours is not done by choice, but rather results from fishers having to increase search time. Upon being relocated, fishers may not identify the best fishing grounds as easily as before, and therefore invest a greater proportion of their time searching for their catch. Further analysis of temporal trends in non-fishing hours, as well as distance traveled should provide us with insights as to why fishers reduced fishing hours.

Previous studies on insular environments suggest that vessels move to distant places, which might be translated as increased costs (Stevenson et al., 2013). Nevertheless, they do not use counterfactuals that could help account for system- or fleet-level changes that occur through time. Others have used similar satellite-tracking systems to show that fishing effort accumulates near the edges of spatial closures, yielding greater catches (Murawski et al., 2005). Yet, these vessel tracks do not cover the pre-reserve period, making it difficult identify the contribution of spatial closures to the observed spatial distribution of fishing vessels. Recent work by Elahi et al. (2018) identified that total fishing effort in a focal region where a short-term MPA was implemented showed little change, likely indicating that fishers redistributed fishing effort to compensate for the reduction in available space. Our data is assembled in a similar way, with fishing positions before and after the implementation of PIPA and vessels grouped into treated and control groups. Our BACI design, along with our difference-in-differences analysis allows us to make causal inferences about the effect that large scale marine protected areas have on fishing effort.

The different responses observed between purse seiners and longliners might have two possible explanations. It is likely that PIPA did not contain habitat that longliners would consider optimal. Therefore, the sporadic fishing events that occurred there are of little importance to the fleet, and it is unlikely that the implementation of PIPA has an effect on them. Alternatively, the differences may be due to the nature of each fishing gear. Purse seiners are often constrained by seafloor and termocline depth, and have a smaller spatial footprint (Kroodsma et al., 2018). Tuna purse seiners are known to have greater proportion of null sets (*i.e.* where purse seines effectively cast their nets, but no catch is obtained) during El Niño years, where the termocline deepens in the Eastern Pacific (Dreyfus-Leon, 2015). On the other hand, longliners may be more flexible as to where they can deploy their longlines. Ortuño-Crespo et al. (2018) evaluated the ecological niche of the pelagic longline fleet, and suggest that the fleet may be under-utilizing the ocean, meaning that they can easily redistribute elsewhere.

Our work suggests that the implementation of LSMPAs can have important implications for purse seiners, and less so for longliners. We also show that fishing effort is redistributed to areas close by. Future management interventions that aim to close large portion of the oceans should consider how fishing effort will change in space and through time, and the ecological implications of this redistribution to ensure that fishing effort is not just displaced elsewhere, leading to overfishing in adjacent waters.

6 Appendix

Figure S1: Stream of fishing events by vessels through time. Each line represents a vessel, with dots indicating months with fishing activity and colors indicating the pre and post periods.

Table S1: Fishing hours from GFW for purse seiners (n = 106; 38 control, 68 treatment). Asterisks indicate significance levels. Numbers in parenthesis represent heteroskedastic-robuste standard errors.

