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USING AN UPDATED ECOSYSTEM MODEL OF THE EASTERN TROPICAL PACIFIC OCEAN TO EXPLORE POTENTIAL IMPACTS OF INCREASED FISHING EFFORT ON FLOATING OBJECTS

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SUMMARY

The Inter-American Tropical Tuna Commission (IATTC) has annually reported on 7 ecological indicators derived from the “ETP7” ecosystem model since 2019 as one of several strategies to facilitate an ecosystem approach to the management of tuna fisheries in the eastern Pacific Ocean (EPO) as mandated by the Antigua Convention. Reporting of bycatch interactions and ecological indicators provide a transparent long-term view of the EPO ecosystem and the potential impacts that may be attributed to the tuna fishery. New bycatch data have been added to the model annually from 2019. However, since 2003 when the ETP7 model was developed, it has not been revised to include new longline bycatch estimates or re-calibrated to time series data that have since become available from stock assessments. The model was restructured to contain multi-stanza delay-difference models for small and large sizes of 10 taxa, and biological parameters of functional groups were updated where possible and the model rebalanced to ensure the model was thermodynamically stable using the new “PREBAL” diagnostic tool in Ecopath software. The Ecosim model was then fit to time series of relative biomass or abundance, fishing mortality,

and catch (retained and discards) for 10 groups and catch only for another 16 groups.

Ecological indicator values from the updated ETP7 model, herein called “ETP-21”, complemented results of previous assessments that suggested that the EPO ecosystem structure has changed substantially over the history of the fishery. However, as a result of incorporating new data from the longline fishery, the fishing impacts on the ecosystem appear more pessimistic than in the 2019 assessment with a significant decline in the mean trophic level of the catch from 4.77 in 1991 to 4.65 in 2018, which coincided with an increase in the number of OBJ sets. Under fishing effort scenarios reflecting the possible tuna conservation measures to be put in place on termination of Resolution C-20-06 in 2021, the model predicted declines in the biomass of bigeye, yellowfin and skipjack tunas by 0.67–3% over the simulation period 2018–2024. Small and large sharks were impacted more heavily, declining in biomass by 13.8% and 10.4%, respectively. This decreased the predation mortality on predominant FAD-associated bycatch species (dorado, wahoo, and marlins), which resulted in increases in their biomasses by up to 3.3%. Perpetual increases in purse-seine fishing effort on FADs, coupled with the impacts of the industrial longline and coastal fisheries and a changing climate, is likely to continue to alter the structure and dynamics of the ETP ecosystem. The need for updated trophic information, particularly predator stomach contents data and experimental determination of consumption rates, is discussed to improve the ecosystem model and the reliability of forecast outputs.

1. INTRODUCTION

The Inter-American Tropical Tuna Commission (IATTC) is one of the few tuna Regional Fisheries Management Organizations (tRFMO) that has pursued an ecosystem approach to the management of their tuna fisheries to explicitly recognize the potential for fishing activities to have ecological and environmental impacts that extend beyond that of the target species. Article VII 1(f) of the Antigua Convention—entering into force in 2010—articulates the IATTC’s commitment to the long-term ecological sustainability of the eastern Pacific Ocean (EPO) ecosystem by adopting “*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 Strategic Science Plan (SSP), adopted by the Commission in 2018, states 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 common 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. Nonetheless, 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 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 the significant alteration 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.,* tuna, billfish, sharks) that often 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, the most common of which being the mean trophic level of the catch (TL_c). The TL_c has been used to show major changes in the targeting practices of fisheries, usually in response to fishing-induced changes in ecosystem structure whereby the abundances of large predators have been depleted. Pauly *et al.* (1998) described this phenomenon as "fishing down the food web" whereby fisheries adapt by increasingly targeting smaller species, thus resulting in a progressive decline in the TL_c and a change in ecosystem structure being dominated by highly productive species that often have low economic value (Christensen 1998; Daskalov 2002; Roux *et al.* 2013). For example, Polovina *et al.* (2009) found a decline in the catches of apex predators—bigeye and albacore tunas, billfish and blue shark—by the Hawaiian tuna longline fishery in the North Pacific subtropical gyre ecosystem resulted in the proliferation in the abundance 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 recognized in national and international instruments (*e.g.* The Agreement on the International Dolphin Conservation Program (AIDCP)) and fisheries policies in various forms of EAF (Moffitt *et al.* 2016). Notwithstanding, the fishing industry itself has had to be increasingly proactive in developing and implementing fishing practices and policies (*e.g.* codes of conduct, best practices) that help to address public concerns over the ecological sustainability of specific fishing activities. Such initiatives by the industry have been used to attain eco-labelling accreditation by 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 marketplace where consumers have become increasingly educated on sustainability issues (Gutiérrez *et al.* 2012)—such as dolphin-caught yellowfin tuna and the use of artificial fish aggregating devices (FADs) in tuna fisheries—and give accredited products the required credence to pass socio-political scrutiny by the general public as being socially acceptable for consumption.

The EPO supports among the largest and most valuable fisheries in the world (Joseph 1994). Using primarily purse-seine and longline, these fisheries target a range of high-trophic-level tunas and billfish across a region of over 50 million km^2. The catches, composed mainly of skipjack, yellowfin, and bigeye tunas, have increased steadily over the last decade (IATTC, 2018), to the point that the bigeye stock can be considered fully-exploited (Xu *et al.* 2018). The major impact on the target species, especially bigeye tuna, in the EPO is a result of the increased effort and efficiency of the purse-seine fishery associated with setting on floating objects (primarily drifting artificial FADs) that aggregate small size classes of these tunas—and a range of other non-target species (Bromhead *et al.* 2003; Dagorn *et al.* 2013). The IATTC has implemented a range of conservation and management measures to reduce the fishing mortality on small tunas, including a 72 day EPO-wide closure for purse-seine vessels, a 30-day closure of the "corralito" where small tunas are abundant, and limits on fleet capacity. Together, these measures are not sufficient to prevent further increases in fishing mortality on bigeye tuna, mainly due to the continuing increase in numbers of floating-object sets, and the IATTC staff is recommending additional measures to prevent the *status quo* conditions from being breached (SAC-12-08; SAC-12-16). To place the issue in context, the effort on FADs in the EPO has increased five-fold in the past 25 years, from 2,556 sets in 1993—generally accepted as the beginning of the FAD fishery in the EPO—to 15,488 sets in 2017 (SAC-09-03) (Figure 1). In the preceding two five-year periods (2008–2012 and 2013–2017) FAD sets increased by 48% and 46%, respectively.

FADs therefore, have the potential to accumulate the biomass of target species from distances of up to several kilometers (Itano and Holland 2000; Schaefer and Fuller 2007), that allow the fishery to more efficiently extract a greater biomass of these species than setting on free-swimming 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 by scientists regarding less productive species, such as elasmobranchs, which have been identified in ecological risk assessments to be among the most vulnerable taxa to becoming unsustainable in the EPO as a result of tuna fisheries (Griffiths *et al.* 2018; Duffy *et al.* 2019).

The objective of the present paper was to update the Ecopath with Ecosim model of the eastern tropical Pacific Ocean (ETP) ecosystem developed by Olson and Watters (2003) with new time series of catch data to calculate updated values for a range of ecological indicators as a means of assessing the historic and recent (2018) status of the ecosystem. Given the increasing use of FADs in the EPO, a secondary aim of the paper was to simulate the potential consequences of increasing and decreasing fishing effort on FADs over the next 10 years on the biomass of target tuna species, bycatch species, and the structural integrity of the ecosystem.

2. METHODS

2.1. Updating procedure for the ETP7 Ecopath ecosystem model

Ecological analyses were conducted using the ETP7 EwE model Olson and Watters (2003) that was updated from EwE software version 5.1 to 6.5 in 2019 by Griffiths and Fuller (2019) and further updated to version 6.6 in the present study. The model area covers 20°N–20°S and 150°W to the continental shelf break along the coast of the Americas, covering approximately 32.8 million km^2. The data initially used to parameterize the model, the balancing procedure, and calibration to time series data are described in Olson and Watters (2003).

