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Leveraging quantitative techniques for fostering holistic approaches to support sustainable fisheries

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Leveraging quantitative techniques for fostering holistic approaches to support sustainable fisheries

A Dissertation submitted in partial satisfaction of the requirements for the degree Doctor of Philosophy

in

Marine Biology with a Specialization in Interdisciplinary Environmental Research

by

Jordan Haley DiNardo

Committee in charge:

Professor Brice X. Semmens, Chair
Professor Octavio Aburto-Oropeza
Professor Jeffrey M. Haydu
Professor Ed Parnell

2024

The Dissertation of Jordan Haley DiNardo is approved, and it is acceptable in quality and form for publication on microfilm and electronically.

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DEDICATION

To the generations of women before me who courageously pushed boundaries and paved the way for my journey today. I am so grateful.

And to my parents, thank you for all your love and support.
I would not be the woman I am today without you.
This is for you.

EPIGRAPH

“Eventually man, too, found his way back to the sea. Standing on its shores, he must have looked out upon it with wonder and curiosity, compounded with an unconscious recognition of his lineage.”

- Rachel Carlson

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ABSTRACT OF THE DISSERTATION

Leveraging quantitative techniques for fostering holistic approaches to support sustainable fisheries

by

Jordan Haley DiNardo

Doctor of Philosophy in Marine Biology with a Specialization in Interdisciplinary Environmental Research

University of California San Diego, 2024

Professor Brice X. Semmens, Chair

Fisheries play a vital role ensuring nutritional security, while also holding economic and cultural importance globally. Considering their significance, it is essential to focus efforts to ensure our fisheries are sustainable. Efforts to improve the sustainability of fisheries embrace diverse approaches, comprising both top-down and bottom-up strategies. While top-down tactics entail development of fisheries management measures and international cooperation, bottom-up

approaches emphasize community-driven initiatives. In pursuit of sustainable fisheries, adopting a holistic approach that incorporates both top-down and bottom-up strategies can be more effective than relying solely on one strategy. In this context, leveraging the best available science is key to inform and enhance these strategies, maximizing their effectiveness. My dissertation explores this notion by conducting science to inform and improve current top-down and bottom-up strategies, all aimed at enhancing the sustainability of fisheries. Chapter 1 investigates the recovery of the northern Pacific Bluefin Tuna (*Thunnus orientalis*) in the eastern Pacific Ocean after intensive international management to recover the species. Chapter 2 reconstructs population trends of Caribbean parrotfishes (family Scaridae) by leveraging a long-term citizen-scientist diving program to explore parrotfish dynamics and provide insights to promote parrotfish conservation more effectively. Chapter 3 investigates consumer perception and understanding of seafood ecolabels, a market-based tool aimed at promoting sustainable fisheries practices from the bottom-up, highlighting the need for increased community education and engagement. Collectively this dissertation underscores the need for diverse strategies to achieve fisheries sustainability.

Chapter 1

Standardization of CPUE for Pacific bluefin tuna (*Thunnus orientalis*) from the United States multispecies, recreational fishery in the eastern Pacific Ocean

Jordan H. DiNardo, Julia Coates, Brice X. Semmens

Abstract

Pacific bluefin tuna (PBF), *Thunnus orientalis*, are apex predators in the Pacific and highly valued in terms of both economics and food production. However, since the 1990s, the PBF stock has been highly depressed, ostensibly due to overharvest and mismanagement. More recently, aggressive reductions in PBF catch limits globally have resulted in a significant increase in PBF in the western Pacific Ocean (WPO), indicating the stock is recovering. However, it remains unclear whether the apparent benefits of fisheries management have manifested in catch in the eastern Pacific Ocean as PBF stock assessments and harvest strategy projections only include abundances indices from the WPO. In this study, we use catch and effort data from the CPFV multispecies fishery over the last two decades to track the progress of the PBF recovery in the EPO. Our results indicate early signs of recovery in the EPO that can be attributable to favorable environmental conditions and ongoing management efforts. Collectively, this work enables a Pacific-wide assessment of PBF recovery efforts by supporting the development of stock-wide assessments and harvest strategy projections, implementation of future management measures, and investigation of climate change impacts on PBF in the EPO.