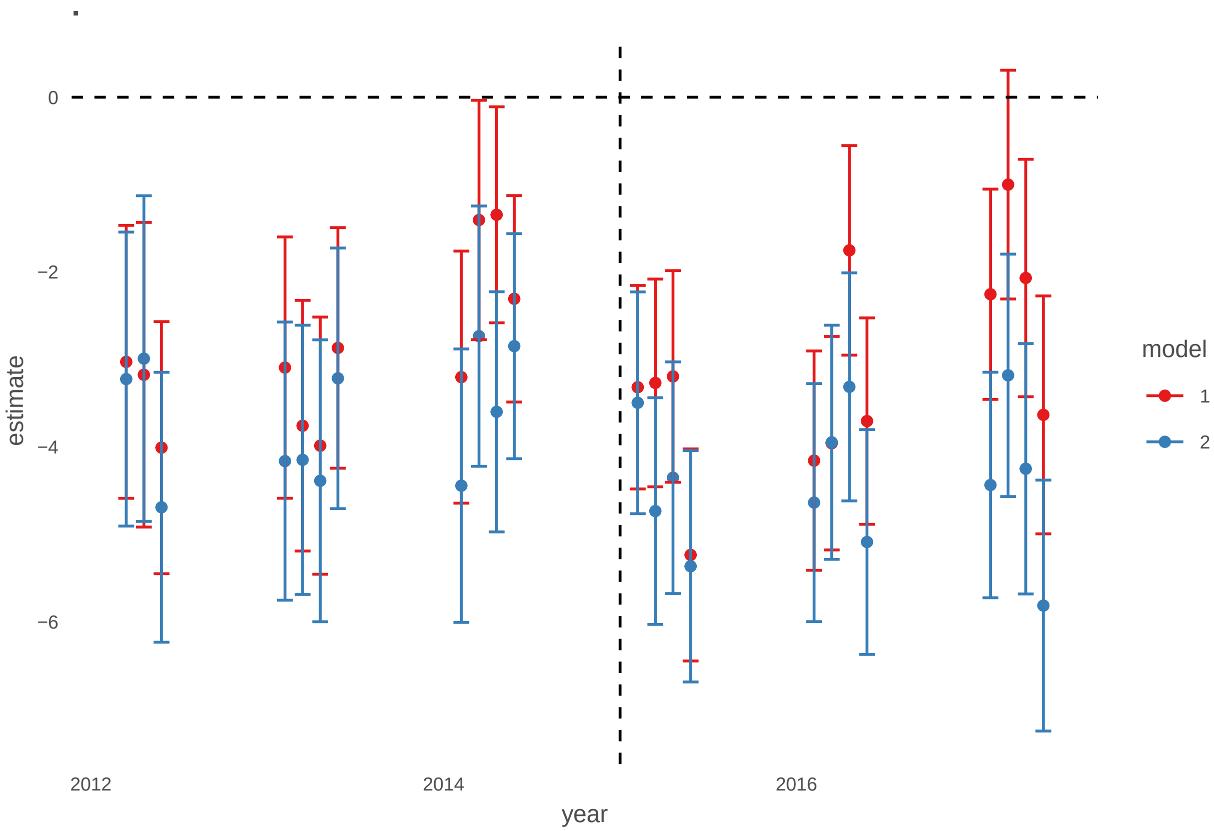
	Dependent variable:			
	hours			
	(1)	(2)	(3)	(4)
post	8.394***	9.334***	14.934***	17.284***
	(0.267)	(0.254)	(0.312)	(0.403)
treated	−1.069***	−0.770***	−0.993***	0.259
	(0.249)	(0.234)	(0.219)	(0.269)
post:treated	−0.701**	−0.899***	−0.530*	−0.580**
	(0.308)	(0.295)	(0.283)	(0.292)
Constant	11.738***	11.281***	7.574***	10.077***
	(0.220)	(0.292)	(0.318)	(0.422)
Month FE	No	Yes	Yes	Yes
Year FE	No	No	Yes	Yes
Flag FE	No	No	No	Yes
Observations	37,840	37,840	37,840	37,840
R^2	0.087	0.136	0.179	0.192
Note:	*p<0.1; **p<0.05; ***p<0.01			