Three fisheries were included in the model: purse-seine, pelagic longline, and pole-and-line. However, to properly characterize the purse-seine effort in a modelling environment, the fishery was divided into three separate fisheries defined by their predominant set type: sets in association with natural or artificial floating objects (OBJ), sets on free-swimming tuna schools not associated with floating objects (NOA), and sets made on dolphins (DEL). Annual catch of each species from the IATTC tuna, bycatch, and discard databases were assigned to a relevant functional group defined in the model.

2.1.1 Characterization of the eastern tropical Pacific Ocean ecosystem in Ecopath

The year 1993 was the initial Ecopath model reference period chosen by Olson and Watters (2003) to characterize the static description of the trophic flows in the ETP model, since predator dietary data and high quality observer data for Class 6 purse-seine vessels were available. The reference year remained unchanged in the present assessment since insufficient additional dietary information has been collected across multiple trophic levels in the intervening years (but see Olson *et al.* 2014; Duffy *et al.* 2015; 2017; 2021) to warrant updating the model's diet matrix.

During 2019–2020, IATTC staff improved the catch estimates for bycatch species reported to the IATTC by CPCs as Task I data for a range of smaller coastal fisheries (*e.g.* longline, gillnet), which had previously been aggregated into a general category called "other fisheries". Upon disaggregation it became apparent that significant catches of species (*e.g.* silky shark) were taken in coastal longline fisheries (also see IATTC, 2014). Therefore, data for "other fisheries" was disaggregated into its constituent fisheries and added to the Ecopath model, which resulted in the model requiring rebalancing since the additional catches increased the fishing mortality for several functional groups. This required reparameterization of the production to biomass ratio (P/B) ratio, which is equivalent to total mortality (Z), as the model interprets the additional fishing mortality as a loss in net biomass available for predators and prey. As a consequence, the consumption to biomass ratio (Q/B) also requires adjustment in order for the P/B ratio to fall within biologically plausible range, generally 0.05–0.3 for most living, non-planktonic groups (Christensen et al. 2009).

The model was also improved by using available stock assessment data and life history parameters for commercially-important species to define multi-stanza delay-difference sub-models within Ecopath. These sub-models link energy flows between the different ontogenetic stanzas believed to have very different ecological and biological characteristics, in particular diet composition. In the original ETP7 model there were several species represented as separate ontogenetic functional groups (*e.g.* small and large yellowfin tuna) but they were not linked, resulting in these groups acting as independent biomass pools. In other words, the biomass of an older life stanza for a particular species was not directly influenced by changes in the biomass of the young life stage. Each multi-stanza delay-difference model required input parameters of biomass, Z (≈P/B) and consumption (Q/B) for a "leading" stanza, and the model then calculates values for these parameters for the remaining stanza.

2.1.2 Rebalancing the ETP7 model

Once all model parameters were estimated from the available datasets and literature for 1993, multi-stanza models were constructed for yellowfin tuna, bigeye tuna, swordfish, sailfish, dorado, and wahoo. Balancing then began by first focusing on the functional groups for which the most reliable information was available, such as from stock assessments or quantitative population surveys. The model was attempted to be balanced by first focusing on preliminary ecotrophic efficiency (EE) values, which is the proportion of a group's biomass that is utilized within the system, and production to consumption ratios (P/Q) estimated by Ecopath for these key species. For most living groups, the primary aim during balancing is to allow Ecopath to estimate an EE of around 0.95 for species that are either heavily fished or are expected to experience high predation rates (Christensen *et al.* 2009). In contrast, highly abundant and/or highly productive species (*e.g.* microzooplankton), or high trophic level predators that have very few natural predators in the absence of fishing (*e.g*. large sharks) are expected to have low EE values of 0.1–0.3. This indicates that only a small percentage of the species' biomass is utilized within the system, while the remainder is transferred to detritus or exported from the system.

In cases where the EE value was unrealistic—for example large yellowfin tuna initially had an EE of over 1,000,000—the biomass estimate was checked, followed by an inspection of the predation mortality and fishing mortality calculated by Ecopath. Commonly, the most scientifically defensible option to reduce the EE of a group—when all other parameters appear plausible—is to slightly reduce the proportion it contributes to the diets of predators that have a very high biomass or are highly productive. The converse applies where the EE is too low, requiring greater predation mortality when all other parameter values are considered valid. These are common approaches used by Ecopath practitioners since the diet composition of most marine predators is often highly variable in space and time and an Ecopath model attempts to capture this variability for a single year. In many respects the initial diet composition of a group should be viewed as a reasonable starting point with an expectation of fine-tuning using knowledge of the species and the system (see Christensen et al. 2009).

After the EE values were considered reasonable, the P/Q estimate for each group was checked to ensure it was biologically realistic. In most cases, P/Q should be 0.1–0.3. In cases where P/Q was unrealistic, such as for large bigeye tuna initially having a P/Q of 0.004, the P/B and Q/B values were inspected and revised appropriately. It was often the case that daily ration estimates from the literature were underestimated or unreliable, while the P/B value was underestimated due to mis-specified von Bertalanffy growth parameters from which natural mortality (M) was estimated, or a lack of reliable fishing mortality information, which was generally the case for bycatch species. Further searches of the literature or sourcing unpublished work or expert opinion often provided more reliable information that assisted in producing more reasonable P/B estimates.

Once the aforementioned process was completed for the groups having the most reliable information, it was then applied sequentially to prey of these groups occupying lower trophic levels, and so on.

2.1.3 Diagnostics of Ecopath model validity and stability

On completion of the balancing process, 'best practice' procedures were followed (Heymans *et al.* 2016) and "PREBAL" (Link 2010) diagnostics tests undertaken to assess the biological and ecological validity and overall stability of the balanced model. Once it was confirmed that the underlying biological and ecological assumptions of the Ecopath model had been satisfied using the recommendations of Link (2010) and Heymans *et al.* (2016), the thermodynamic stability of the model was assessed. The model was transferred to Ecosim and run for 1000 years in the absence of fishing to assess the model's temporal stability and to observe whether any group approached extinction, increase indefinitely, or demonstrate erratic or oscillatory trends in their biomass. The model produced a stable 'flat line' for all functional groups with no indication of temporal instability. These diagnostic tests confirmed that the Ecopath model was robust and ready to be used for forecasting changes to the structure of the ecosystem under fishing effort scenarios specified in Ecosim.

2.1.4 Calibration of the ETP7 model to time series data

Although Ecopath is a static representation of the ecosystem, the state of the ecosystem can be estimated using Ecosim (Walters *et al.* 1997). To improve the realism of Ecosim model predictions, it is common for practitioners to calibrate the model to time series of observed[1] population and/or catch trends for specific functional groups, ideally a range of groups representing different trophic levels. The original ETP7 Ecosim model was calibrated to biomass, total mortality and catch for four functional groups representing two of the most economically important species in the EPO tuna fishery: small (<90 cm FL) and large (>90 cm FL) yellowfin tuna, and small (<80 cm FL) and large (>80 cm FL) bigeye tuna.

The present study calibrated the Ecosim model to data time series for 28 of the model's 39 functional groups (Table 1). For species having a stock assessment undertaken in the EPO, time series of biomass, *F* and catch time series were used (all commercially important species), while biomass derived from fishery independent surveys (see survey descriptions by Gerrodette *et al.* 2008) and reported or observed catches were used for cetaceans. In the case of biomass data time series, values were scaled relative to the value for the year used to characterize the Ecopath model (1993). For each of the remaining functional groups, annual catch (retained + discards) for the period 1993–2018 was used.

