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AN UPDATED ECOSYSTEM MODEL OF THE EASTERN TROPICAL PACIFIC OCEAN: ANALYSIS OF ECOLOGICAL INDICATORS AND THE POTENTIAL IMPACTS OF FAD FISHING ON ECOSYSTEM DYNAMICS

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SUMMARY

The Inter-American Tropical Tuna Commission (IATTC) has adopted an ecosystem approach to the management of tuna fisheries in the eastern Pacific Ocean (EPO), as mandated by the Antigua Convention. However, demonstrating ecological sustainability using single-species stock assessments is impractical and cost-prohibitive, given the large number of species with which the EPO tuna fishery interacts, the paucity of catch and biological information for many species of lesser economic or conservation significance, and the fact that these assessment models do not consider the multidimensional predator-prey relationships that can be directly or indirectly impacted by fishing activities. Since 2017, the IATTC has reported time series of catches of non-target species (*e.g.*, sharks, rays, sea turtles, marine mammals, large and forage fishes) and a range of ecological indicators derived from an updated Ecopath with Ecosim (EwE) ecosystem model of the eastern tropical Pacific Ocean (ETP). Together, these data and indicators provide a transparent long-term view of the EPO ecosystem and the potential impacts that may be attributed to the tuna fishery. The ecosystem model can be used in concert with stock assessment models of target species to simulate potential management scenarios that may enable managers to adopt appropriate conservation and management measures that maximize ecological and economic benefits.

This document presents an assessment of the EPO ecosystem using seven ecological indicators that, together, describe changes in the structure and dynamics of the EPO ecosystem during 1970–2017 due to tuna fishing. The results clearly show that the ecosystem structure has changed substantially over the history of the fishery, first due to the increase in industrial fishing from the 1970s, but most markedly since the purse-seine fishery on floating objects—mostly fish-aggregating devices (FADs)—began its dramatic expansion in 1993. The ecosystem model was used to predict the ecological consequences of

continued increases in effort in this fishery, and the potential impacts of limiting its effort as a conservation and management measure, primarily to reduce the fishing mortality of skipjack and small bigeye and yellowfin tunas. The model simulations indicate that, even if the rate of effort increase observed in the fishery over the past 10 years is reduced by 50%, the biomass of some target tuna species may be reduced by up to 62%. The model predicts that limiting the number of floating-object and unassociated sets to the 2016-2018 average would maintain the ecosystem structure in its present state and slightly increase the biomass of most target tuna species, but a significant reduction in purse-seine effort (and most likely longline effort as well) would be needed to restore the EPO ecosystem to its state prior to the expansion of the FAD fishery. Updated trophic information, particularly predator stomach contents data and experimental determination of consumption rates, is needed to improve the ecosystem model and the reliability of forecast outputs.

1. INTRODUCTION

The ecosystem approach to fisheries (EAF)—often used interchangeably with ecosystem-based fisheries management (EBFM)—is a concept that recognizes the broader ecological and environmental impacts that fisheries have upon their supporting ecosystem. The foundations of EAF can be traced back to the 1982 UN Convention on the Law of the Sea, and has since been explicitly or implicitly adopted in a number of legally binding and voluntary instruments developed by coastal States and international organizations (see review by Garcia *et al*., 2003). However, a plethora of ecological, socio-economic and political issues have hampered the implementation of EAF in many parts of the world to the present day, and there are very few examples of EAF operating within tactical fisheries management frameworks.

The Inter-American Tropical Tuna Commission (IATTC) has explicitly defined an ecosystem approach to the management of tuna fisheries. Article VII 1(f) of the 2010 Antigua Convention articulates the IATTC's commitment to the long-term ecological sustainability of the eastern Pacific Ocean (EPO) ecosystem by making it responsible for "*conservation and management measures and recommendations for species belonging to the same ecosystem and that are affected by fishing for, or dependent on or associated with, the fish stocks covered by this Convention…*". Furthermore, the IATTC's Strategic Science Plan (SSP) proposed in 2018, includes an explicit goal (Goal L) to "*evaluate the ecological impacts of tuna fisheries*".

However, demonstrating ecological sustainability can be difficult in practice, due to the general paucity of the types of biological and catch information that would be required for a large number of non-target species to be assessed using traditional stock assessment approaches. Also, such single-species models fail to account for the complex predator-prey relationships that ultimately control the structure and internal dynamics of entire marine ecosystems and that can be easily compromised by fishing activities.

Ecosystem models, however, are a powerful tool designed to disentangle the complex multidimensional trophic relationships between individual species and the environment, allow researchers to better understand the functioning of marine ecosystems, and facilitate the forecasting of impacts by specific perturbations such as fishing and climate change. There are now several examples of ecosystem models being used to demonstrate how industrialized tuna fisheries have been responsible for significant alterations of the structure and dynamics of marine ecosystems (Cox *et al*., 2002; Polovina *et al*., 2009; Griffiths *et al*., 2019). This is mainly a result of tuna fisheries impacting target and non-target species (*e.g.*, tunas, billfishes, sharks) that occupy high trophic levels (TL > 4.0) and can exert strong predatory regulation of species populations at lower trophic levels (Baum and Worm, 2009; Griffiths *et al*., 2013).