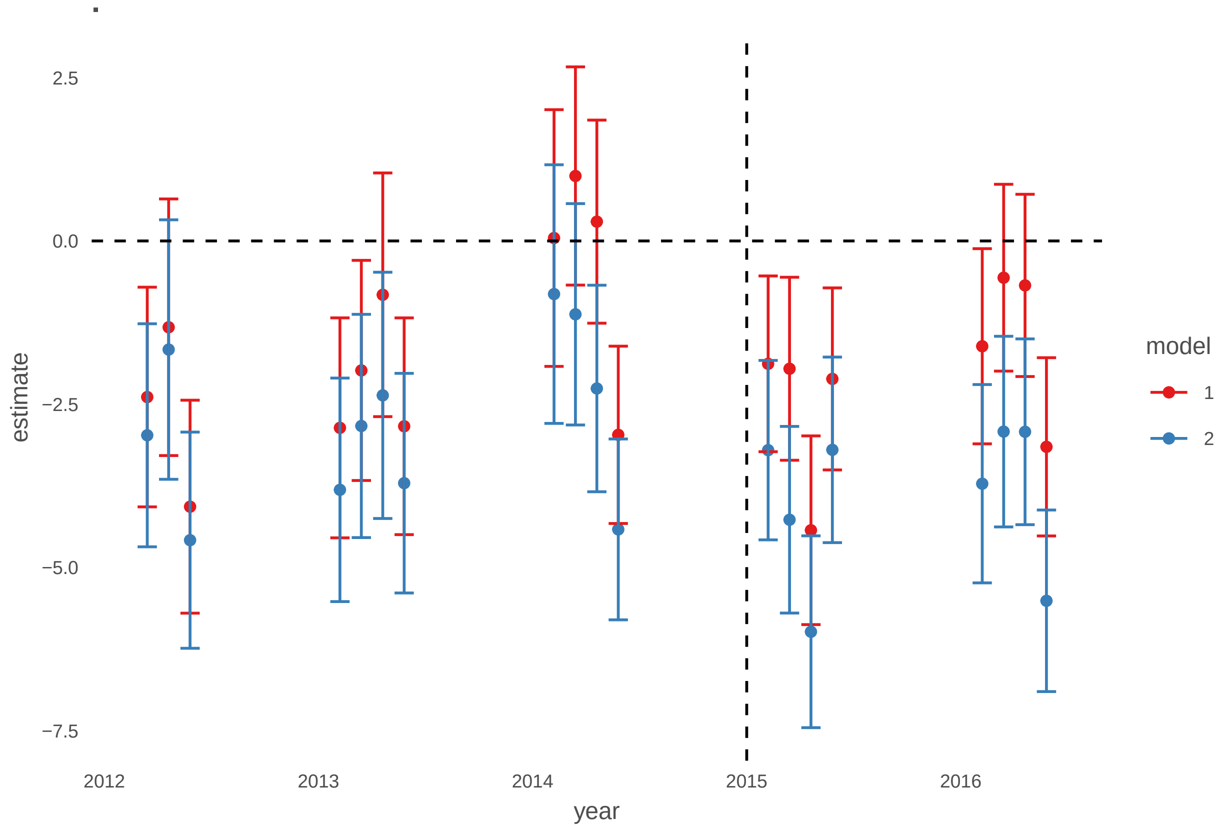
Table S2: Fishing hours from GFW for longliners (n = 203; 88 control, 115 treatment).. Asterisks indicate significance levels. Numbers in parenthesis represent heteroskedastic-robuste standard errors.

	Dependent variable:			
	hours			
	(1)	(2)	(3)	(4)
post	-1.241^{***}	-1.026^{***}	-0.944^{***}	13.292^{***}
	(0.116)	(0.116)	(0.157)	(0.330)
treated	2.595^{***}	2.708^{***}	2.639^{***}	1.493^{***}
	(0.101)	(0.101)	(0.102)	(0.116)
post:treated	0.053	-0.069	-0.029	-0.060
	(0.133)	(0.133)	(0.134)	(0.135)
Constant	39.032^{***}	38.105^{***}	38.388^{***}	36.993^{***}
	(0.089)	(0.131)	(0.150)	(0.178)
Month FE	No	Yes	Yes	Yes
Year FE	No	No	Yes	Yes
Flag FE	No	No	No	Yes
Observations	227,873	227,873	227,873	227,873
R^2	0.010	0.015	0.016	0.030
Note:				*p<0.1; **p<0.05; ***p<0.01