The data for the 28 functional groups were imported into Ecosim and weighted on an arbitrary scale of 1–5 according to the perceived reliability of the data and its relative importance in driving the model fitting procedure. The 38 most sensitive interactions (*i.e.* the total number of functional groups in the model minus 1) between predators and prey were identified using the non-linear search procedure within Ecosim. The prey vulnerability rate (v)—the rate at which a prey can move between vulnerable and invulnerable states—for the most sensitive predator-prey interactions was then iteratively adjusted until the sums of squares (SS) was minimized to produce the 'best' model fit. This process was repeated 20 times, each from different starting values—and v values being reset before each iteration—to minimize the chance of the non-linear search procedure becoming 'trapped' in local parameter minima. The matrix of vulnerability values by predator/prey interaction that contributed to the fit with the lowest SS from the 20 iterations was then used as the 'optimal' vulnerability matrix.

In searching for the best combination of v values, the time series data were linked to an estimated trend of primary productivity anomalies, in the form of a forcing function, forced upon the "Large phytoplankton" and the "Small producers" groups. A variance value of 10 was used for the model fitting, implying that the model should attempt to capture any abrupt variability in the biomass and catch data

[1] In this paper, the term "observed" refers to quantities that are actually measured (*e.g*. catch), model-derived quantities such as biomass and fishing mortality derived from stock assessment models, or lengths/weights derived from length-weight relationships or age-length curves to differentiate between values estimated by Ecosim (i.e. "predicted").

through time. This procedure did not significantly reduce the SS and so the forcing function was removed and not included in any of the final Ecosim simulations.

Once the optimal vulnerability search was complete, *v* values were inspected to ensure they were ecologically realistic. Although some adjustments were initially made to *v* values that often resulted in better visual fits of the Ecosim model to observed data, they often resulted in a poorer statistical fit (lower SS) or required unrealistic parameter values, so they were disregarded. Further detailed descriptions of fitting Ecosim models to time-series data can be found in Christensen *et al.* (2009).

2.1 Ecological indicators

Annual values for 7 ecological indicators describing the ETP were estimated in Ecosim. A full description of available ecological indicators and the justification for use of the 7 indicators chosen to characterize the EPO ecosystem is provided in Griffiths and Duffy (2019). These consist of three catch-based (TL_c, Marine trophic index, Fishing in Balance index) and four community-based indicators (Mean trophic level of the community for trophic levels 2.0–3.25, ≥3.25–4.0, and >4.0), based on the recommendations of Shannon *et al.* (2014). A brief 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 indicator of how 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 by overfishing can result in fisheries progressively targeting species at decreasing trophic levels to remain profitable. Studies that have documented this "fishing down the food web" (Pauly *et al.* 1998), have shown that the TL_c decreased by around 0.1 of a trophic level per decade and is the magnitude of change considered in the present assessment.

2.2.2 Marine trophic index (MTI)

The marine trophic index (MTI) is similar to TL_c, but it only includes high trophic level species (TL>4.0) that are usually the first indicator of 'fishing down the food web'. Some ecosystems, however, have changed in the opposing 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". In other situations, the MTI can increase due to improved catch reporting, whereby previously unreported catches of discarded predatory species, such as sharks, are now recorded.

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 causing disruption to the functionality of the ecosystem. FIB incorporates the MTI and can provide an indication of overfishing when catches do not increase as expected (or a decrease in TL_c) given 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 community (TL_{MC}) modelled by Ecopath was described by Shannon *et al.* (2014), where they estimated the mean trophic level for specific components of an ecosystem. These indicators allow the researcher to examine changes in the ecosystem structure after biomass removals by fishing. In the case of the EPO, TL thresholds were 2.0–3.25, ≥3.25–4.0, and >4.0. These indicators can be used in unison to detect trophic cascades, whereby a decline in biomass of $TL_{MC4.0}$ due to fishing, would be expected to increase the biomass of $TL_{MC3.25}$, as predation pressure is reduced, which in turn would decrease the biomass of $TL_{MC2.0}$, which would subsequently experience higher predation pressure.

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 their relative proportion 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. Thus, in the case of an Ecopath model, the relative difference in the biomass of functional groups.

2.2 Simulating the ecological impacts of proposed tuna conservation measures

Simulations of maintaining or changing the number of OBJ sets were undertaken in Ecosim to explore the potential ecological consequences of changing fishing effort regimes over the next 6 years to 2024. Specifically, the following two management scenarios were simulated.

Status quo: maintaining the 2018 effort levels for all fisheries to 2024,

OBJ increase: linearly increasing the number of OBJ sets from 11,871 sets in 2018 to 13,883 in 2024.

3 RESULTS AND DISCUSSION

3.1 Rebalancing and fitting the model to time series data

The rebalancing process successfully resulted in a thermodynamically stable model where the biomass of no functional group deviated by more than 0.01% over a 1000-year simulation in the absence of fishing. The subsequent model fitting to time series data resulted in good fits to observed biomass and/or catch data for most functional groups (Figures 2 and 3). Exceptions were fits to catch for small yellowfin tuna and biomass for small bigeye tuna prior to 2000, which was also identified as a period of abrupt change in the stock assessments for these species (Minte-Vera *et al.* 2020; Xu *et al.* 2020).

Another exception was large sharks where although the general biomass trend was reproduced by Ecosim, the estimated catches were significantly underestimated prior to 2007 and overestimated thereafter. This is likely a result of using silky shark population indicators to represent a large number of shark species whereby some species may have had opposing trends in biomass. For example, it is now well accepted that the populations of silky and oceanic white-tip sharks have declined significantly over the past three decades (BYC-10 INF-A), but the total catch of other large sharks such as blue, mako and hammerhead sharks have increased over the past decade (see Figure J-3 in IATTC, 2020a). Therefore, using the declining silky shark population biomass trend, the model predicted a concordant trend in catch. Similar explanations can be made for the poor catch predictions made for toothed whales, spotted dolphins and mesopelagic dolphins prior to about 2005, where fishery independent surveys established estimates of absolute abundance for individual species, while catches are available for a suite of species.

Overall, the results from the model fits suggest that the ETP-21 Ecosim model is capable of reproducing previous biomass states of key functional groups, and therefore providing some confidence that the model is likely capable of predicting the future state of the ecosystem for prescribed fishing effort regimes.

3.2 Status of the EPO ecosystem as interpreted from ecological indicators

Figure 4 shows the trends in ecological indicator values for assessments conducted in 2019 (gray lines) and 2021 (color lines). The addition of the new 'industrial' longline data and the disaggregated data submitted by CPCs for their domestic fisheries significantly changed the magnitude of values for all 7 indicators. Most importantly, the peak in the mean trophic level of the catch (TLc) in 1991 was estimated to be around 4.69 and 4.77 in the 2019 and current assessments, respectively. These values declined to 4.65 in 2017, in both assessments indicating that the decline in TLc is likely to be larger than previously assumed. In the current assessment, TLc declined by about 0.12 of a trophic level over the past 28 years, or 0.04 trophic levels per decade.

After 1991, TL_c continued to decrease to around 1996, due to the rapid expansion of the fishery from 1993 where there were increasing catches in the intervening period of high trophic level bycatch species that also aggregate around floating objects (*e.g.* sharks, billfish, wahoo and dorado). This expansion is seen in the FIB index that exceeds zero during the same period, and a change in the composition of the community indicated by a sharp decrease in Shannon's index (Figure 4). From the early 2000s, TL_c, MTI, and Shannon's index all show a continual decline, while the FIB gradually increased further from zero to its peak in 2018 at 0.97—substantially higher than the peak of 0.66 in the 2019 assessment (Figure 4). Both TL_c and MTI reached among their lowest historic levels in 2018 (Figure 4).

The aforementioned indicators generally describe the change in the exploited components of the ecosystem, whereas community biomass indicators describe changes in the structure of the ecosystem once biomass has been removed due to fishing. Figure 4 shows the biomass of the $TL_{MC4.0}$ community in 1986 (4.493) just prior to the increase in FAD fishing effort in 1993 but has continued to decline to 4.47 in 2018. As a result of changes in predation pressure on lower trophic levels, between 1993 and 2018 the biomass of the $TL_{MC3.25}$ community increased from 3.801 to 3.829, while the biomass of the $TL_{MC2.0}$ community increased from 3.092 to 3.107.