Introduction

The Pacific bluefin tuna (PBF), *Thunnus orientalis,* is a highly valued commodity in the seafood industry and an important source of protein worldwide. As a delicacy in sushi and sashimi markets, a single fish can sell for a few million US dollars in Japan, depending on size, quality, and time of year (McCarthy, 2020). To meet global demand for the species, several tuna fisheries predominantly originating from Japan, Taiwan, Korea, Mexico, and the United States (U.S.) exploit PBF in their respective regions of the north Pacific Ocean, the western Pacific Ocean (WPO) and the eastern Pacific Ocean (EPO). Landings records dating back to the early 1900s indicate PBF catch was highest in 1935 (~ 47,635 tons; WPO: 36,217 tons, EPO: 11,418 tons), thereafter fluctuating in a downward trend to a low in 1990 (8,653 tons; (ISC, 2022). PBF exhibit a trans-pacific migration, where immature PBF (ages 1-2) in the WPO migrate to the EPO (Inagake et al., 2001), where they spend 3-4 years as juveniles before returning to the WPO to spawn at age ≥ 3 years (Kawazu et al., 2020; Madigan et al., 2014, 2017; Tawa et al., 2017). Due to this ontogenetic migration pattern of PBF, fisheries operating in the WPO target immature and mature PBF, while fisheries in the EPO target juveniles PBF (ISC, 2022).

PBF are jointly managed by two regional fisheries management organizations (RFMOs) – the Western and Central Pacific Fisheries Commission (WCPFC) in the WPO and the Inter-American Tropical Tuna Commission (IATTC) in the EPO. However, science to support PBF management, including stock assessments that assess stock status, is conducted by the International Scientific Committee (ISC) for Tuna and Tuna-like Species in the North Pacific Ocean. The ISC concluded PBF was severely depleted in 2010 (ISC, 2012), prompting the RFMOs to establish catch limits by country in 2011 (WCPFC, 2010) and 2012 (IATTC, 2012). The ISC updated the stock assessment in 2014 and found no change in stock status. The stock was still considered overfished and experiencing overfishing (ISC, 2014). Accordingly, the WCPFC adopted and implemented a provisional multi-year rebuilding plan for PBF with the initial rebuilding target (median spawning stock biomass (SSB) between 1952 to 2014) to be met within 10 years (in 2024) with at least 60% probability (WCPFC, 2014). The implementation and progress of this rebuilding plan was reviewed based on the results of stock assessments and SSB projections conducted by ISC that compared projected trajectories of PBF SSB under varying management scenarios (e.g., spatial catch limits by fish size) relative to the likelihood of meeting the initial rebuilding target and potential long-term rebuilding goals (still to be defined). As recommended by further analyses size-specific catch limits in the WPO were further reduced in

2014 (WCPFC, 2013) and 2015 (WCPFC, 2014) and catch limits in the EPO reduced in 2015 and 2016 (IATTC, 2014). In 2017, the WCPFC adopted a second rebuilding target (20%SSB_0) to be met 10 years after the initial rebuilding target or by 2034, whichever is earlier, with at least 60% probability (WCPFC, 2017). These two rebuilding targets comprise the PBF rebuilding plan and this is considered to represent the PBF harvest strategy.

PBF has shown signs of recovery following the adoption of the harvest strategy (WCPFC, 2017) and establishment of age-specific catch limits (ISC, 2022). In the 2022 stock assessment, the ISC concluded the stock size had significantly increased, although the stock was still overfished. In fact, since the implementation of the 2015 rebuilding plan, the recovery of PBF has occurred at a faster rate than anticipated. The stock exceeded the initial rebuilding target five years earlier than expected and is likely (probabilities > 90%) to meet the second rebuilding target by 2029 (ISC, 2022). It is unclear if this recovery of PBF, as documented, is reflected in the EPO since the ISC stock assessments and the harvest strategy projections only contain abundance information from the WPO. Given that PBF are the target of economically valuable Mexican and U.S. commercial purse seine fisheries and the U.S. recreational fishery (NOAA Fisheries, 2023) that have non-negligible catches of PBF, it is imperative to assess and document their recovery in the EPO.

