The blue paradox: Preemptive overfishing in marine reserves

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Most large-scale conservation policies are anticipated or announced in advance. This risks the possibility of preemptive resource extraction before the conservation intervention goes into force. We use a high-resolution dataset of satellite-based fishing activity to show that anticipation of an impending no-take marine reserve undermines the policy by triggering an unintended race-to-fish. We study one of the world's largest marine reserves, the Phoenix Islands Protected Area (PIPA), and find that fishers more than doubled their fishing effort once this area was earmarked for eventual protected status. The additional fishing effort resulted in an impoverished starting point for PIPA equivalent to 1.5 y of banned fishing. Extrapolating this behavior globally, we estimate that if other marine reserve announcements were to trigger similar preemptive fishing, this could temporarily increase the share of overextracted fisheries from 65% to 72%. Our findings have implications for general conservation efforts as well as the methods that scientists use to monitor and evaluate policy efficacy.

blue paradox | overfishing | marine reserves | marine protected areas

Perhaps the most policy-relevant insights from environmental economics are that people respond to incentives and that these responses can be leveraged to design effective environmental policies. This basic notion underpins the near-ubiquitous calls by economists to put a price on carbon, create markets for ecosystem services, and implement conservation measures that harness human behavioral responses.

An equally important—but understudied—class of incentive challenges is brought on by the anticipation of a new policy. Because policy aspirations and announcements allow people to anticipate changes on the horizon, an incentive for preemptive behavior could undermine the very outcome that was targeted in the first place. A particularly striking example involves the gun control debate in the United States, where calls for tougher legislation in the wake of mass shootings like the 2012 Sandy Hook massacre have led to a surge in firearm sales (1, 2).

In the environmental domain, there is perhaps no better illustration of preemptive behavior undermining a policy objective than the US Endangered Species Act of 1973 (ESA). The red-cockaded woodpecker, for example, paradoxically saw declines in its habitat after it gained protected status under the ESA. Knowing that they now risked costly land-use restrictions upon settlement by woodpecker colonies, private landowners in North Carolina and elsewhere were incentivized to deforest their properties in advance of this happening. Similar anticipatory behaviors have been observed with regard to land development (3), water withdrawals (4), and climate change policy (5), and are underpinned by a rich theoretical literature on the underlying mechanisms (6–9). In environmental economics, this phenomenon has been generalized as the "green paradox" (7). For example, the green paradox suggests that credible climate policy in the future will induce more rapid extraction of fossil fuels in the present. The very prospect of climate policy can thereby engender a perverse acceleration in climate change.

This line of reasoning suggests that the anticipation of a new conservation policy may give rise to a set of incentives that is distinct from—but possibly just as important as—the incentives arising from the policy's implementation. While this kind of preemptive behavior has been well-documented for landowners, gun owners, and owners of natural resource extraction rights, it has not been studied in the commons. For vast swaths of the ocean, no single owner has exclusive rights and so must compete against others for extraction. This raises an important and unanswered question: Does the announcement of a future conservation policy lead to preemptive extraction, even in a commons?

We study this question in the context of large-scale marine conservation. An estimated 7% of the world's ocean is now designated as marine protected areas, or MPAs, a marine spatial management approach that restricts human activities (10). Many of these areas are further designated as marine reserves, a special type of "no-take" MPA that prohibits all extractive fishing. The last few decades have witnessed explosive growth in MPA numbers and coverage (Fig. 1). While the trend toward ever-increasing numbers has tapered off recently, this has been more than offset by a smaller quantity of extremely large MPAs that are greater than 100,000 km^2 in size. These "mega-MPAs" require complex negotiations between multiple stakeholders (often including various governments), but such is their stature that the largest 20 MPAs now account for 70% of total MPA coverage (10).

The many and varied motivations for implementing MPAs—and specifically no-take marine reserves—include conserving habitat, increasing abundance of fish inside the marine reserve, and increasing the production and spillover of fish outside the marine reserve, among others. Proponents argue that marine reserves are a critical tool in modern conservation efforts and that sealing off large parts of the world's oceans from commercial fishing pressure will enable threatened fish stocks and entire marine ecosystems to rebuild and replenish. This rationale

Fig. 1. MPA expansion over time, including no-take marine reserves and other types of MPAs (1900–2017). The figure shows the cumulative quantity and coverage of MPAs at a global level. Source: Ref. 10.

hinges on the idea that fishing pressure inside the marine reserve will be reduced to zero upon implementation. Evidence from some of the world's largest marine reserves appears to confirm this necessary condition.

For example, consider the Phoenix Islands Protected Area (PIPA), a 408,459 km^2 swath of central Pacific ocean roughly the size of California (Fig. 2), which is the focus of this study. As part of the island nation of Kiribati's exclusive economic zone (EEZ), PIPA is perhaps the most well-studied and celebrated marine reserve in the world (11–17). Before receiving full protected status, PIPA existed as a lesser form of MPA that prohibited fishing activity in only a small fraction of its waters. This management status proved to be a matter of some confusion and controversy within the international conservation community (18, 19). However, following a period of increasing political pressure and funding provisions, PIPA was finally closed to all commercial fishing activity on January 1, 2015—thus establishing its new designation as an official no-take marine reserve. The impact was dramatic. Fishing activity within PIPA swiftly collapsed to zero after the reserve was implemented (11, 12).