Together, these indicators show that there has likely been a change in the ecosystem structure over the 40-year analysis period. The consistent patterns of change in each ecological indicator, particularly in the mean trophic level of the communities since 1993, certainly warrant the continuation, and ideally an expansion, of monitoring programs of fisheries in the EPO.

3.3 Simulating the potential ecological impacts of changing FAD fishing effort in the EPO

3.3.1 Biomass changes for key species

Maintaining and increasing the number of OBJ sets both resulted in the same direction—but differing magnitude—of change in the biomass of the 15 functional groups shown in Figure 3. Specifically, this was a decrease in biomass of all target tuna species (yellowfin, bigeye and skipjack tunas) and small and large sharks, and an increase in biomass of bycatch species including both small and large size classes of marlins, dorado and wahoo, as well as rays and turtles (Figure 5).

For functional groups that were predicted to decline in biomass, under the *status quo* scenario the predicted decline in biomass in target species was less than 3%, while the decline in small and large sharks was 8.6% and 6.8%, respectively (Figure 5). Under scenario where the number of OBJ sets increased linearly to 2024, the biomass of target species was expected to decrease by 4.7% and 5.4% for yellowfin and bigeye tunas, respectively, but only 0.67% for skipjack. However, the decline in the biomass of small and large sharks substantially decreased to 13.8% and 10.4%, respectively.

For functional groups that were predicted to increase in biomass, under the *status quo* scenario the increase in biomass for bycatch species that are frequently retained was expected to be between 1.1% (large marlins) and 3.3% (small dorado), and between 0.7% and 1.1% turtles and rays and, respectively, which are released or discarded (Figure 5). Although many bycatch species that are frequently retained due to their economic value (*e.g*., billfish and dorado) have a high affinity with floating objects, it may appear surprisingly that the biomass of these species groups increased by a further 1–2% under the increasing OBJ sets scenario. Although this increase seems counterintuitive given the increase in fishing mortality on these species, this type of result is common in ecological systems, where 'the predator of my predator is my friend'. The increase in FAD effort caused a substantial decrease in the biomass of the primary predators of these species (*i.e.* small and large sharks and tunas). The predation pressure was so high (*e.g.* small wahoo 1.42 yr^{-1}, large dorado 1.654 yr^{-1}) that the increase in fishing mortality from 2018 to 2024 (small wahoo 0.006 yr^{-1} to 0.008 yr^{-1}, large dorado 0.002 yr^{-1} to 0.006 yr^{-1}) was negligible by comparison, thus allowing the biomasses of these bycatch species to increase.

3.3.2 Changes in ecosystem structure

Figure 6 shows the simulated changes in values of seven ecological indicators under the two management scenarios. As for biomasses of key species, both scenarios resulted in the same direction—but differing magnitude—of change in the values from each ecological indicator. Continuing the 2018 fishing effort regime to 2024 will result in declines in TL_c, MTI, Shannon's index, and $TL_{MC4.0}$, and increases in the FIB, $TL_{MC3.25}$, and $TL_{MC2.0}$. Increasing the number of OBJ sets is predicted to have the same directional change as continuing the *status quo*, but the magnitude of this change is higher. Together, these indicators show that maintaining or increasing the number of OBJ sets even over a period as short as 6 years is likely to change the dynamics of the ETP ecosystem further from its likely already altered state assumed to be caused primarily by decades of industrial fishing (FIB index; Figure 4). Although the 6-year simulation period is too short to identify potentially detrimental ecological consequences of fishing, such as trophic cascades, perpetual increases in purse-seine fishing effort, coupled with the impacts of the industrial longline and coastal fisheries and a changing climate, may eventually drive the ecosystem to a tipping point from which its altered internal dynamics can no longer be reversed by any level of fisheries management intervention (see Travis *et al.* 2014).

3.4 Considerations for future work

There are a number of assumptions to consider when interpreting the simulation results of the present study. Since the primary objective of the work was to explore the potential ecological impacts of maintaining or increasing the number of OBJ sets, which has dramatically increased by around 50% every 5 years since 1993, it was assumed that there would be no change in effort in any other EPO tuna fishery during the 6-year simulation period. This is unlikely to be the case, particularly for the longline fishery, where effort in the EPO approximately doubled in the period 2008–2015 (Griffiths and Duffy 2017). Since the longline fishery primarily targets large bigeye tuna, swordfish, and albacore, a continued increase in OBJ effort, as has been observed over the past 10 years, may begin to present a significant interaction with the purse-seine fishery that targets small bigeye tuna. As was predicted by the model, even increasing the number of OBJ sets by 2,012 over the next 6 years may have a reasonable impact on small bigeye tuna (5.37%), which was predicted to result in a reduction in the adult stock biomass of 3.78%.

An important consideration when interpreting the results from the ETP-21 model is that the model's underlying diet matrix—the component of the model that defines the trophic linkages between species in the ecosystem—has remained unchanged from the original ETP7 model as is based on stomach content data from fish collected over two decades ago. This period was before the expansion of the FAD fishery, and as demonstrated in the results of the analyses in this paper, the trophodynamics of the ecosystem is likely to have significantly changed, bringing in to question the realism of ecosystem model predictions for use in tactical fisheries management. 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 that have been observed in the EPO over the past decade, in the form of some of the strongest El Niño events on record (Kintisch 2015; Cai *et al.* 2018), it stands to reason that there is a critical need to collect trophic information from 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 (*i.e.* phytoplankton), which can have a significant impact in oligotrophic oceanic ecosystems that are often thought to be controlled by 'bottom-up' processes (Hunt and McKinnell 2006).

Sampling of predator stomachs has been successful in collaboration with observers from the IATTC and national programs (*e.g.* Olson and Galvan-Magaña 2002; Olson *et al.* 2014), albeit nearly two decades ago. Therefore, a consideration for future research at the IATTC arising from this work is a trophic sampling program relying on observer collections, as well as through collaboration with CPCs, and other stakeholders (*e.g.* universities, research institutes), who may have a vested interest in having access to an up-to-date and reliable ecosystem model that can be used to explore the potential ecological consequences of future anthropogenic and/or environmental impacts.

Similarly, many of the biological parameters in the ETP-21 model have been derived from other ocean basins or related species, thus introducing some uncertainty in the magnitude of estimated trophic flows in the ecosystem. The consumption to biomass ratio (Q/B) is an influential parameter in Ecopath models, as it describes the energy requirements of predators and the required standing biomass of their prey. Unfortunately, it is one of the most difficult parameters to measure for pelagic fishes, since large and highly specialized facilities are required to hold large predators such as tunas in order to undertake experiments on 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).

Fortuitously, the IATTC's Achotines facility in Panama is equipped to undertake such experiments, with several species of large pelagic fishes available for capture for experiments in nearby waters, where the continental shelf break exists around 12 km from the coast. The staff is exploring the possibility of undertaking projects at the Achotines Laboratory to begin to fill the data gaps in our knowledge in the trophic ecology of pelagic fishes in the EPO, which will in turn provide the most reliable parameter estimates for future ecosystem models for the EPO. The IATTC staff are currently undertaking a comprehensive review of methods to experimentally determine Q/B that may be applied to a range of pelagic predators that occupy different trophic levels in the EPO ecosystem and are locally abundant in the waters adjacent to the Achotines facility (SAC-10 INF-E).

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Table 1. Time series of data used to calibrate the ETP-21 Ecopath model to 2018.
Tabla 1. Series de tiempo de datos utilizadas para calibrar el modelo Ecopath ETP-21 a 2018.