The U.S. recreational fishery operates along the west coast of the U.S. and Baja California, Mexico (ISC, 2022) and is largely made up of Commercial Passenger Fishing Vessels (CPFVs). CPFVs typically can accommodate up to 100 paying passengers (mean: 20.09; SD: 9.89) on fishing trips ranging from hours to days, although multi-day trips typically accommodate fewer passengers (typically up to 40 passengers). Recreational fishers target multiple species, including PBF, using hook-and-line. While the fishery operates year-round, peak fishing season for PBF and other tuna species is in summer and fall. The PBF fishing season is largely informed by the seasonal north-south migrations of PBF in the EPO. During the spring PBF generally reside off the southern coast of Baja California, Mexico. As water temperatures increase in the summer PBF migrate northwest into the southern California Bight and further north into central California in the fall. As waters cool in the winter PBF return to the south (Boustany et al., 2010; Domeier et al., 2005; Fujioka et al., 2015; Kitagawa et al., 2007). These spatio-temporal dynamics of PBF in the EPO are most likely driven by seasonal prey abundance induced by favorable environmental conditions (Boustany et al., 2010; Kitagawa et al., 2007).

Recreational anglers are restricted to a daily bag limit on single day trips and a possession limit (i.e., trip limit) on multi-day trips. When the WCPFC and IATTC reduced commercial catch limits in the north Pacific Ocean, the U.S. agreed to reduce recreational catch of PBF commensurately (20-45% reduction from the 2012-2014 average catch). In response, the Pacific Fishery Management Council adopted a reduced PBF daily bag and possession limit in 2014, which the National Marine Fisheries Service implemented in 2015 (NOAA, 2015). In conformance with the federal regulations, the California Fish and Game Commission adopted a change in its daily bag limit from 10 PBF per day to 2 PBF per person per day, with a change in the possession limit (for trips 3 days or longer) from 30 PBF to 6 PBF (NOAA, 2015).

Following the Pacific-wide reductions in PBF catch limits, there are indications that the documented recovery of PBF in the WPO may have extended across to the EPO. Recreational fishermen in southern California are catching more and larger juvenile PBF and can exploit the stock longer into the season than in the past, all of which are signs of a preliminary recovery in the EPO (Hendricks, 2017; horizonadmin, 2023; Lee H, *personal communication*). However, this has yet to be formally substantiated through a quantitative analysis. In this study, we developed an index of relative abundance for PBF in the EPO based on U.S. CPFV multi-species recreational fishery logbook data and satellite derived environmental data using a delta-GAM modeling framework. With our findings, we aim to track the progress of the PBF recovery in the EPO and by doing so enable the means to track the recovery of PBF Pacific-wide and inform more effective management.

Materials and Methods

Data Sources

Fisheries Catch and Effort Data

To develop a CPUE index for PBF, we obtained recreational catch and effort data (1998 – 2021) from California Commercial Passenger Fishing Vessel Fleet (CPFV) logbooks. A logbook must be completed for each day (up to 24 hours) of fishing activity, and includes information on the location where fish were caught, port of landing, number of anglers, hours fished, and number of fish kept and released by species, and must be submitted monthly. CPFV logbooks contain data from vessels targeting both highly migratory species (HMS; e.g., PBF) and non-HMS (e.g., rockfishes) in the U.S. exclusive economic zone (EEZ), including state waters, and in waters off Baja California, Mexico from 1980 to the present. While CPFV logbook data collected from U.S. waters are reported by California Department of Fish and Wildlife (CDFW) fishing blocks (10 X 10 arcminutes), data collected from Mexican waters are reported by larger, overlapping fishing blocks.