The ecological consequences of fishing activities have generally only been described using a few ecological indicators, most commonly the mean trophic level of the catch (TL_c). This has been used to show major changes in the targeting practices of fisheries, usually in response to fishing-induced changes in ecosystem structure which deplete the abundance of large predators. Pauly *et al*. (1998) described this phenomenon as "fishing down the food web", whereby fisheries adapt by targeting increasingly smaller species, resulting in a progressive decline in the TL_c and a change in ecosystem structure to dominance by highly-productive species of often low economic value (Christensen, 1998; Daskalov, 2002; Roux *et al*., 2013). For example, Polovina *et al*. (2009) found that a decline in the catches of apex predators—bigeye and albacore tunas, billfishes, and blue shark—by the Hawaiian tuna longline fishery in the North Pacific subtropical gyre ecosystem resulted in a proliferation of smaller mid-trophic level species (dorado, sickle pomfret, escolar, and snake mackerel) and a reduction in the TL_c from 3.85 to 3.66.

The potential for wild and aquaculture fisheries to alter the integrity of marine ecosystems through direct and indirect impacts on target and associated non-target species has been formally recognised in national and international instruments and fisheries policies in various forms of EAF (Moffitt *et al*., 2016). Nonetheless, the fishing industry has had to be increasingly proactive in developing and implementing fishing practices and policies (codes of conduct, best practices) that help to quell public concerns over the ecological sustainability of specific fishing activities. Such industry initiatives have been used to attain eco-labelling accreditation from a growing number of organizations such as the Marine Stewardship Council (MSC), Friend of the Sea (FoS), and the Global Aquaculture Alliance (GAA). More recently, other organizations, such as Fair Trade USA, have developed congruent certification processes for social aspects of the fisheries supply chain (Bailey *et al*., 2016). Together, these certifications now serve as an important marketing tool in a market where consumers have become increasingly educated on sustainability issues (Gutiérrez *et al*., 2012).

The EPO supports some of the largest and most valuable fisheries in the world (Joseph, 1994). Using primarily purse-seines and longlines, these fisheries target a range of high-trophic-level tunas and billfishes across a region of over 50 million km^2. Catches have increased steadily over the last decade (IATTC, 2018), to the point that there is now concern that two of the three main species—skipjack and bigeye tunas—are experiencing overexploitation (SAC-10-06; SAC-10-09). The major impact on the target species, especially bigeye, in the EPO is a result of the increased effort and efficiency of the purse-seine fishery on floating objects (primarily drifting artificial fish-aggregating devices (FADs)) that aggregate small tunas, together with a range of other, non-target species (Bromhead *et al*., 2003; Dagorn *et al*., 2013). The effort on FADs in the EPO has increased five-fold in the past 25 years, from 2,556 sets in 1993, when the FAD fishery began, to 15,488 sets in 2017 (Figure 1); during 2008–2012 and 2013–2017, the number of FAD sets increased by 48% and 46%, respectively.

FADs can accumulate target species from distances of up to several kilometres (Itano and Holland, 2000; Schaefer and Fuller, 2007), thus allowing the fishery to extract a greater biomass of these species more efficiently than setting on unassociated schools of tunas, which are more widely and heterogeneously distributed across the high seas. The increased FAD effort in the EPO has also increased the catch of numerous FAD-associated non-target species (Hall and Roman, 2013; Lezama-Ochoa *et al*., 2017) (Figure 1), which has raised concerns regarding less productive species, such as the silky shark, that have been identified in ecological risk assessments as among the species most vulnerable to becoming unsustainable in the EPO as a result of tuna fisheries (Griffiths *et al*., 2018; Duffy *et al*., *in review*).

The main objective of this document is to update the Ecopath with Ecosim (EwE) model of the eastern tropical Pacific Ocean (ETP) ecosystem, developed by Olson and Watters (2003), with new time series of catch data, in order to calculate updated values for a range of ecological indicators as a means of assessing the historic and current (2017) status of the ecosystem. Given the increasing use of FADs in the EPO, a secondary aim is to simulate the potential effects of increasing and decreasing fishing effort on FADs over the next 10 years on the biomass of target species and key non-target species, and the structural integrity of the ecosystem.

2. METHODS

2.1. The ETP7 Ecopath ecosystem model of the eastern tropical Pacific Ocean

The ETP7 version of the Ecopath model developed by Olson and Watters (2003) was used to calculate ecological indices and simulate the potential ecological impacts of FADs on the ETP ecosystem. The model area was defined as between 20°N and 20°S from 150°W to the continental shelf break along the coast of the Americas, covering approximately 32.8 million km^2. The data used to parameterize the model and the balancing procedure and to calibrate it to time series data, and the general modelling approach, are described in Olson and Watters (2003).

The reference year for the static description of the trophic flows in the ETP model is 1993, when predator diet data and high-quality observer data for the purse-seine fishery by Class-6 vessels were available. However, it should be noted that the magnitude of trophic flows changes at each time step in Ecosim simulations with changes in the predicted (or input) values of production, consumption and fishing mortality, to maintain mass-balance. Three fisheries (purse-seine, pelagic longline, and pole-and-line) were included in the model; to facilitate changes in purse-seine effort in a modelling environment, the purse-seine fishery was divided into three separate fisheries, defined by their predominant set type: on natural or artificial floating objects (OBJ), on unassociated tuna schools (NOA), and on dolphins (DEL).

2.2. Ecological indicators

In exploited pelagic ecosystems, fisheries that target large piscivorous fishes act as apex predators. Over time, fishing can cause the overall size composition of the catch to decrease and, in general, the TLs of smaller organisms are lower than those of larger organisms. There are various ecological indicators that can be used to describe the status of the ecosystem, usually in reference to a particular benchmark year. In the EPO, the FAD fishery began to develop significantly in 1993, which, coincidentally, was the year used to characterize the Ecopath model of the ETP by Olson and Watters (2003).