Even when marine reserves like PIPA are ultimately implemented, the path toward designating protected status is almost always long and arduous. It typically involves public meetings, scientific studies, expert testimony, fisher rebuttal, and blue-ribbon panels. Successful implementation is thus the product of months or even years of policy deliberation and planning. While this process promotes transparency and stakeholder engagement, it also serves as a palpable signal to existing resource extractors that the area may be closed to fishing in the future. This policy telegraph opens up the possibility for a paradoxical race-to-harvest well before the marine reserve is actually enforced.

Whether the anticipation of a possible future closure alters incentives to extract fish in the short run is an issue that has received almost no attention in the popular or scientific literature. On the one hand, one may expect that anticipation of a future marine reserve triggers a race to fish, just like the race to purchase guns or to convert endangered species habitat. On the other hand, in an open access setting, we might expect fishing effort to already have been applied to the point of zero profit, suggesting that still further application of effort would not be undertaken. Yet if marine reserve anticipation does trigger a race to fish, this could be a potentially serious cause for concern. After all, the primary purpose of most marine reserves is to rebuild fish stocks and replenish the local ecosystem by reducing fishing pressure.

Here we provide an empirical test of a "blue paradox"—that is, whether anticipation of a marine reserve catalyzes a race-to-fish that undermines the intended conservation objectives. Until recently, data limitations would have made conducting such a test virtually impossible. One needs fishing effort data at a sufficiently high temporal frequency (and covering a long enough time horizon before the marine reserve implementation) to capture anticipatory effects. Moreover, the data must be of sufficiently high spatial resolution such that fishing effort is observed both over the potential marine reserve itself and over an appropriate counterfactual control region. There are two reasons that this latter requirement is especially important for causally estimating how conservation outcomes are affected by preemptive fishing behavior. First, in most fisheries (including PIPA), fish stocks are not directly observed. Second, depending on where a fishery lies in its transitional dynamics (e.g., if it is out of steady state), an increase in fishing effort can coincide with higher or lower stock levels. These two facts imply that simply observing an increase in PIPA fishing effort levels alone would not provide a sufficient basis for concluding that fish stocks are also falling there. The detection of a blue paradox therefore requires a research design that can identify an exogenous increase in fishing effort relative to what would have occurred without the announcement; only then can we conclude that the increase in fishing effort led to a reduction in the fish stock. In our setting, this is best achieved by establishing a control region with stock dynamics mirroring that of PIPA before the policy announcement or the onset of any preemptive harvesting. The control region therefore serves as a plausible counterfactual of how fishing effort and fish stocks would have evolved in the absence of the impending marine reserve. A blue paradox can then be said to have occurred if PIPA fishing effort increased relative to that of the control area during the lead-up to policy implementation. We discuss this so-called difference-in-differences research design—also referred to as a before-after-control-impact approach—and our selection of a control area in greater detail below.

Fishing effort data with the necessary temporal and spatial resolutions were recently made available through the Global Fishing Watch (GFW) initiative (12), which provides satellite-derived estimates of global fishing effort in unprecedented detail. More broadly, our use of GFW data enables statistical analysis at temporal and spatial resolutions significantly exceeding that of all previous empirical studies on terrestrial preemptive resource extraction. The novelty of these data therefore allows not only detection of a blue paradox but also richer statistical analyses than previously possible in the broader study of preemptive resource extraction.
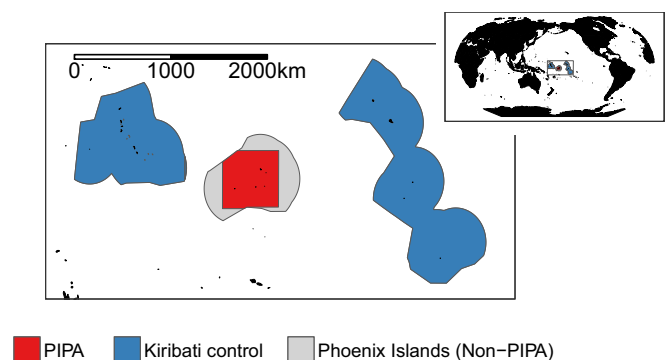
Fig. 2. Map of the Kiribati EEZ. The shaded areas denote the treatment region PIPA (red), a control region made up of the Line and Gilbert Islands (blue), and the remaining part of the Phoenix Islands not used as part of the control region (gray).