Since the CPFV fleet is a multi-species fishery, it is especially challenging to discern fishing trips targeting PBF and consequently determine an accurate measure of angler effort. Furthermore, CPFV logbooks only document target species of a given trip at the family level. These complexities are evident in a considerable number of trips with small numbers of PBF caught along with other species (Stohs, 2016). While the CDFW defines PBF effort to only include trips where PBF was caught, this classification can create bias in CPUE standardization efforts by excluding trips where PBF was targeted but none were caught (Stohs, 2016). To rectify some of these challenges, we turned to ecological and expert knowledge. Fishermen and scientists who participate in or are familiar with the CPFV fleet suggest PBF angling effort to be trips ranging in duration from a half day to 14 days (Bellquist L, Hellmers E, Snodgrass O, *personal communication*). This definition likely avoids the bias mentioned above by accounting for trips that targeted PBF but had 0 PBF catch. Accordingly, we identified unique trips within the CPFV logbooks and their corresponding trip type (quarter-day, half-day, full-day, overnight, and multi-day) using operational trip details (departure dates and times, return times, and total hours spent fishing, number of anglers) to filter the CPFV logbook data to only 'PBF-targeting' trips, as defined above, for CPUE calculation and standardization.

The operations of a fishery usually vary in time and space in response to the availability and abundance of target species, environmental factors, and management. To account for some of these factors, we identified the corresponding year, month, and fishing location (CDFW fishing block) for each trip. Many trips, particularly multi-day trips, visited multiple CDFW fishing blocks or changed the number of anglers during the trip. Trips falling into these categories were split into 'trip events' to capture each unique fishing tactic performed during a given trip. We derived the nominal CPUE for each trip event by dividing PBF catch (number of PBF kept) by angler hours (number of anglers multiplied by total hours spent fishing). We cross-referenced the number of anglers reported for each trip event against reported vessel specific angler-capacities to ensure effort was not inflated by inaccurate entries of the number of anglers. Those trip events that were not consistent with vessel capacities were omitted (7 trip events).

Finally, to address the disproportionate impact of sparse catch records, we omitted trip events associated with CDFW fishing blocks that were fished by only one unique CPFV in any given year and month (therefore data were limited to trip events associated with CDFW blocks, years, and months where more than 1 CPFV fished). The resulting dataset contained a total of 17,495 unique trips comprised of 22,273 unique trip events of both positive and 0 PBF catch events.

Environmental Data

Since the availability of PBF can be influenced by the environment, we chose to include environmental variables into the standardization process to distinguish its effects on the relative abundance of PBF in the EPO from that of management efforts. We incorporated both broadscale climatic variables representing indices of the El-Nino Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) and finescale, spatially explicit variables representing sea surface temperature (SST; degrees Celsius), sea surface height (SSH; m), and sea surface chlorophyll *a* (CHLA; mg/m^3) measurements, into the GAM modeling framework. These variables have been shown to affect PBF physiology (Blank et al., 2004; Kitagawa, 2013) and distribution (Boustany et al., 2010; Hahlbeck et al., 2017; Runcie et al., 2019). For broadscale, climatic variables, we obtained monthly time series of ENSO (NOAA PSL 2023) and PDO (NOAA NCEI 2023) indices. For fine-scale, spatially explicit variables, we derived monthly time series of SST, SSH, and CHLA measurements by geographic location corresponding to the centroid of the CDFW fishing blocks from corresponding satellite SST (NASA Ocean Biology Processing Group, 2017; Saha et al., 2018), SSH (Lopez, 2018), and CHLA data (NASA Ocean Biology Processing Group, 2017, 2018). We matched monthly timeseries for all environmental variables spanning the temporal period of the fisheries data to the corresponding CPFV trip event by month, year, and location (for fine-scale variables).

Modeling Framework: delta-GAMs

We developed a GAM framework for standardizing CPUE of PBF based on CPFV logbook catch and effort data and overlapping environmental variables within the EPO from 1998 to 2021. Owing to the large number of zero catches of PBF within the CPFV logbook data, we employed a delta-GAM framework. The delta approach models the probability of presence using a binomial GAM, and the non-zero catches using a gamma GAM. The product of these two results represents a standardized index of PBF CPUE. Given the respective error distributions, we used a logit link function in the binomial GAM and a log link function in the gamma GAM.