6.1 Other model specifications

6.1.1 Year as dummy variables





estimate
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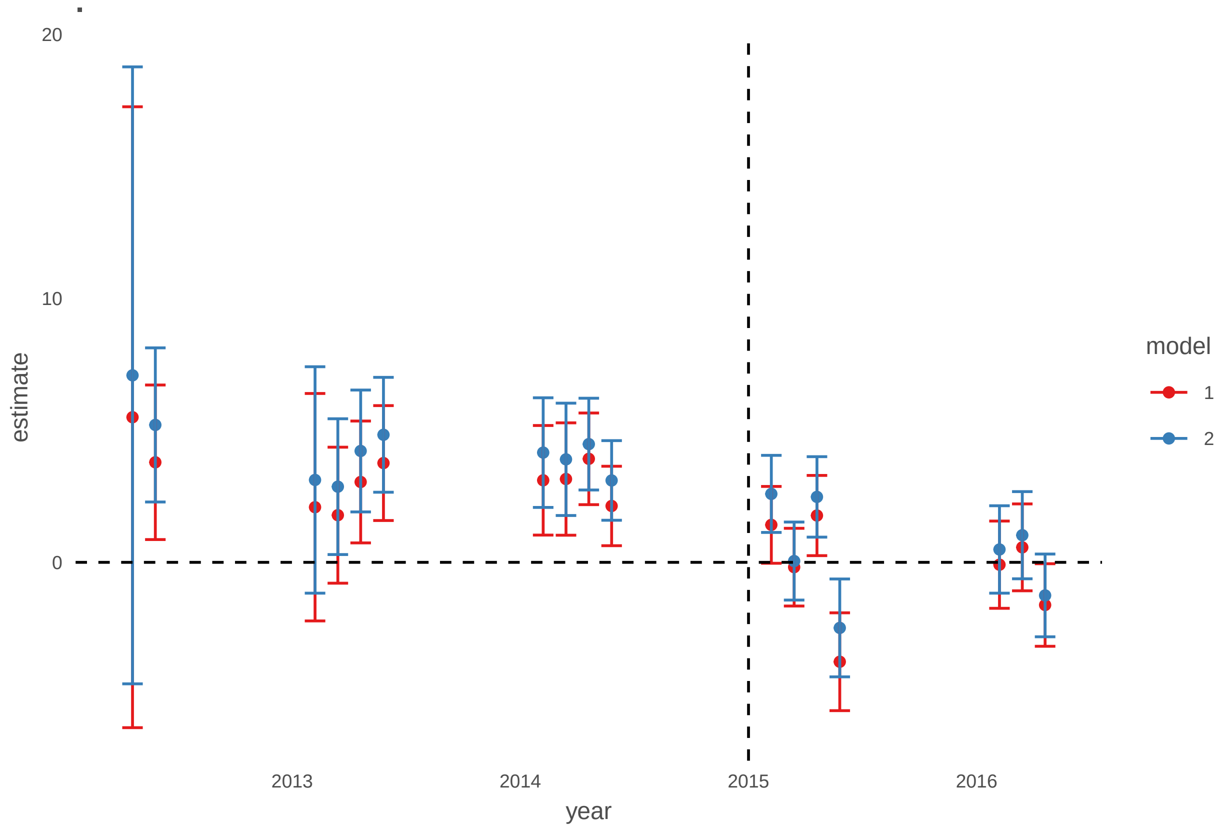
0.0
−2.5
−5.0
−7.5
2012
2013
2014
2015
2016
year
model
1
2