Functional group	Data types	Fisheries	Time period	Data source
Small yellowfin tuna	Biomass, *F*, Catch	All	1979–2018	Minte-Vera *et al.* (2020), IATTC (2020b)[2]
Large yellowfin tuna	Biomass, *F*, Catch	All	1979–2018	Minte-Vera *et al.* (2020)[2]
Small bigeye tuna	Biomass, *F*, Catch	All	1979–2018	Xu *et al.* (2020)[2]
Large bigeye tuna	Biomass, *F*, Catch	All	1979–2018	Xu *et al.* (2020)[2]
Small dorado	Biomass, *F*, Catch	All	2008–2014	Aires-da-Silva *et al.* (2017)
Large dorado	Biomass, *F*, Catch	All	2008–2014	Aires-da-Silva *et al.* (2017)
Small sharks	Catch	All	1993–2018	IATTC unpublished data
Large sharks	Relative abundance, Catch	Purse-Seine	1994–2018	Lennert-Cody *et al.* (2019)
Toothed whales	Numbers, Catch	Purse-Seine	1999, 2001, 2004, 2006, 2006	Gerrodette *et al.* (2008)
Spotted dolphins	Numbers, Catch	Purse-Seine	1999, 2001, 2004, 2006, 2006	Gerrodette *et al.* (2008)
Mesopelagic dolphins	Numbers, Catch	Purse-Seine	1999, 2001, 2004, 2006, 2006	Gerrodette *et al.* (2008)
Skipjack	Catch	All	1993–2018	IATTC unpublished data
Albacore	Catch	All	1993–2018	IATTC unpublished data
Pacific bluefin tuna	Catch	All	1993–2018	IATTC unpublished data
Auxis spp.	Catch	All	1993–2018	IATTC unpublished data
Small marlins	Catch	All	1993–2018	IATTC unpublished data
Large marlins	Catch	All	1993–2018	IATTC unpublished data
Small sailfish	Catch	All	1993–2018	IATTC unpublished data
Large sailfish	Catch	All	1993–2018	IATTC unpublished data
Small swordfish	Catch	All	1993–2018	IATTC unpublished data
Large swordfish	Catch	All	1993–2018	IATTC unpublished data
Small wahoo	Catch	All	1993–2018	IATTC unpublished data
Large wahoo	Catch	All	1993–2018	IATTC unpublished data
Rays	Catch	All	1993–2018	IATTC unpublished data
Turtles	Catch	All	1993–2018	IATTC unpublished data

[2] Multimodel estimates were computed taking into account all models used in the benchmark assessment and their weights

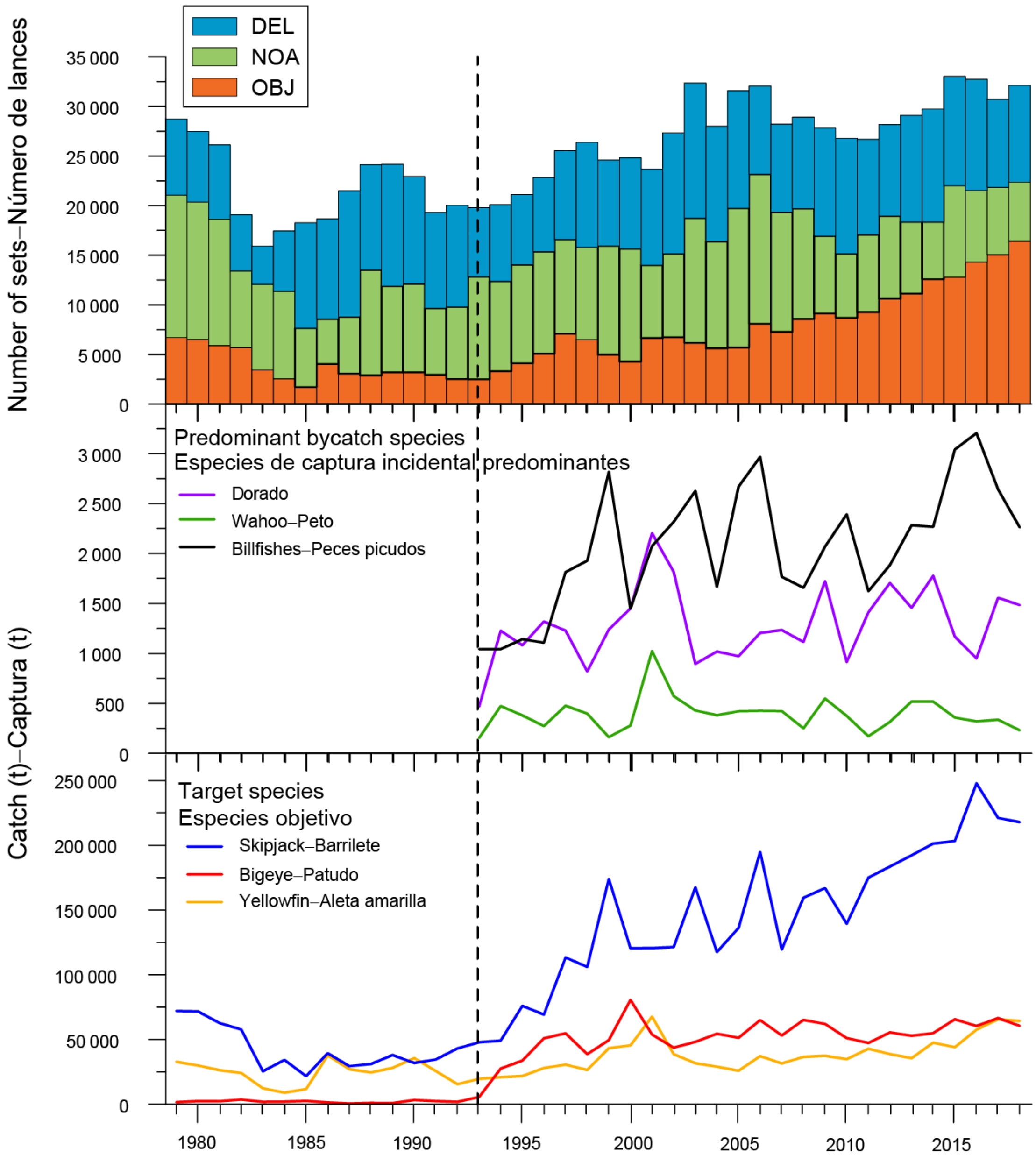

FIGURE 1. Time series of annual number of sets on dolphins (DEL), unassociated tuna schools (NOA) and floating objects (OBJ) in the EPO purse-seine fishery for 1979–2018 (top panel), with the total annual reported catch from OBJ sets for the predominant retained bycatch species of dorado, wahoo and billfishes (middle panel), and target species of skipjack, yellowfin and bigeye tunas (bottom panel). The dashed vertical line indicates the approximate year (1993) when the fishery began to increasingly set on artificial drifting floating objects and when reporting of non-target species began.

FIGURA 1. Series de tiempo del número anual de lances sobre delfines (DEL), cardúmenes de atunes no asociados (NOA) y objetos flotantes (OBJ) en la pesquería de cerco en el OPO para 1979-2018 (panel superior), con la captura anual total notificada de lances OBJ para las especies de captura incidental retenidas predominantes, a saber el dorado, el peto y los peces picudos (panel central) y especies objetivo, a saber el atún barrilete, aleta amarilla y patudo (panel inferior). La línea de trazos vertical indica el año aproximado (1993) en que la pesquería empezó a realizar cada vez más lances sobre objetos flotantes artificiales a la deriva y se empezó a informar sobre las especies no objetivo.