Fulton *et al.* (2005) undertook a simulation study using two ecosystem models, and found that indicators at the community or ecosystem level were the most reliable indicators of ecosystem change under fishing. Despite the widespread use of TL_c as the primary indicator of ecosystem change (Pauly *et al*., 1998; Sibert *et al*., 2006; Polovina *et al*., 2009), Fulton *et al.* (2005) found that no single indicator was able to provide a complete representation of the ecosystem, and suggested that simultaneous use of a variety of indicators is required to detect the full range of impacts from fishing. In general, these indicators are either catch-based or community-based, and together can provide a complete picture of the current and historic internal dynamics and integrity of an ecosystem (Shannon *et al*., 2014).

In the analysis of the EPO, seven indicators (three catch-based (TL_c, Marine trophic index, Fishing in Balance index), and four community-based (mean trophic level of the community for trophic levels 2.0–3.25, ≥3.25–4.0, and >4.0)) were used, based on the recommendations of Shannon *et al*. (2014). A description of each indicator is provided below.

2.2.1. Mean trophic level of the catch (*TL_c*)

The mean trophic level of the catch (TL_c) by fisheries can be a useful metric of ecosystem change because it integrates an array of biological information about the components of the system. TL_c is also an indicator of whether fisheries are changing their fishing or targeting practices in response to changes in the abundance or catchability of traditional target species. For example, declines in the abundance of large predatory fish caused by overfishing can result in fisheries targeting species at progressively lower trophic levels. Studies of this “fishing down the food web” phenomenon (Pauly *et al*., 1998), have shown that the TL_c decreased by around 0.1 of a trophic level per decade. TL_c is calculated for a specific year as:

$$TL_C = \sum_{i=1}^{n} \frac{(Y_i \cdot TL_i)}{Y_C}$$

where Y_i is the catch of functional group *i*, TL_i is the trophic level of functional group *i*, and Y_c is the total catch of all functional groups.

2.2.2. Mean trophic index (MTI)

The mean trophic index (MTI) is essentially the same as TL_c, but it includes only high-trophic-level species (TL>4.0), which are usually the first indicator of 'fishing down the food web'. In ecosystems where high-trophic-level predators are exploited (*e.g.* industrial tuna fisheries), the MTL will often be identical to the TL_c. Some ecosystems, however, have changed in the other direction, from lower to higher TL communities, sometimes as a result of improved technologies to allow exploitation of larger species—referred to as "fishing up the food web" (Branch *et al*., 2010)—but this can also be an artefact of improved catch reporting, if previously unreported catches of discarded predatory species, such as sharks, are recorded (Stergiou and Tsikliras, 2011).

The MTI is calculated for a specific year as:

$$MTI = \sum_{i=1}^{n} \frac{\left(Y_{TL_i \geq 4.0} \cdot TL_i\right)}{Y_{C(TL_i \geq 4.0)}}$$

where $Y_{TL_i \geq 4.0}$ is the catch of functional group *i* with a TL≥4.0, TL_i is the trophic level of functional group *i*, and $Y_{C(TL_i \geq 4.0)}$ is the total catch of all functional groups with a TL≥4.0.

2.2.3. Fishing in Balance (FIB) index

The FIB index (Pauly *et al*., 2000) provides an indication of whether fisheries are balanced in ecological terms and not disrupting the functionality of the ecosystem. FIB incorporates the MTI, and is defined for year *k* as:

$$FIB = \log\left[Y_k \left(\frac{1}{TE}\right)^{MTI_k}\right] - log\left[Y_0 \left(\frac{1}{TE}\right)^{MTI_0}\right]$$

where Y_0 and MTI_0 are the total catch and MTI, respectively, in the reference year, TE is the transfer efficiency (*i.e.* the ratio between the sum of exports from a specified trophic level plus the flow that is transferred to the next trophic level, and the throughput of the specified trophic level), while 0 is the year used as a baseline, in this case 1993. FIB can provide an indication of overfishing when catches do not increase as expected (or TL_c decrease) given the available productivity in the system, or if the effects of fishing are sufficient to compromise the functionality of the ecosystem (FIB<0). In contrast, FIB can indicate an expansion of the fishery (*e.g.* increase in diversity and/or biomass of bycatch) (FIB>0).

2.2.4. Mean trophic level of the modelled community (TL_{MC})

The mean trophic level of the modelled community (TL_{MC}) was described by Shannon *et al*. (2014), who used Ecopath to estimate the mean trophic level of specific components of an ecosystem, thus allowing changes in the ecosystem structure after biomass removals by fishing to be examined. The three TL categories for the EPO were 2.0–3.25, ≥3.25–4.0, and >4.0. Each indicator is calculated within the Ecopath software as (using TL 3.25-4.0 as an example):

$$TL_{MC3.25} = \sum_{i=1}^{n} \frac{\left(B_{M_i 3.25} \cdot TL_i\right)}{B_{MT(M_i 3.25)}}$$

where $B_{M_i 3.25}$ is the modelled biomass of each functional group *i* with a TL≥3.25, TL_i is the trophic level of functional group *i*, and $B_{MT(TL_i 3.25)}$ is the total modelled biomass of all functional groups with a TL≥3.25. These indicators can be used in unison to detect trophic cascades, in which a decline in biomass of $TL_{MC4.0}$ due to fishing increases the biomass of $TL_{MC32.5}$ as predation pressure is reduced, which in turn increases predation pressure on $TL_{MC2.0}$, thus reducing its biomass.

2.2.5. Shannon's index

Shannon's index (*H*) (Shannon, 1948) is widely used in ecology as a measure of species diversity, that is, species richness and the relative proportions of species in a community (or 'evenness'), generally measured in terms of biomass or number of individuals. Since the number of functional groups in an Ecopath model is fixed, the index essentially measures evenness, or the relative difference in the biomass of functional groups.