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To preview our findings, this paper makes three primary contributions. First, we document that policy-preemptive resource extraction is possible even in a commons such as the ocean where no single fisherman has the exclusive right to extract the resource. We discuss several potential mechanisms underlying this surprising result. Second, we precisely quantify the magnitude of increased fishing effort due to anticipatory fishing behavior; we show that there was a 130% increase in fishing effort due to policy anticipation, which is equivalent to 1.5 y of banned fishing following marine reserve implementation. We also extrapolate these findings to a global scale to investigate the possible implications if marine reserves were significantly expanded to the levels recommended by prominent conservation groups. These results contribute to the ongoing debate about the conservation efficacy of marine reserves and MPAs more generally and offer a previously unexplored reason for the puzzlingly poor track record of many of the world's MPAs (e.g., refs. 20–22). We argue further that failing to account for these effects likely biases many of the existing approaches that scientists use to measure the conservation efficacy of marine reserves and MPAs in the first place. Third and finally, we demonstrate the direct policy relevance of high-resolution satellite data as they apply to natural resource governance.

Results

Our empirical results focus on fishing effort within PIPA in the lead up to its formal marine reserve designation on January 1, 2015. We focus on PIPA because it has received considerable attention in both the popular and scientific literature and has been generally lauded as one of the most successful implementation case studies among marine reserves globally (e.g., refs. 11–17). PIPA is also among the largest marine reserves in the world, which is relevant given the growing prominence of mega-MPAs described in the Introduction. According to official sources, the primary motivations for why PIPA was originally earmarked for protected status are as follows: (*i*) PIPA's remoteness, size, and the healthy conditions of its shallow-water reef fish populations meant that it could act as an ecological benchmark for remote oceanic areas, and (*ii*) PIPA was subject to significant global and regional threats, including climate change, increasing human populations and mobility, and high fishing pressure on depleted large pelagics like tuna (23).

As noted above, to determine whether anticipation of PIPA caused a blue paradox over that region, we must establish a counterfactual in the form of a suitable control region. That is, we must define a control region that plausibly exhibits the same trends in fishing effort over PIPA, had the marine reserve never been implemented or anticipated. We can then measure any potential blue paradox effects using a standard difference-in-differences approach, which evaluates relative changes in effort across these two regions over time. This relative comparison allows us to abstract from complications associated with endogenous correlations in fishing effort and fish stocks, which depend on the transitional dynamics of the fishery as it moves toward steady state. A difference-in-differences approach also avoids spurious conclusions based on absolute changes in PIPA fishing effort that may be driven by other, unobservable factors.

We define our control region as the waters around the Line and Gilbert Islands, the other two island chains of Kiribati, shown as shaded blue areas in Fig. 2. This "Kiribati control" region is chosen for three reasons. First, it is within the Kiribati EEZ and so is subject to similar national and international policies that affect fishing effort. Second, it is relatively far removed from the Phoenix Islands region that encloses PIPA at distances between 500 and 1,000 km, which helps to reduce the potential for fishing spillovers across regions. Third, the PIPA and the Kiribati control regions have mostly the

same target fish stocks—skipjack, yellowfin, bigeye, and albacore tuna—which further helps to preserve an apples-to-apples comparison (24). Despite these considerations, this region may still not serve as a valid control for PIPA if its fishing trends diverged from that of PIPA's had there not been anticipation of a future marine reserve. Unfortunately, one can never test this "parallel trends" assumption directly. Instead, we present the most stringent possible indirect test by showing that there were no differential fishing effort trends between PIPA and the Kiribati control regions before the anticipation of the PIPA marine reserve.

Our key empirical results are depicted in Fig. 3. Each data point shown in *Top* indicates total daily fishing hours per 1,000 km^2 within the PIPA (red) and Kiribati control (blue) regions during the January 1, 2012 to December 31, 2016 period. These high-frequency observations are aggregated from GFW's boat-level data, which use various classification algorithms to assign boat activity as either "fishing" or "not-fishing" (12). January 1, 2015 marks the first date of PIPA implementation (dashed vertical line). September 1, 2013 corresponds to the earliest mention of a possible full closure of PIPA to fishing according to global Google search activity for the terms "Phoenix Islands Protected Area" and "closure" (19). To reduce noise, we fit a restricted cubic spline (solid lines) to flexibly model the relationship between fishing effort and time separately for each region and for the periods before and after PIPA implementation. Our spline functions use six evenly spaced knots (25).

We highlight three results from Fig. 3. First, trends in fishing effort from January 2012 to August 2013 in PIPA and the Kiribati control regions are parallel and nearly identical. Furthermore, there appears to be no change in fishing effort for the Kiribati control region after August 2013. This suggests that fishing around the Line and Gilbert Islands serves as a reasonable control for fishing in PIPA. Second, consistent with the presence of a blue paradox, there is a clear divergence in fishing effort after public awareness about the closure began to grow (around September 1, 2013) and before the start of PIPA implementation on January 1, 2015. While we cannot directly measure boat captains' expectations about a possible future closure, this surge in preemptive fishing within PIPA relative to the control region occurred presumably because fishing vessels anticipated the eventual arrival of the marine reserve before its implementation. Third, fishing in PIPA falls to nearly zero after the marine reserve is established, consistent with previous literature (11, 12), but fishing in the control region is little affected. This again confirms the validity of our control area.