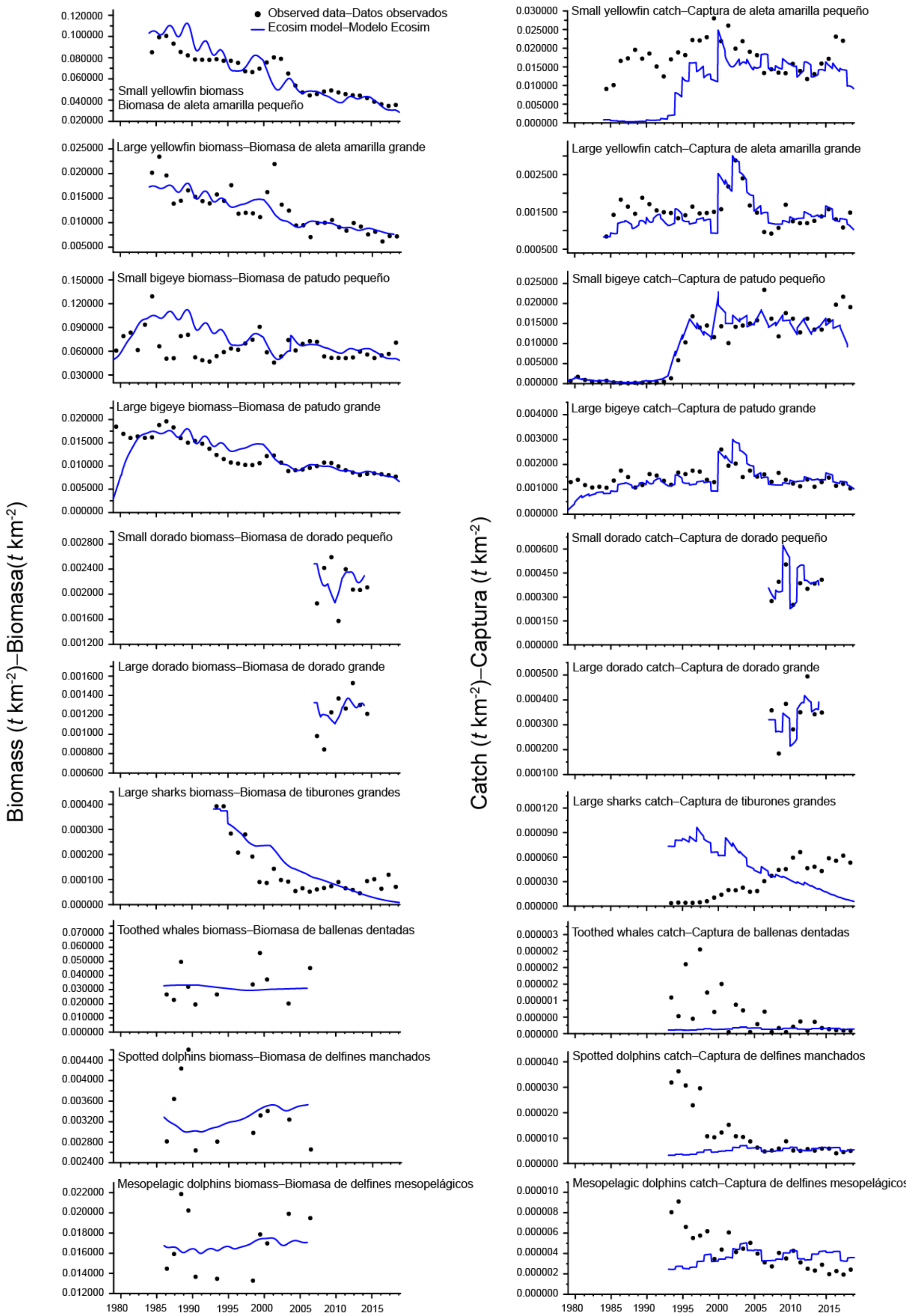

FIGURE 2. Ecosim model fits (solid lines) to observed[1] annual time-series data (black dots) for biomass (t km^{-2}) and catches (t km^{-2}) for 10 functional groups in the ETP-21 ecosystem model for 1979–2018.
FIGURA 2. Ajustes del modelo Ecosim (líneas sólidas) a datos observados[1] de series de tiempo anuales (puntos negros) para la biomasa (t km^{-2}) y las capturas (t km^{-2}) para 10 grupos funcionales en el modelo ecosistémico ETP-21 para 1979-2018.

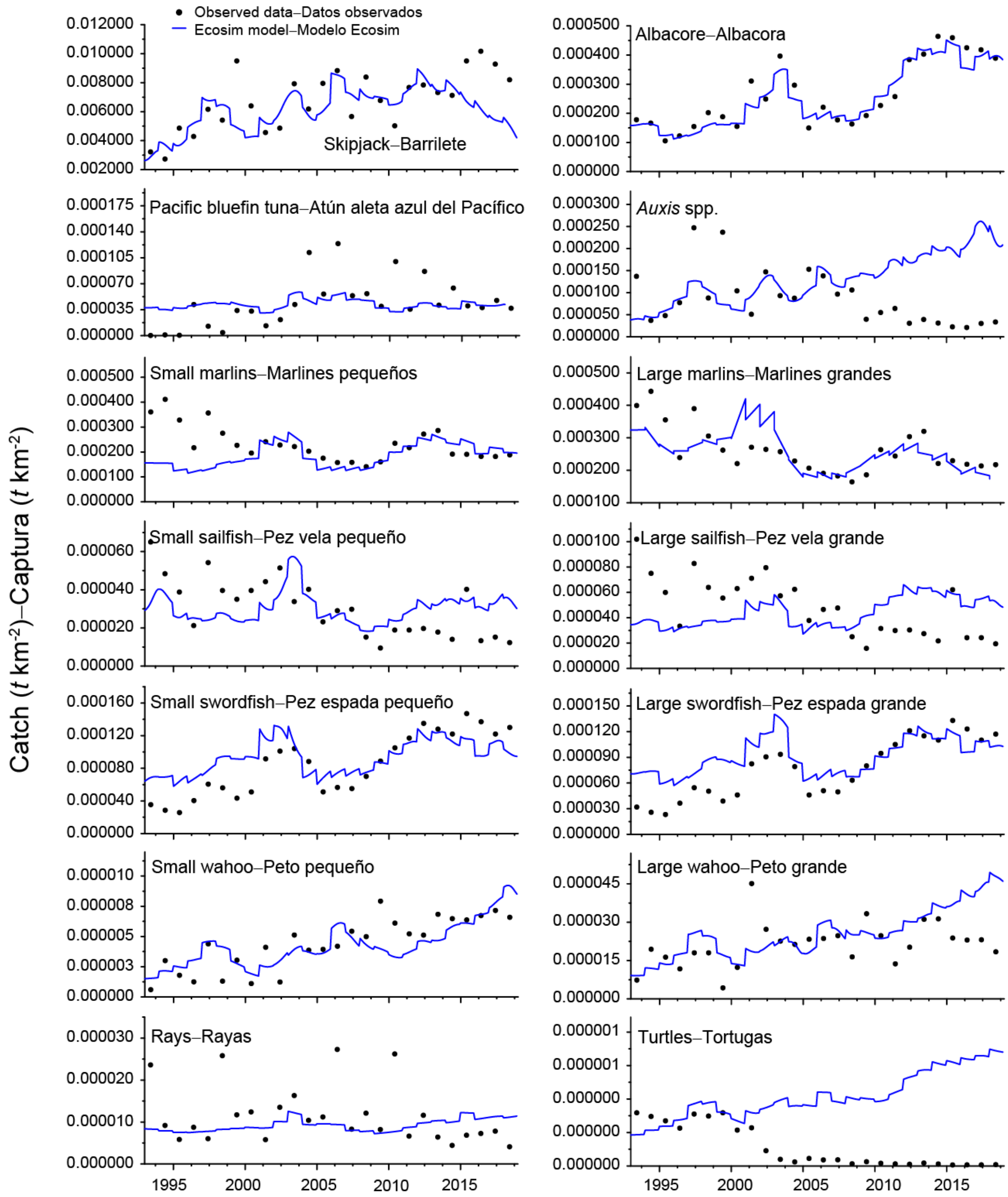

FIGURE 3. Ecosim model fits (solid lines) to observed annual time-series of catch data (t km^{-2}) (black dots) for 14 functional groups in the ETP-21 ecosystem model for which catch data were available.
FIGURA 3. Ajustes del modelo Ecosim (líneas sólidas) a series de tiempo anuales observadas de datos de captura (t km^{-2}) (puntos negros) para 14 grupos funcionales en el modelo ecosistémico ETP-21 para los que se disponía de datos de captura.