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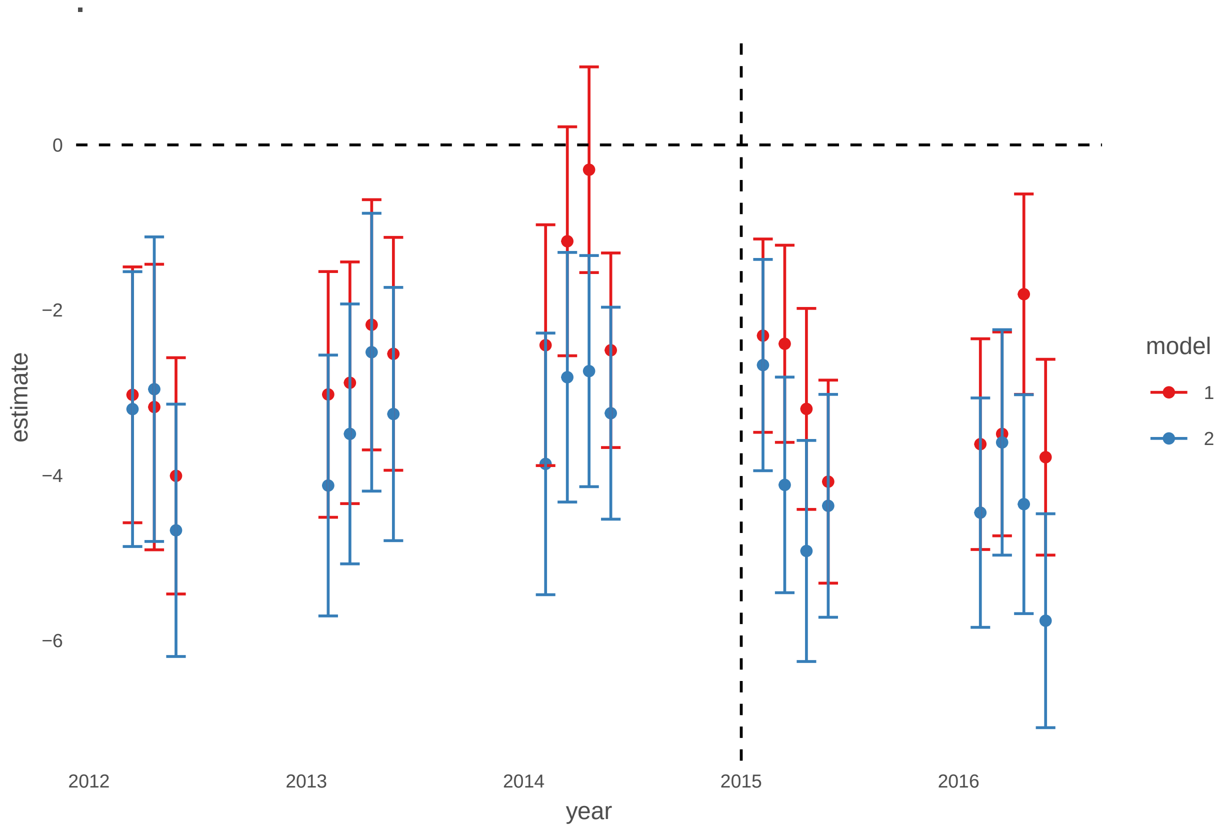