Fig. 3, *Bottom* examines whether this difference in preemptive fishing is statistically significant. It shows the estimated restricted cubic spline function for the difference in fishing effort across the PIPA and control regions (see Eq. **1** in *Materials and Methods*). The associated 90% CI, which is robust to serial correlation and heteroscedasticity of arbitrary form within a 60-d time window (26), shows that this difference in fishing before PIPA implementation is statistically significant for the period starting late 2013 until PIPA implementation.

We explore a variety of alternate specifications to confirm the robustness of our results in the *SI Appendix*. For example, the divergence in fishing effort between PIPA and the Kiribati control region seen in Fig. 3 may also have arisen if other fishing effort determinants such as fish biomass, fishing costs, and fish prices were diverging across these regions during this exact same period. While we do not observe fish stock and fishing costs at the required spatial and temporal resolutions, we show that controlling for region-specific sea surface temperatures—a key environmental condition that affects both biomass and the cost of fishing—has no impact on the shape of the estimated spline

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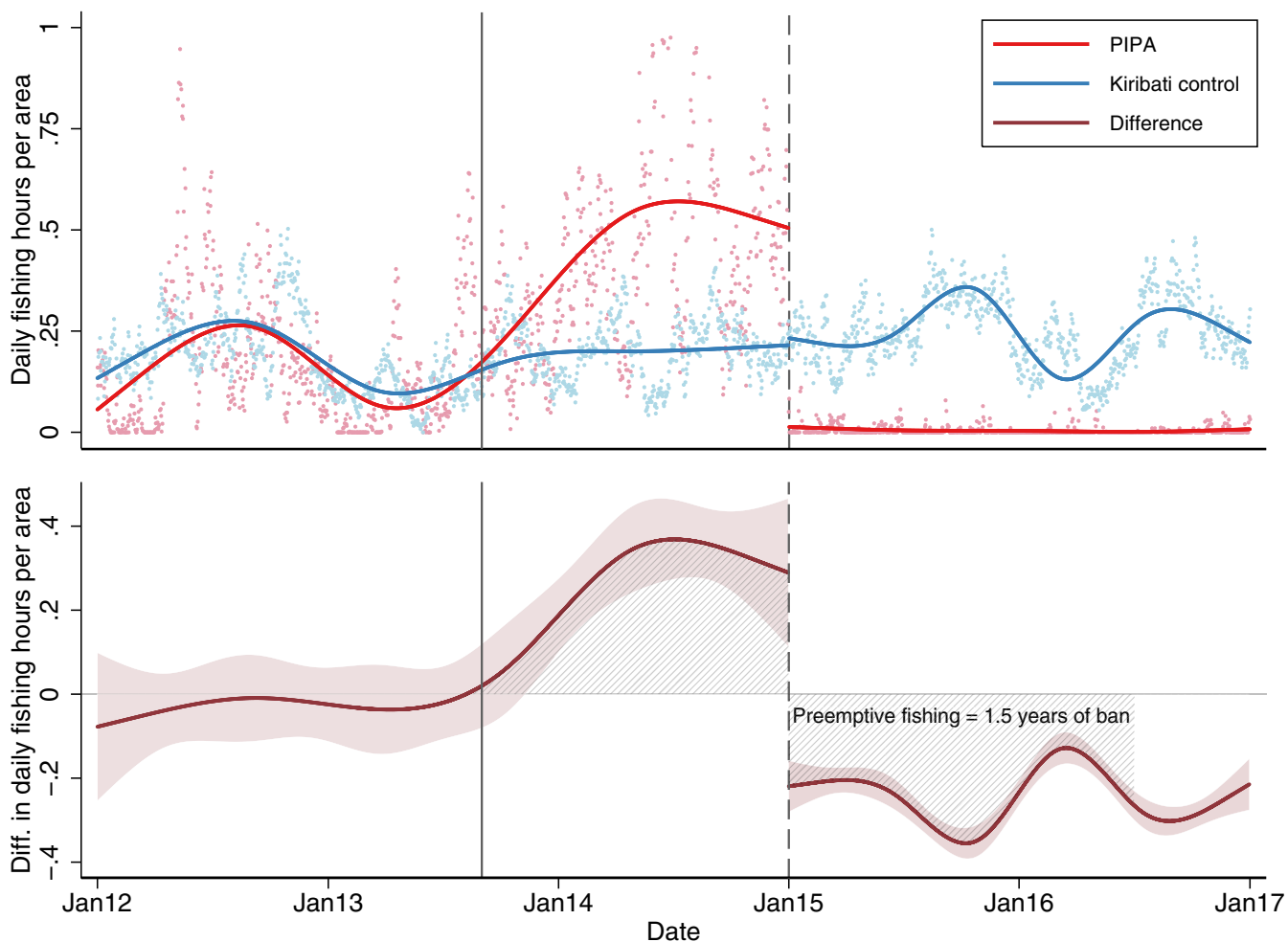
Fig. 3. (*Top*) Total daily fishing hours per 1,000 km² over PIPA (red) and Kiribati control region (blue). Vertical solid line marks first earliest PIPA mention from a Google search (September 1, 2013). Vertical dashed line marks the start of PIPA implementation (January 1, 2015). Solid lines show separately estimated restricted cubic spline functions for the periods before and after PIPA implementation and within PIPA and Kiribati control areas. The panel *y* axis has been truncated at 1 to aid visual inspection, though the spline functions are estimated on the full sample of observations. (*Bottom*) Point estimates (purple) from a restricted cubic spline function for the difference in daily fishing between PIPA and Kiribati control regions, before and after PIPA implementation (see Eq. **1** in *Materials and Methods*). Shaded area (purple) shows serial correlation and heteroscedasticity robust 90% CIs. Hatched shaded area before PIPA implementation shows total preemptive fishing. Hatched shaded area after PIPA implementation shows equivalent avoided fishing.