We fit both GAMs using the 'mgcv' package (v1.8-39; Wood 2023) in R (4.1.3) and tested six model structures ranging in complexity (Supplemental information; Table 1.1). We treated the variables of year and month as fixed effects smoothed with a thin plate spline and a cyclic cubic regression spline (useful for smoothing over variables that vary over a cycle), respectively. We treated vessel as a random effect, and location was derived from the centroid of the CDFW fishing blocks smoothed with a Duchon spline (useful for spatial). We treated all broadscale, climatic and fine-scale, spatially explicit variables as fixed effects smoothed with a thin plate regression spline.

We used Akaike Information Criterion (AIC) to select the best performing models (Table 1.1) and thus the set of variables that best explain the observed variability in probability of presence and relative abundance of PBF in the EPO. When models performed similarly (within 2 AIC values) we used ensemble modeling to predict PBF CPUE. In this instance the predicted probability of presence represents an average across the models, weighted by their AIC weights. While it is best practice to validate models with datasets independent to their development (e.g., cross validation with test and train datasets), we were limited by the modeling framework and consequently dataset size. Noting the number of coefficients in a GAM cannot surpass the number of data points, which would be the case for the gamma GAM dataset (which includes only trips with non-zero catches of PBF), we chose to validate the best performing binomial and gamma GAMs using a bootstrap procedure. We produced 100 sets of randomly simulated responses from the binomial and gamma GAMs separately, which were then multiplied to generate 100 sets of randomly simulated PBF CPUE predictions. Spearman's correlation coefficient (Spearman's ρ) was used to validate between the observed and each set of simulated CPUE predictions. We derived a mean and 95% quantile range (QR) of Spearmean's ρ across simulations.

In effort to distinguish variability in PBF CPUE attributable to the changes in the environment from variability attributable to changes in management, we predicted PBF relative abundance in space and time with and without the effects of the environment. To predict PBF relative abundance with the influence of the environment, environmental factors we allowed to vary whereas when predicting PBF relative abundance without the effect of the environment we kept the environmental factors included in the model constant at their respective mean measurements.