Shannon's index is defined as:

$$H = -\sum_{i=1}^{k} p_i \, ln(p_i)$$

where k is the total number of functional groups in the model, and p_i is the proportion of the total ecosystem biomass represented by functional group *i*.

Given the potential utility of combining ecological indicators to describe the structure and internal dynamics of the EPO ecosystem, annual indicator values were estimated from a time series of annual landing and discard data, by species, for the three purse-seine fishing modes, the pole-and-line fishery, and the longline fishery in the EPO during 1970-2017. The annual catch of each species in the IATTC tuna, bycatch, and discard databases was assigned to a relevant functional group defined in the ETP ecosystem model, and the Ecosim model was refitted to the time series of catches to estimate the seven ecological indicators (TL_c, MTI, FIB, Shannon's index, $TL_{MC\ 2.0}$, $TL_{MC\ 3.25}$, $TL_{MC\ 4.0}$).

2.3. Calculating ecological indices and simulating the ecological impacts of FADs

The previous version of the ETP7 model was built in Ecopath 5.1, and has time series of catch, effort, and in some cases total mortality (dolphins) and fishing mortality (yellowfin, bigeye and skipjack tunas) up to 1999. Since the Olson and Watters (2003) report was published, both the functionality of the EwE software (currently version 6.5) and the IATTC observer data have improved substantially. Therefore, the model was converted to Ecopath 6.5, and the annual catch of each species from the IATTC tuna, bycatch, and discard databases was assigned to the relevant functional group defined in the model. The Ecosim model was then refitted to the time series of catch and effort for 1970–2017. Annual ecological indicator values for the ETP were estimated in Ecosim.

Simulations of changing FAD fishing effort were undertaken in Ecosim to explore the potential ecological consequences of varying FAD fishing effort over the next 10 years. Specifically, the following four management scenarios were simulated:

- **Scenario 1**: continue the increasing effort trend in the OBJ fishery over the past 10 years by increasing the number of sets linearly by 100% during 2017-2026;
- **Scenario 2**: reduce the rate of effort increase in the OBJ fishery by increasing the number of sets by 50% linearly during 2017-2026;
- **Scenario 3**: constrain purse-seine fishing effort, by set type (OBJ, NOA and DEL), to the average number of sets during 2016–2018, and hold it at that level, and longline effort at the 2017 level, during 2017-2026.

- **Scenario 4**: implement an annual limit of 15,831 sets for the OBJ and NOA fisheries combined, and during 2017-2027 (a) increase the proportion of OBJ sets by 1% per year; (b) decrease the proportion of NOA sets each year correspondingly to avoid exceeding the set limit; (c) maintain the number of DEL sets at the 2016–2018 average; (d) maintain longline effort at the 2017 level.

3. RESULTS AND DISCUSSION

3.1. Status of the EPO ecosystem using ecological indicators

TL_c and MTI values increased steadily during 1970-1991, from 4.65 and 4.67 to 4.69 and 4.70, respectively (Figure 2). Between 1991 and 1997, as the purse-seine effort on FADs increased rapidly, TL_c decreased gradually to 4.65, due to the increasing catches of tunas and bycatches of other high-trophic-level species that also aggregate around floating objects, such as sharks, billfishes, wahoo and dorado. This expansion is seen in the FIB index, which is mostly negative prior to 1993 and almost entirely positive after that, and in Shannon's index, which decreased steadily after 1993 (Figure 2). By the early 2000s, TL_c, MTI, and Shannon's index all show a gradual decline, while the FIB index increased to a peak of 0.66 in 2017 (Figure 2). In 2017, both TL_c and MTI reached their lowest historic levels of 4.64 and 4.65, respectively (Figure 2). Since its peak in 1991, TL_c has declined by 0.05 of a trophic level, or 0.02 trophic levels per decade.

These indicators generally describe changes in the exploited components of the ecosystem, whereas community biomass (TL_{MC}) indicators describe changes in the structure of the ecosystem once biomass has been removed by fishing. Figure 2 shows the biomass of the $TL_{MC\mathit{4.0}}$ community was at one of its highest values (4.449) in 1993, but had declined to 4.443 by 2017. As a result of changes in predation pressure on lower trophic levels, during 1993-2017 the biomass of the $TL_{MC\mathit{3.25}}$ community increased from 3.800 to 3.803 while, interestingly, the biomass of the $TL_{MC\mathit{2.0}}$ community also increased, from 3.306 to 3.308.

Together, these indicators show that the ecosystem structure has changed over the 48-year analysis period. However, these changes, even if a direct result of fishing, are not considered ecologically detrimental, but the patterns of change, particularly in the mean trophic level of the communities since 2010, warrant continuing, and possibly expanding, monitoring of the fisheries in the EPO.

3.2. Simulating the potential ecological impacts of changes in fishing effort on FADs

3.2.1. Biomass changes for key species

Scenarios 1 and 2 resulted in a significant reduction in the biomass of all key target (yellowfin, bigeye and skipjack tunas), byproduct (large size classes of marlins, dorado and wahoo), and bycatch species of conservation significance (sharks and rays) (Figure 3). The only exceptions were small marlins, small wahoo, and sea turtles. Given the affinity of small tunas for FADs, it is not surprising they showed among the largest decreases in biomass, especially small bigeye tuna, which decreased by 62% and 80% under 50% and 100% increases in the number of OBJ sets, respectively. Similar or greater decreases in biomass were predicted for small and large sharks (Figure 3), mainly juvenile and adult silky sharks.