functions. Furthermore, this response is robust to the inclusion of monthly effects to account for seasonally in biomass and fishing costs (*SI Appendix*, Fig. S1). Regarding fish prices, while we also do not observe these at the necessary spatial temporal resolution, we have little reason to believe they diverge for fishers in the two regions. This is both because the fished species are the same in both regions and because most of the fishing is done by foreign vessels who are selling to common global markets. Finally, we illustrate the robustness of our response function to different statistical modeling assumptions and classification procedures. The estimated spline function is largely unaffected by the number of knots selected (*SI Appendix*, Fig. S2) or by different assumptions regarding the extent of serial correlation in the error term (*SI Appendix*, Fig. S3). Finally, our core result is robust to the use of an alternative measure of fishing effort, based on a neural net classification procedure (12) (*SI Appendix*, Fig. S4).

Having established the statistical significance of our results, we also consider their economic and conservation significance. The extra preemptive fishing effort in PIPA before the ban represents a 130% increase in fishing relative to the Kiribati control

region. To interpret the magnitude of this preemptive effect, the gray-hatched area before PIPA implementation in Fig. 3, *Bottom* shows the total amount of preemptive fishing in PIPA relative to the Kiribati control region implied by the point estimates of the spline function. The gray-hatched area after PIPA implementation has the same area and thus can be used to determine how many days of the fishing ban would be needed to offset the extra preemptive fishing. The amount of extra fishing effort due to the blue paradox is equivalent to fishing effort avoided during the first 1.5 y of the marine reserve.

Discussion

Humans are extremely adept at forming expectations and responding to new information. This behavioral response plays out in many sectors of the economy, from finance to housing and labor markets to gun control and to endangered species protection. We have empirically demonstrated that large-scale conservation efforts are also susceptible to preemptive behavior that undermines intended policy goals. Specifically, we find that fishing effort in PIPA more than doubled (relative to a plausible counterfactual) in anticipation of its impending closure.

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Despite the magnitude of these numbers, there are reasons to view our empirical results as a lower bound on the blue paradox effect. Recall that PIPA had already been designated as a lesser form of MPA in 2008, 7 y before its ultimate designation as a no-take marine reserve in 2015. Consistent application of our theory suggests that this initial MPA designation—with restrictions on some forms of fishing activity—could have triggered a preemptive increase in fishing effort of its own. Our dataset begins in 2012, and we are therefore not able to observe potential effort changes during this earlier period. However, it seems reasonable to infer that accounting for this initial MPA designation could reveal a larger blue paradox effect in total. Similarly, while we have focused on short-term changes in fishing effort, a blue paradox may also foment long-term harms. For example, the preemptive harvesting phase could trigger irreversible habitat destruction or push fish stocks below important population thresholds. Passing such tipping points would mean that the required recovery phase in terms of biomass significantly exceeds the equivalent harvesting phase we have calculated. In the case of PIPA, it may well be that the preimplementation surge in fishing effort corresponds to much more than 1.5 y of the enforced ban. A related concern is that our analysis largely abstracts from the starting conditions for fish biomass and demographics. Whether a blue paradox causes fish stocks to collapse depends importantly on stock conditions at the onset of the preemptive harvesting phase. For any given degree of additional fishing, a less healthy stock will intuitively face higher risk than a more healthy stock. Again in the specific case of PIPA, evidence suggests that many of the key target species were already being overfished before the onset of the blue paradox that we observe here (e.g., ref. 27). This would not only imply increased risk to the long-term stability of these fish stocks but could be viewed as another reason why our empirical estimates are actually conservative.