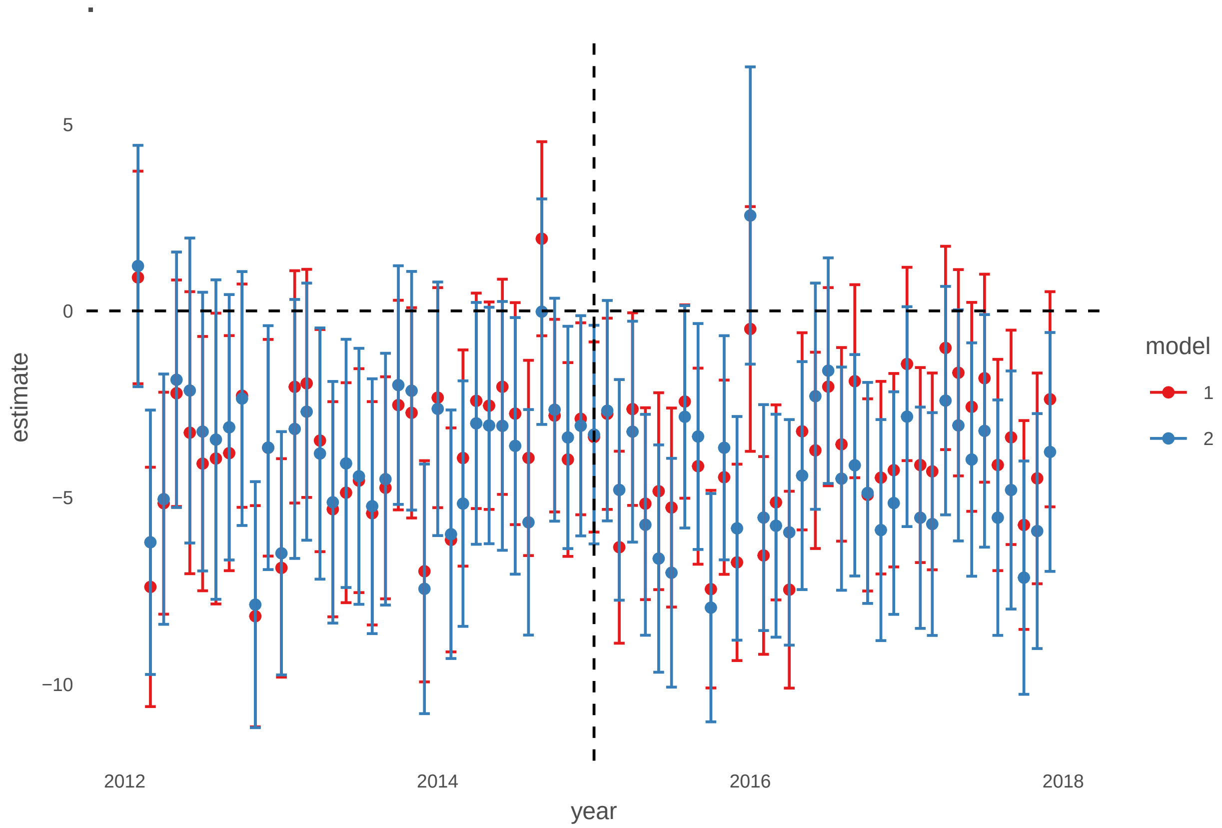
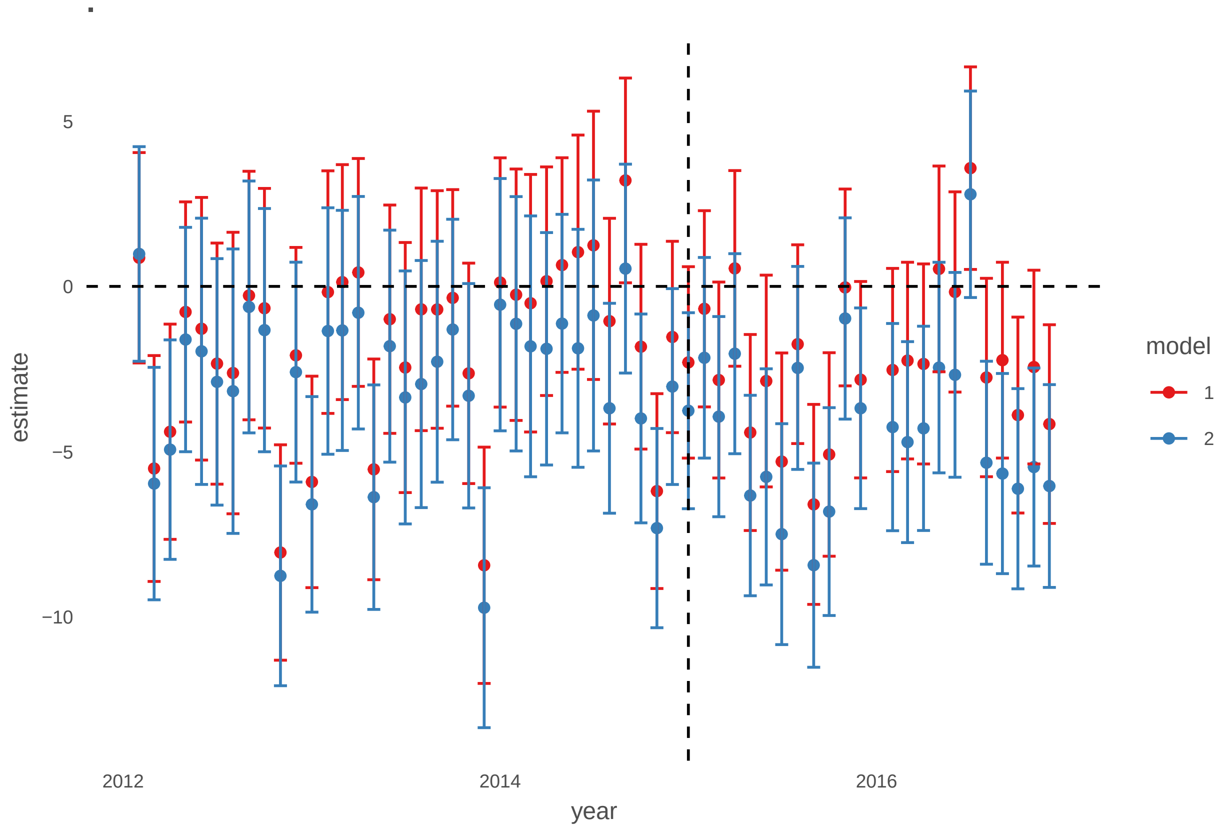