Results for Scenarios 3 and 4 were remarkably similar, with predicted increases in the biomass of target species, the largest for small yellowfin (18% and 17%, respectively), skipjack (12% and 7%) and small bigeye (8% and 2%). An exception was large bigeye tuna, which decreased by 27% and 32% under Scenarios 3 and 4, respectively. Increases in the biomass of byproduct species (small marlins and large wahoo) and bycatch species (sea turtles and rays) were also predicted under these two scenarios.

A particularly important distinction between Scenarios 1 and 2 and Scenarios 3 and 4 is that the biomass reductions for large bigeye tuna, large marlins, and small and large sharks predicted by the latter two were almost half of those predicted by Scenarios 1 and 2. Given that the purse-seine fishery primarily catches small bigeye, this result may be due to the simulation continuing longline fishing effort at the 2017 level of 284,353,250 hooks (IATTC Public Domain Data), the second highest longline effort ever recorded in the EPO tuna fishery. This may also explain the biomass declines predicted in Scenarios 1 and 2 for small and large sharks and large marlins, which are also primarily caught in the longline fishery (SAC-10-03).

3.2.2. Changes in ecosystem structure

Figure 4 shows the predicted values of seven ecological indicators under the four simulated scenarios. Scenarios 1 and 2, involving substantial increases in OBJ effort, resulted in a predicted decline of up to 3% in the TL_c and MTI, a small increase in the FIB index, and declines in the biomass of the $TL_{\geqslant 4.0}$ community and Shannon's index. Together, these indicators show that such significant increases in the number of floating-object sets over the next 10 years are likely to continue to change the structure and dynamics of the ETP ecosystem.

Scenario 3 and 4 produced very similar results, predicting a slight decline in the TL_c and MTI, but the other indicators at their 2017 level. These results indicate that a substantial reduction in purse-seine, and also most likely longline, effort is required to restore the ecosystem structure back to even 2010 levels, when the number of floating-object sets and longline effort were about half (6,399 sets/150,681,785 hooks) those of 2017 (11,147 sets/284,353,250 hooks). Restoring the ecosystem structure to pre-1993 conditions, before the expansion of the FAD fishery, is a much more challenging task, and would mean large reductions in fishing effort in all fisheries.

3.3. Considerations for future work

There are a number of assumptions to consider in interpreting these simulation results. Since the focus was the potential ecological impact of the almost 50% increase in the number of sets on FADs every 5 years, no change in effort in any other EPO tuna fishery during the 10-year simulation period was assumed. This is unlikely to be the case, particularly for the longline fishery, whose effort in the EPO approximately doubled during 2008-2015 (Griffiths and Duffy, 2017). Since the longline fishery primarily targets large bigeye tuna and swordfish, a continued increase in OBJ effort, as observed over the past 10 years, may begin to present a significant interaction between these fisheries, since the purse-seine fishery targets small bigeye tuna. The model predicts that even a 50% increase in the number of OBJ sets over the next 10 years will have a significant direct impact on small bigeye tuna, resulting in an 84% reduction in the adult biomass.

An important consideration when interpreting the Ecopath results is that the model's underlying diet matrix—the component of the model that defines the trophic linkages among species in the ecosystem—is based on stomach content data from the early to mid-1990s, when the FAD fishery was just starting, and as the results of these analyses and of Olson *et al*. (2014) show, it is likely that the trophodynamics of the ecosystem have changed significantly, bringing into question the realism of the model predictions. Furthermore, these diet data were supplemented with data from other regions of the Pacific Ocean and beyond, where no local data were available for a particular species or functional group. Given the significant environmental changes observed in the EPO over the past decade, with some of the strongest El Niño events on record (Kintisch, 2015; Cai *et al*., 2018), there is a critical need to collect trophic information for not only species of economic (*e.g.* tunas) or conservation (*e.g.* sharks) importance, but also their prey, and the base of the food web (phytoplankton), which can have a significant impact in oligotrophic oceanic ecosystems that are thought to be controlled by 'bottom-up' processes (Hunt and McKinnell, 2006).

Predator stomach samples were taken by observers from the AIDCP On-Board Observer Program (*e.g.*, Olson and Galvan-Magaña, 2002; Olson *et al*., 2014), albeit over two decades ago. Future research arising from this work should include collection of trophic samples by on-board observers, and through collaboration with CPCs and other stakeholders (*e.g.*, universities, research institutes) interested in an up-to-date and reliable ecosystem model that can be used to explore the potential ecological consequences of future human actions and/or environmental conditions.

Similarly, many of the biological parameters in the ETP Ecopath model have been derived from other ocean basins or related species, thus introducing some uncertainty about the magnitude of estimated trophic flows in the ecosystem. The consumption/biomass ratio (Q/B) is one of the most influential parameters in Ecopath models, as it describes the energy requirements of predators and the required standing biomass of their prey. However, it is also a very difficult parameter to measure in pelagic fishes, since it requires experiments in highly specialized facilities to determine their consumption requirements. As a result, there are very few experimentally derived estimates of consumption and daily ration for tunas (Magnuson, 1969; Olson and Boggs, 1986; Olson and Mullen, 1986).

The IATTC's Achotines Laboratory in Panama is one of the very few facilities in the world where such experiments are possible. Several species of large pelagic fishes are available in nearby waters—with the continental shelf break only 12 km from the coast—and can be easily transferred to a number of large holding tanks to conduct experiments. The results of such experiments would help fill the gaps in knowledge in the trophic ecology of pelagic fishes in the EPO, and lead to reliable parameter estimates for future ecosystem models for the EPO. The IATTC staff has undertaken a comprehensive review of methods to experimentally determine Q/B that can be applied to pelagic predators that occupy different trophic levels in the EPO ecosystem and are abundant near the Achotines Laboratory (SAC-10 INF-E).