Such important caveats notwithstanding, a larger question remains: How might the blue paradox play out on the global stage? Does the unintended race-to-fish—catalyzed by anticipation of impending closures—undermine the very conservation goals that MPAs and marine reserves seek to attain? A back-of-the-envelope calculation suggests significant short-term costs. As a thought experiment, suppose that each major fishing area (MFA) in the world, as classified by the United Nations Food and Agriculture Organization (FAO), expanded its existing MPA coverage to achieve the International Union for Conservation of Nature (IUCN) target of placing 30% of all marine waters in no-take marine reserves by 2030 (28). Fig. 4 shows the current level of protection in each FAO relative to this target (10,
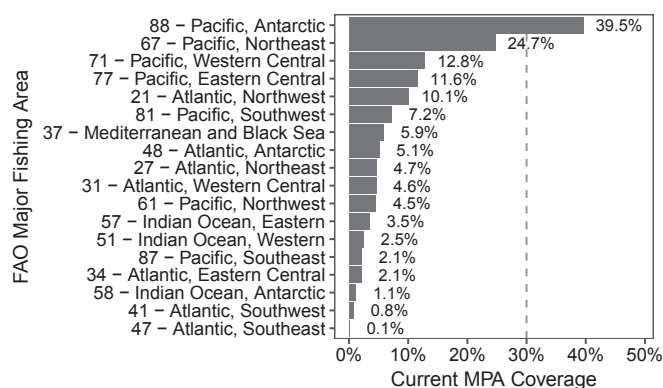
29). The increase in MPA coverage needed to achieve this target varies widely across the world's oceans, though most regions would require significant expansion. We note further that a majority of the world's existing MPAs are not yet full no-take marine reserves, meaning that the necessary increase in protection is even higher than indicated here. Nonetheless, if areas affected by such a protected area expansion were to experience the same short-term surge in fishing effort that we have empirically documented for PIPA—as a fraction of current fishing pressure proportional to the expansion size—then we can estimate the change in overall fishing pressure attributable to this preemptive behavior.

Fig. 5, *Top* shows the current catch-weighted mean fishing pressure across all fisheries in each FAO MFA relative to the fishing pressure that achieves maximum sustainable yield (27). Following standard notation, we denote this ratio as F/F_{MSY}. The bottom panel of Fig. 5 shows the fishing pressure that we could expect during the preemptive harvesting phase following new marine reserve announcements. In some areas, the change is barely noticeable. In the Northeast Pacific, for example, the fishing mortality would increase by a mere 7% while staying below the sustainability target of $F/F_{MSY} \leq 1$. But in the some regions, this preemptive harvesting behavior could mean the difference between achieving a sustainable fishing level and an unsustainable one ($F/F_{MSY} > 1$). And for about one-third of the FAO MFAs, this could imply moving from only moderate overfishing to a more severe overfishing situation ($F/F_{MSY} > 2$). On a global scale, the percentage of fisheries experiencing overfishing would increase from the currently estimated level of 65% to a level of 72%.

While we emphasize that these hypothetical increases in global fishing effort are only transient until the marine reserves are actually implemented, we again note the conservative basis for our empirical estimates. In particular, we have abstracted away from the potentially destabilizing effects of tipping points and longer-term dynamics, such as poor initial stock conditions and the passing of key population thresholds. We have also made the conservative assumption in our extrapolation exercise that the preemptive increase in fishing effort does not apply to any existing MPAs, even though many of these would first need to be upgraded to full no-take status to achieve the IUCN's coverage goal. Similarly, while our global extrapolation exercise includes all global fisheries, the strength and even existence of a blue paradox may be affected by context-specific governance effectiveness, stakeholder inclusiveness in the MPA design process, and existing fishery management systems (particularly those that confer property rights a la total allowable catches and individual transferable quotas). Such issues are beyond the scope of this paper, but are deserving of future research in the form of specialized stock assessments and population modeling efforts, among other analyses.

Although the global consequences of the blue paradox could be significant for large-scale marine conservation, it bears repeating that preemptive resource extraction in the ocean may require a more nuanced explanation than its terrestrial counterpart. The primary mechanism underlying preemptive behavior on land is the presence of secure property rights. Private landowners are incentivized to act in advance of legislation like the ESA because it alters the discounted revenue streams that they expect to obtain from their properties. It is better to sell or develop the resource while that is still possible to be proscribed once legislation is enforced. Simply put, landowners face a "use it or lose it" choice and predictably opt for the former. In contrast, property rights in the ocean are usually far from secure. Standard economic theory tells us that for open access resources, any excess rents (i.e., profits) from fishing should be dissipated through competition. This raises an important question: Why would anticipation of an impending
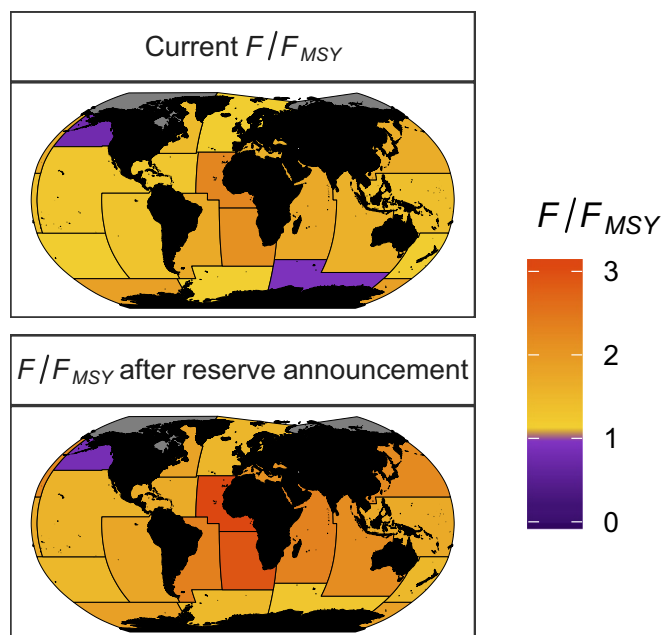
Fig. 4. Current MPA coverage by FAO Major Fishing Area, including both no-take marine reserves and other types of MPAs. IUCN 2030 target of 30% no-take marine reserve coverage shown as vertical dashed line.