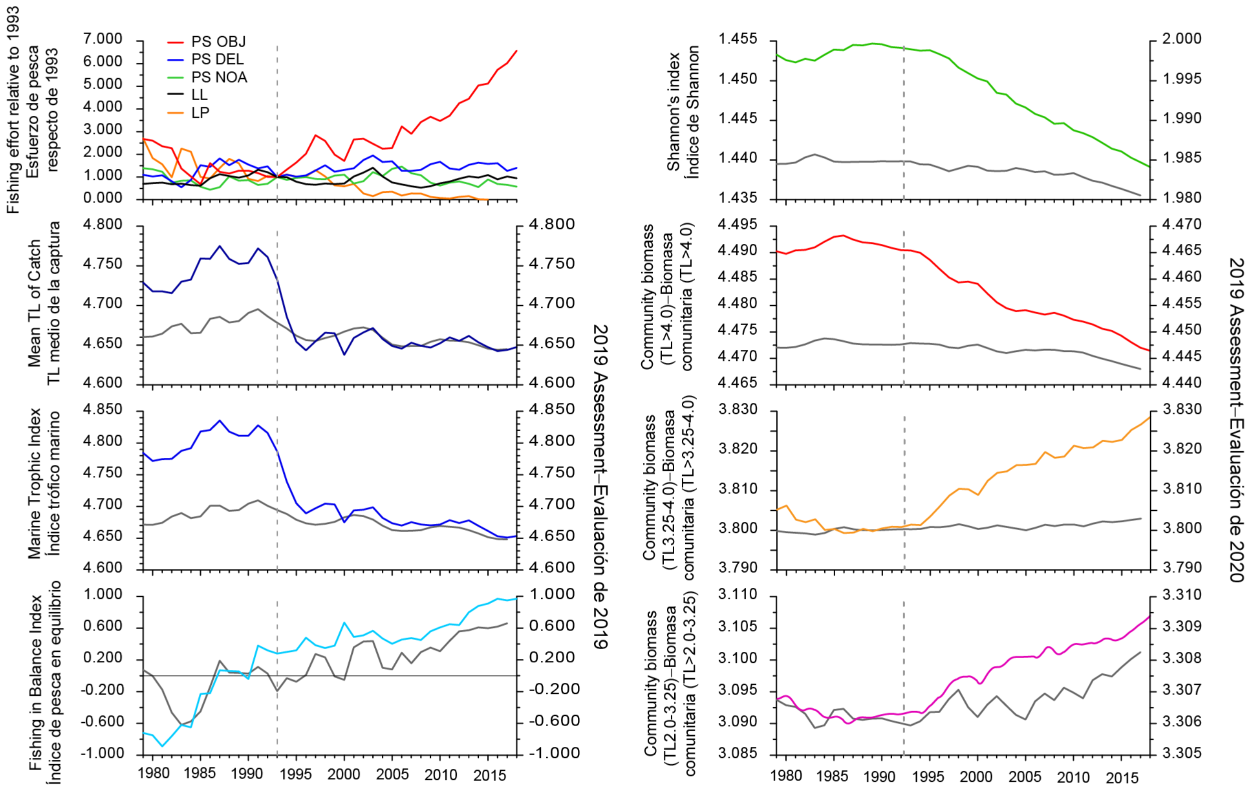

FIGURE 4. Annual values derived from a trophic mass-balance model of the eastern tropical Pacific Ocean ecosystem for seven ecological indicators that describe the changes in different components of the tropical EPO ecosystem, 1979–2018. The indicators include: mean trophic level of the catch, Marine Trophic Index, Fishing in Balance index, Shannon's index and the mean trophic level of the community for trophic levels 2.0–3.25, 3.25–4.0, and >4.0. Solid gray lines on each graph show the results from the 2019 assessment, prior to new longline data being added. An index of fishing effort—represented as effort relative to the ecosystem model's reference year of 1993—is shown for the same time period for purse-seine fisheries (OBJ, DEL, and NOA sets), the longline fishery, and the pole and line fishery. Vertical gray dashed lines denotes the approximate year (1993) when the purse-seine fishery began to change their fishing strategy to fish on artificial drifting floating objects.

FIGURA 4. Valores anuales derivados de un modelo trófico de balance de masas del ecosistema del Océano Pacífico oriental tropical para siete indicadores ecológicos que describen los cambios en diferentes componentes del ecosistema del OPO tropical, 1979-2018. Los indicadores son el nivel trófico medio de la captura, el índice trófico marino, el índice de pesca en equilibrio, el índice de Shannon y el nivel trófico medio de la comunidad para los niveles tróficos 2.0–3.25, 3.25–4.0 y >4.0. Las líneas grises sólidas de cada gráfica muestran los resultados de la evaluación de 2019, antes de la incorporación de nuevos datos de palangre. Se muestra un índice de esfuerzo de pesca, representado como el esfuerzo respecto del año de referencia del modelo ecosistémico de 1993, para el mismo periodo de tiempo para las pesquerías de cerco (lances OBJ, DEL y NOA), de palangre y de caña. Las líneas grises verticales de trazos indican el año aproximado (1993) en que la pesquería cerquera empezó a cambiar su estrategia de pesca para pescar sobre objetos flotantes artificiales a la deriva.