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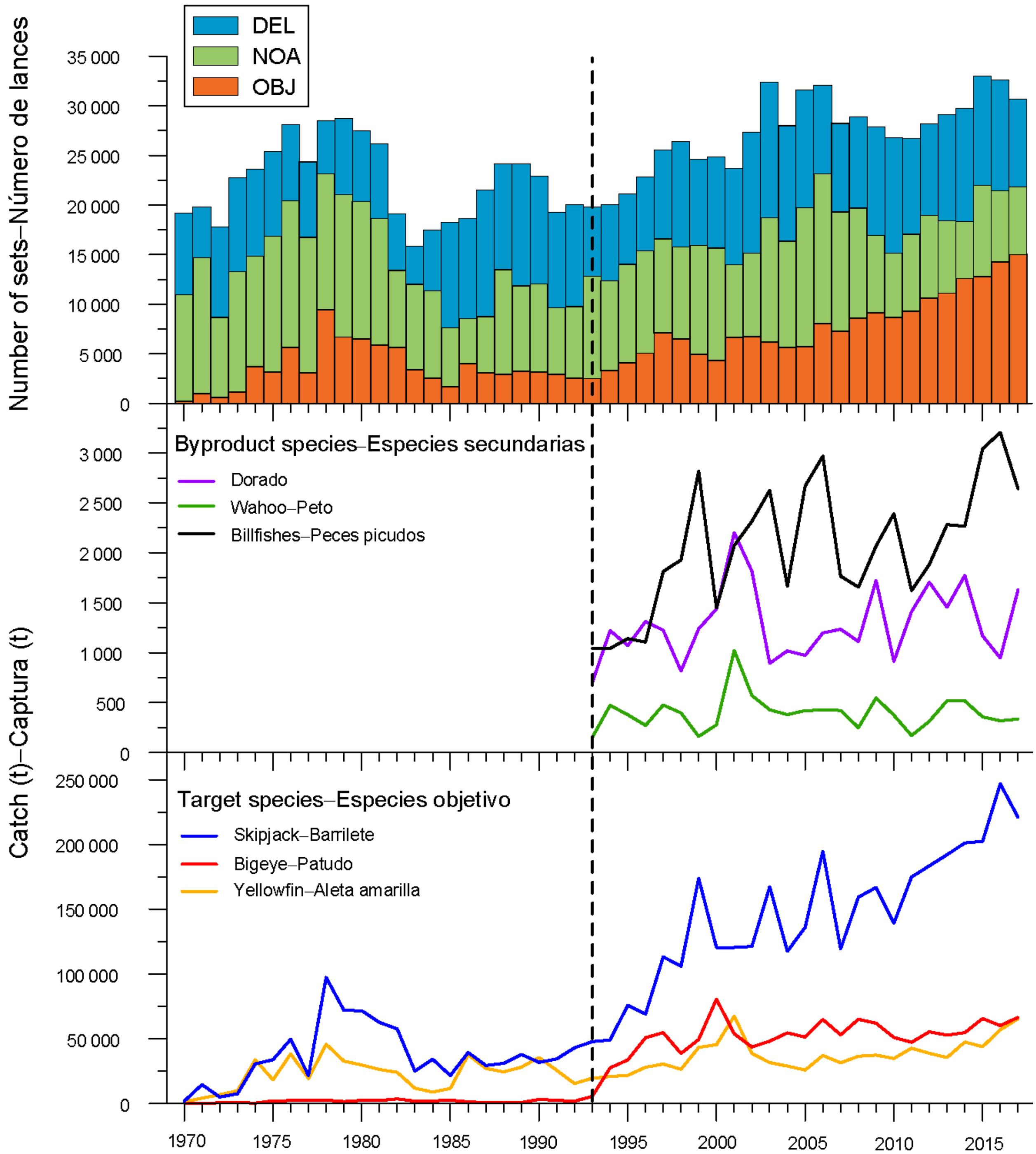

FIGURE 1. Top panel: number of sets in the purse-seine fishery in the EPO, 1970-2017, by type (dolphin (DEL), unassociated (NOA), floating object (OBJ)); Middle panel: (2) reported catches of predominant byproduct species (dorado, wahoo, billfishes) in OBJ sets, 1993-2017; Bottom panel: reported catches of tropical tunas, by species (skipjack, bigeye, yellowfin), 1970-2017. The dashed vertical line indicates 1993, when both the expansion of the FAD fishery and reporting of non-target species began.