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Current F/F_{MSY}

F/F_{MSY} after reserve announcement

Fig. 5. Current global fishing pressure by FAO MFA (*Top*) and expected fishing pressure during a preemptive mining phase following the announcement of no-take marine reserve expansions to achieve the IUCN 2030 target of 30% coverage (*Bottom*).

marine reserve trigger a surge in fishing activity if that region is already being fished at an effort rate that returns zero profit?

We speculate that there are at least three reasons why the standard economic intuition of open-access fisheries might not hold in the case of marine reserves like PIPA and would therefore give rise to a blue paradox. The first is that the zero profit condition for open access resources holds only in steady state. If the resource is out of steady state, for example as it is in the process of being fished down, the announcement of a new closure could trigger changes in the pace of entry, consistent with a blue paradox. The second explanation is that an area earmarked for protected status could already have some kind of formal access rights associated with it. Indeed, PIPA is located within Kirbati's EEZ, which has a well-described set of rules governing extraction from its waters. This may have conferred additional privileges for a select group of fishers by restricting access to outsiders before the enforcement of the no-take rule. A fishery may be similarly subject to the restrictions imposed on it by international agreements between countries. For example, purse seine fishing in the Western and Central Pacific is regulated by the Nauru Agreement (30), which specifies total allowable fishing effort in the form of an annual Vessel Day Scheme (VDS). As a signatory country, Kiribati is allowed to sell off its quota of vessel days to fishers. It is also allowed to trade vessel days with other member nations in the event of a quota shortage or glut. However, total purse seine fishing within its waters (including PIPA) remains bound by the VDS and would thus have conferred some form of property rights to eligible vessels. That said, the VDS program in Kiribati did not include the sizable longline fleet during the study period; though Kirbati began using a longline catch quota management system in 2017 (31). This implies that there were partial property rights over the resource during the study period. The third reason that the standard open access model might not apply is that fishers naturally secure quasi-property rights for themselves through a process of specialization and spatial knowledge. Certain vessels or fleets may hold virtual monopoly power in patches of the ocean simply by virtue of the fact that they

have been fishing there longest, know these waters better than their competitors, and coordinate among each other to maintain the status quo. Supporting evidence for this idea can be gleaned by looking at the relative contributions to total PIPA fishing effort—aggregated at the fleet level—before and during the preemptive harvesting phase. We find that any post-September 2013 increase in fishing effort tends to be proportional to a fleet's existing footprint in PIPA (*SI Appendix*, Figs. S5 and S6). Moreover, this result is robust to defining a fleet at both the country and owner (i.e., company) level. Taking the country-level fleet aggregation as an example, South Korean vessels dominated fishing effort within PIPA in both the "before" and "during" periods. Such a proportional response is at least consistent with a possible "spatial knowledge" equilibrium, in which particular vessels were able to dominate fishing activity within PIPA, securing themselves a kind of de facto property right in the process.

Fully determining the precise causal chain underlying the 2014 surge in PIPA fishing activity—the role of quasi-property rights and other factors—is a subject for future research. While the evidence presented in this paper strongly supports a blue paradox event, our larger goal is to generate a discussion about the role of information and incentives in the anticipation of future conservation policies. If the evidence presented here about preemptive overfishing in prospective marine reserves and other MPAs holds generally, then it would have important implications not only for their biological and ecological efficacy, but also for the methods that scientists use to establish this efficacy in the first place. On the biological and ecological side, we have shown that a blue paradox could lead to a significantly impoverished starting point for the affected reserves. It could even lead to a long-term decline in abundance if, say, the surge in fishing activity pushes fish stocks below some minimum population threshold. On the scientific measurement side, the threat of a blue paradox calls into question monitoring approaches that rely on direct comparisons of fish abundance. For example, preemptive fishing would affect the interpretation of comparisons of abundance before versus after policy implementation, or inside versus outside the newly-established reserve. Such simple approaches are not robust to an endogenous change in fishing effort—a la the blue paradox—and would almost certainly be biased as a result. Concerns about this potential bias take on a greater significance in light of an ongoing debate about the true conservation efficacy of marine reserves and MPAs. Indeed, the empirical evidence in their favor remains worryingly equivocal, despite their increasing popularity and the impressive growth in total MPA coverage (20–22, 32, 33). It may well be that the puzzling prevalence of so-called "paper parks" has as much to do with scientists mistakenly picking up on a blue paradox effect, as it does with the inability of local authorities to police enforcement. Importantly, we also demonstrate with PIPA that size is not a sufficient bulwark against a blue paradox. This too pushes back against conventional wisdom in conservation circles, where the move toward increasingly large marine reserves is underscored by scientific arguments that suggest conversation efficacy is increasing in marine reserve size (e.g., ref. 20).