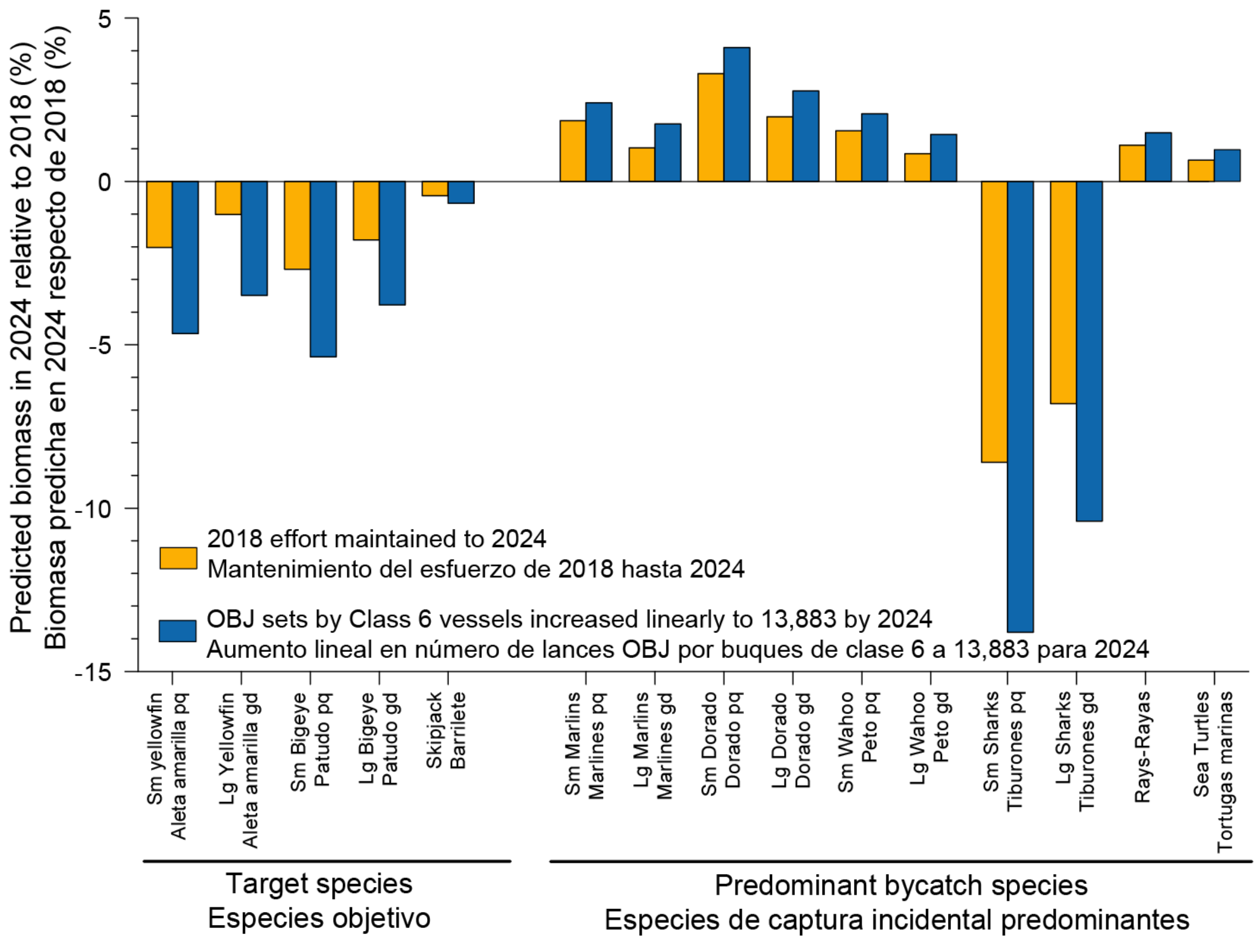

FIGURE 5. Ecosim predicted relative changes in the biomass of key functional groups representing principal target tuna species and predominant bycatch species caught by the purse-seine fishery in the eastern tropical Pacific Ocean in 2024 relative to 2018 under two hypothetical management scenarios. The scenarios included: 1) maintaining the *status quo* of 11,871 OBJ sets in 2018 until 2024, and 2) imposing a linear increase in the number of OBJ sets by Class 6 purse-seine vessels from 11,871 sets in 2018 to 13,883 in 2024. Effort for all other fisheries remained at their 2018 levels for both scenarios.

FIGURA 5. Cambios relativos predichos por Ecosim en la biomasa de grupos funcionales clave que representan las principales especies de atún objetivo y las especies de captura incidental predominantes capturadas por la pesquería cerquera en el Océano Pacífico oriental tropical en 2024 respecto de 2018, bajo dos escenarios hipotéticos de ordenación: 1) el mantenimiento del *statu quo* de 11,871 lances OBJ en 2018 hasta 2024 y 2) un aumento lineal en el número de lances OBJ por parte de buques cerqueros de clase 6, de 11,871 lances en 2018 a 13,883 en 2024. El esfuerzo de todas las demás pesquerías se mantuvo en su nivel de 2018 en ambos escenarios.

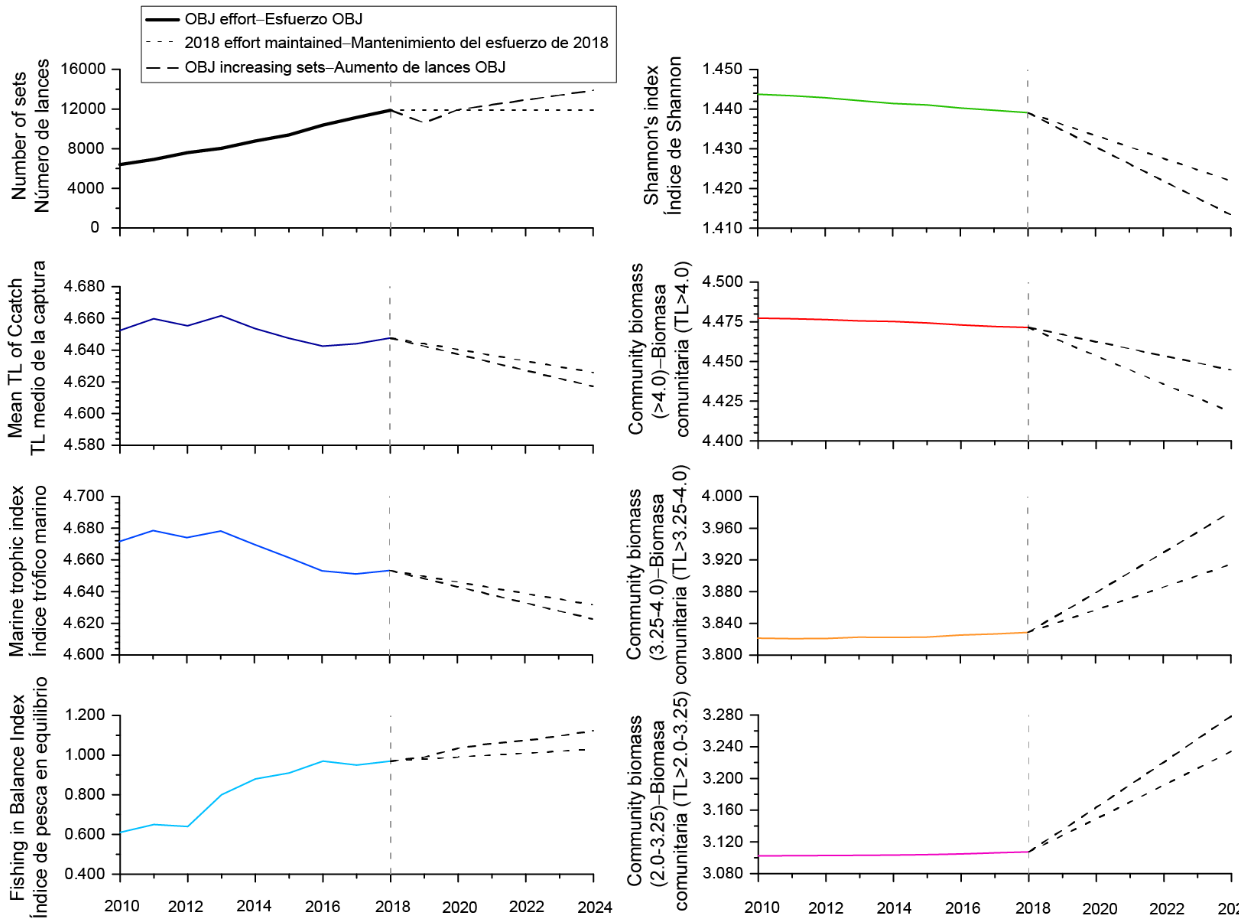

FIGURE 6. Estimated changes in annual values for seven ecological indicators after the simulation of two hypothetical scenarios changing the effort of the purse-seine fishery on floating objects (OBJ) over a 6-year period initiated in 2018 and concluding in 2024. The scenarios included: 1) maintaining the *status quo* of 11,871 OBJ sets in 2018 until 2024, and 2) imposing a linear increase in the number of OBJ sets by Class 6 purse-seine vessels from 11,871 sets in 2018 to 13,883 in 2024. Effort for all other fisheries remained at their 2018 levels for both scenarios. The vertical gray dashed lines denote the year (2018) when the simulations began.

FIGURA 6. Cambios estimados en los valores anuales de siete indicadores ecológicos tras simular dos escenarios hipotéticos en los que cambia el esfuerzo de la pesquería cerquera sobre objetos flotantes (OBJ) a lo largo de un periodo de 6 años que empieza en 2018 y termina en 2024: 1) el mantenimiento del *statu quo* de 11,871 lances OBJ en 2018 hasta 2024 y 2) un aumento lineal en el número de lances OBJ por parte de buques cerqueros de clase 6, de 11,871 lances en 2018 a 13,883 en 2024. El esfuerzo de todas las demás pesquerías se mantuvo en su nivel de 2018 en ambos escenarios. Las líneas grises verticales de trazos indican el año (2018) en que empezaron las simulaciones.