0
−2
−4
−6
estimate
2012
2013
2014
2015
2016
year
model
1
2


5
0
−5
−10
estimate
2012
2014
2016
2018
year
model
1
2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0
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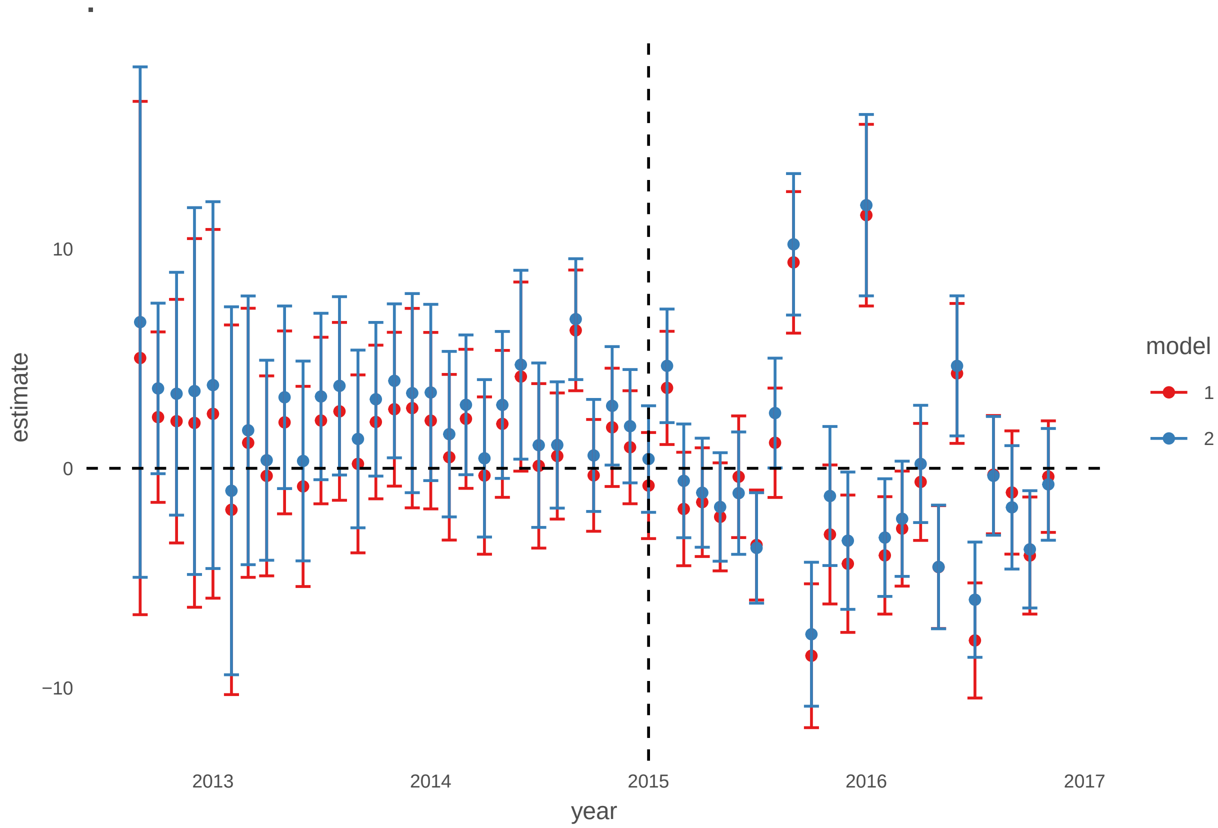
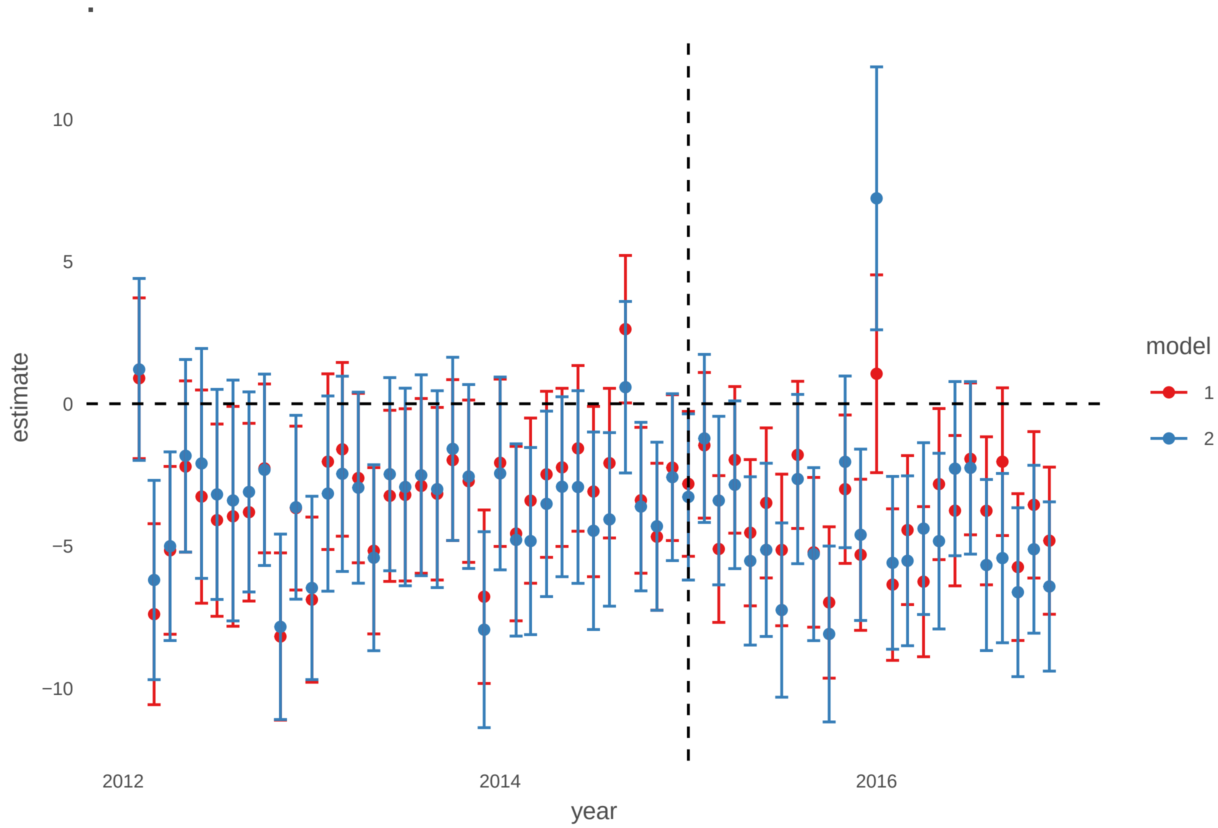
−10
estimate
2012
2014
2016
year
model
1
2



312



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