Taking a step back, these results contribute to the growing appreciation by policy makers and scientists to proceed cautiously when environmental policies and economic incentives are misaligned. In the specific case of marine reserves at least, we suggest a need to reduce the duration of policy design periods. We end on a hopeful note, recognizing that the evidence presented herein would have been impossible only a few years ago due to data limitations. Thanks to the advent of incredibly rich satellite data provided by the likes of GFW, we now have the means to address previously unanswered questions and improve management of our natural resources accordingly.

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Materials and Methods

Data Sources. Fishing effort estimates come from GFW (12). We aggregated total fishing effort by day, region (inside PIPA or in the control region consisting of the Line and Gilbert Islands, which are also part of the Kiribati EEZ), and by fishing vessel flag country and vessel owner. The data come from the time period between January 1, 2012 and December 31, 2016.

Daily sea surface temperature (SST) data over the same 2012–2016 sample period come from NOAA (34) and were obtained through *Google Earth Engine* (GEE) (35). The SST data are observed at a gridded spatial resolution of $0.25°$ ($\sim$25 km) and were aggregated to regional means (i.e., PIPA and Kiribati control regions) within GEE before being exported to disk.

Current MPA coverage estimates by FAO MFA were calculated by overlaying all currently designated marine and coastal MPAs from the Protected Planet MPA database (10) with the current spatial coverage of each FAO MFA (29). Coverage estimates include both no-take marine reserves and MPAs with less restrictive policies.

Empirical Methods. We model total daily fishing hours per 1,000 km^2, y_{it}, for region i during date t as

$$y_{it} = f(t)P_i + g(t) + \gamma P_i + \Pi' \mathbb{Z}_{it} + \epsilon_{it}, \qquad [1]$$

where $f(t)$ and $g(t)$ are restricted cubic splines with knots that are evenly spaced in time and P_i is an indicator variable equal to 1 if i is PIPA and 0 if i is the Kiribati control region. Fig. 3, *Bottom* plots the response functions $\widehat{f}(t)$ with 6 knots, estimated separately for the periods before and after PIPA implementation. Models M1 and M2 in *SI Appendix*, Fig. S2 allow 5 and 7 knots, respectively. The vector $\mathbb{Z}_{it}$ is a set of controls. The benchmark model in Fig. 3 includes only a constant term. Model M1 in *SI Appendix*, Fig. S1 adds a quadratic in region-specific sea-surface temperature to the benchmark model. Model M2 in *SI Appendix*, Fig. S1 augments model M1

by adding month fixed effects. ϵ_{it} is an error term. The benchmark model in Fig. 3 allows errors to be robust to serial correlation and heteroscedasticity of arbitrary form within a 60-d window (26). Models M1 and M2 in *SI Appendix*, Fig. S3 allow arbitrary serial correlation across 15- and 90-d time windows, respectively. Model M3 in *SI Appendix*, Fig. S3 allow arbitrary serial correlation with a given month. Model M2 in *SI Appendix*, Fig. S4 uses an alternative neural net classification procedure for determining fishing effort, rather than our benchmark logistic regression classification procedure.

Current fishing pressure estimates for each FAO MFA were calculated using a catch-weighted mean of fishing pressure (F/F_{MSY}) from all fisheries caught in the area from a global database of fisheries status estimates (27). Expected fishing pressure estimates following an announcement of no-take marine reserve expansion to 30% marine reserve coverage are calculated for each fishing area by first determining the difference between 30% and the current MPA coverage, and then increasing a fraction of the current fishing pressure equivalent to this expansion fraction by the empirically determined increase in fishing pressure observed inside PIPA (+130%). We highlight that since many current MPAs are not yet no-take marine reserves, the necessary expansion fraction for each FAO MFA will be even higher than given here. For the Pacific Antarctic area (which, with 39.5% MPA coverage, is the only area with global coverage already exceeding the 30% target), we assumed an expansion fraction of 0%. Similarly, these calculations were also done for all individual fisheries globally to determine the number of fisheries that would move into an overfishing situation ($F/F_{MSY} > 1$) under a marine reserve expansion announcement scenario.

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