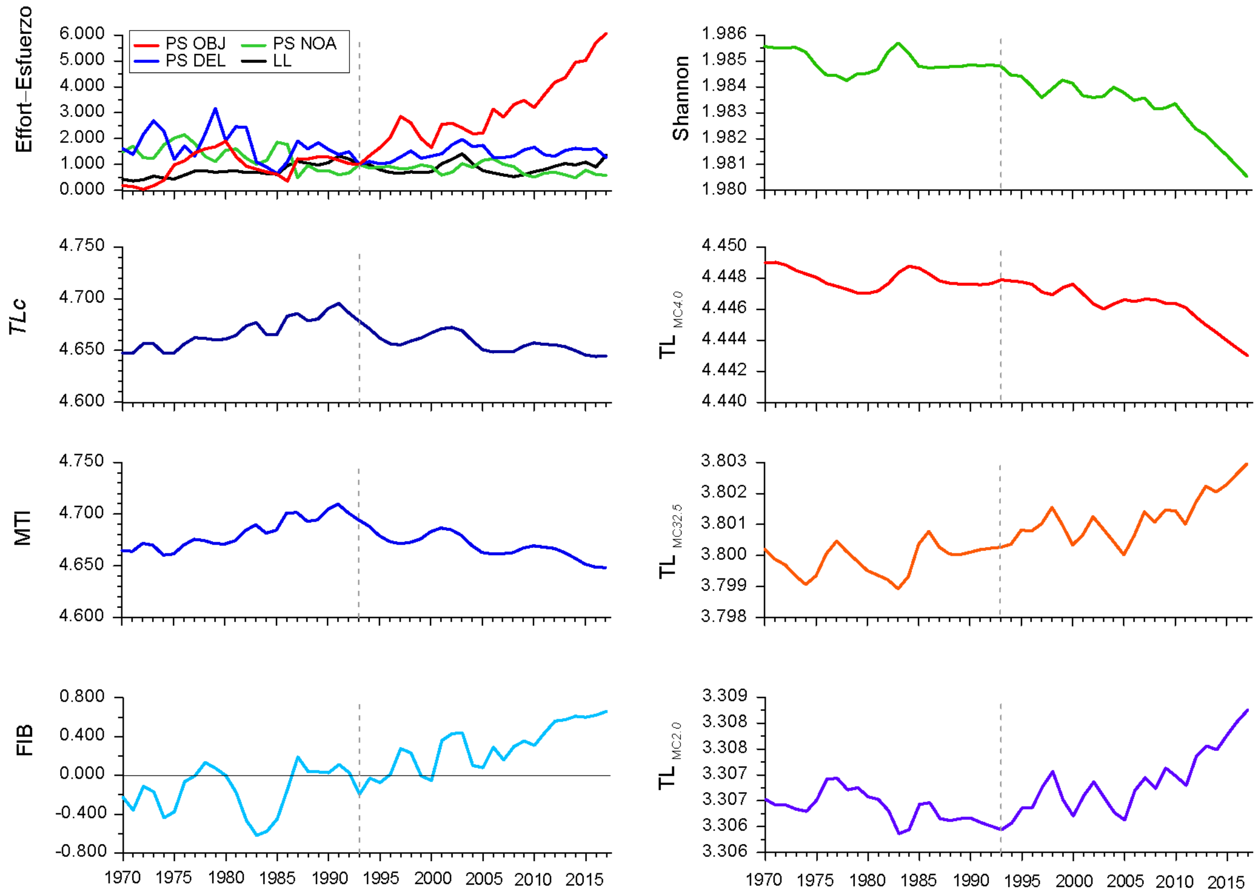

FIGURE 2. Index of fishing effort (effort relative to 1993 effort), by purse-seine set type (dolphin (DEL), unassociated (NOA), floating object (OBJ)) and longline, 1970-2017 (top left); and annual values for seven ecological indicators that describe the changes in different components of the tropical EPO ecosystem, 1970–2017, derived from a trophic mass-balance model of the pelagic eastern tropical Pacific Ocean ecosystem . See text for descriptions of indicators. The dashed vertical line indicates 1993, when the expansion of the FAD fishery began.

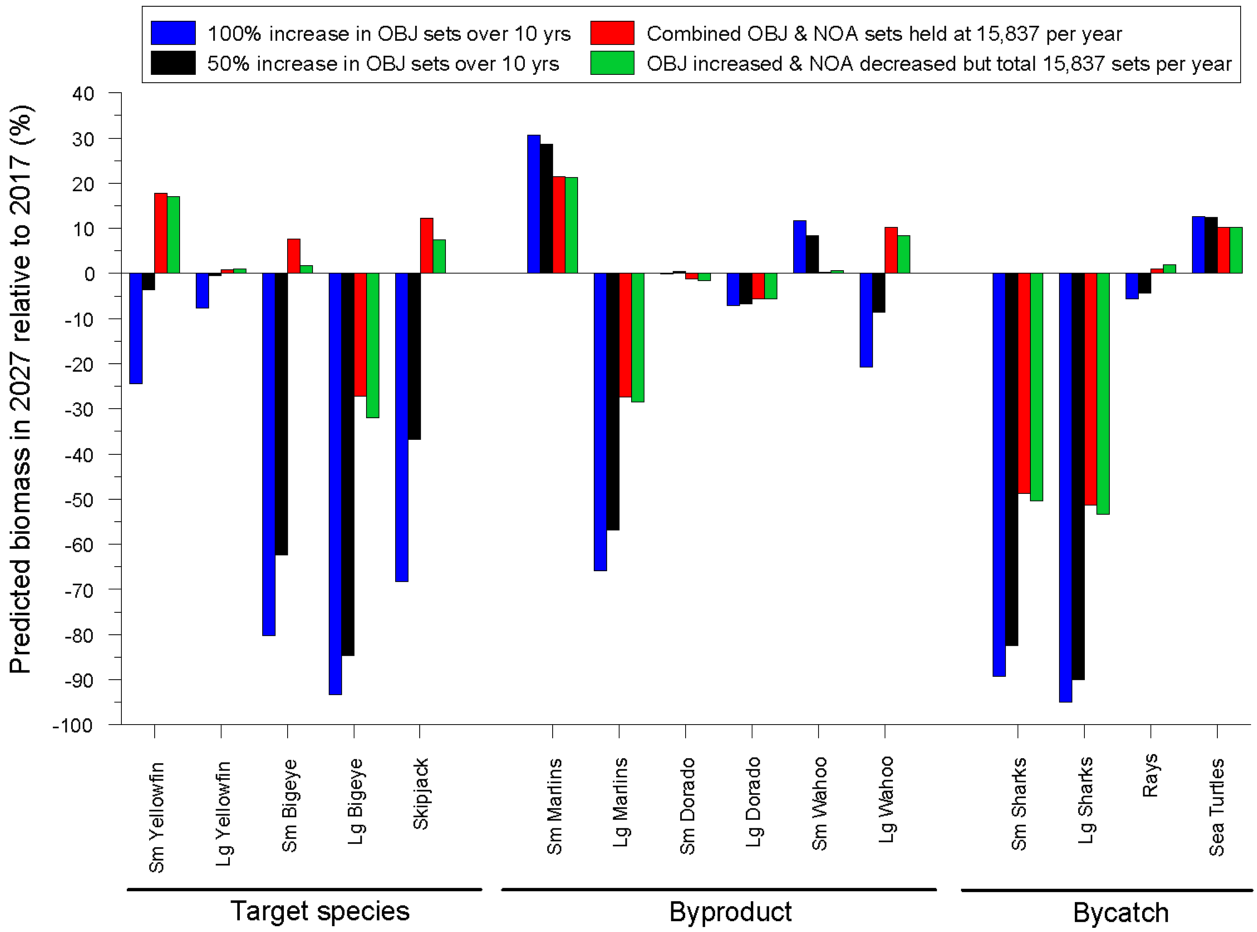

FIGURE 3. Relative changes predicted by Ecosim in the biomass of key functional groups representing primary target species, byproducts, and bycatches of the purse-seine fishery in the eastern tropical Pacific Ocean, 2017-2027, under four hypothetical management scenarios. See section 2.3 of text for details of the scenarios.

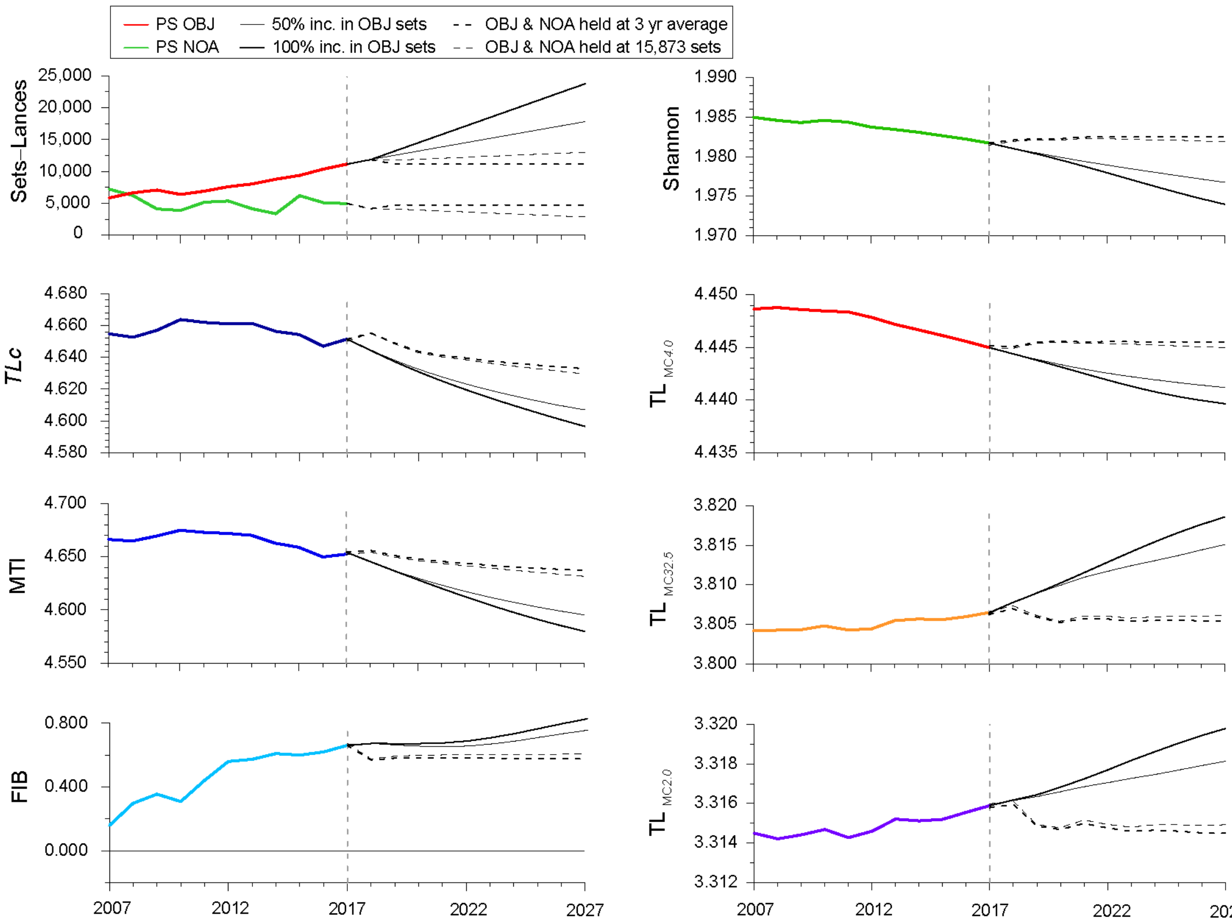

FIGURE 4. Projected changes in annual values for seven ecological indicators in four scenarios with different levels of effort by the purse-seine fishery on floating objects (OBJ) during 2017-2027. See section 2.3 of text for details of the scenarios. The dashed vertical line indicates 2017, the starting year for the simulations.