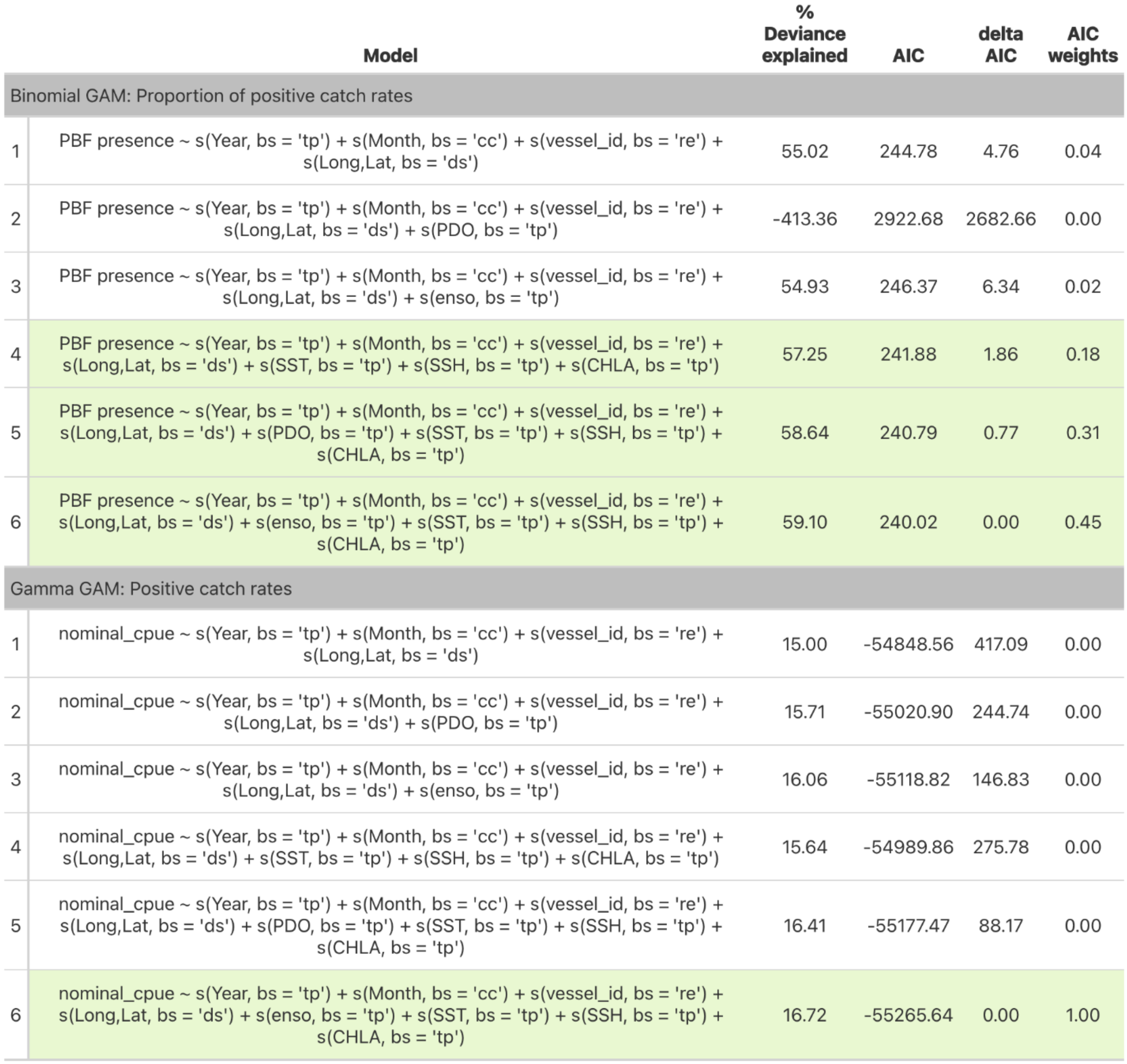
To visualize the results of these two prediction frameworks (with and without the effects of the environment), we aggregated the predicted delta-GAM outputs (product of best-performing binomial and gamma GAMs) of 100 simulations by year to develop an annual timeseries of standardized PBF CPUE and by CDFW fishing block and year to develop annual maps of standardized CPUE for PBF in the EPO. Annual standardized CPUE maps and time series are represented by median PBF CPUE and associated 90% confidence interval (CI) of CPUE. Lastly, we developed response curves for each predictor variable incorporated into the binomial and gamma GAM using the ‘mgcViz’ package (v 0.1.0; Fasiolo et al., 2021) to understand how the environment influences variability in PBF relative abundance. Response curves display how each variable included in the model (excluding the random effect ‘vessel’) affects the prediction when all other variables are held constant (also known as marginal effects).

Results

Model selection, evaluation, and validation

Model selection indicated the best performing binomial GAMs to be model 4, 5, and 6 (explaining an average of 58.33% of deviance of the presence/absence data across the three models) (Table 1.1). The best performing gamma GAM was model 6 (explaining 16.72% of deviance of the non-zero abundance data). Comparing the standardized CPUE values predicted by the delta GAM to the nominal CPUE values resulted in a positive and highly significant Spearman’s correlation coefficient (mean ρ=0.154 with 100 simulations; 95% QR: 0.021; $p<0.0001$), indicating the delta GAM can reliably be used to standardize PBF CPUE in time and space.

Table 1.1: Model description and selection of binomial GAMs that model proportion of positive catch rates and gamma GAMs that model positive catch rates. Model descriptions, % deviance explained, AIC, delta AIC, and model weights are provided for each binomial and gamma GAM tested.



	Model	% Deviance explained	AIC	delta AIC	AIC weights
Binomial GAM: Proportion of positive catch rates					
1	PBF presence ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds')	55.02	244.78	4.76	0.04
2	PBF presence ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(PDO, bs = 'tp')	-413.36	2922.68	2682.66	0.00
3	PBF presence ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(enso, bs = 'tp')	54.93	246.37	6.34	0.02
4	PBF presence ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(SST, bs = 'tp') + s(SSH, bs = 'tp') + s(CHLA, bs = 'tp')	57.25	241.88	1.86	0.18
5	PBF presence ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(PDO, bs = 'tp') + s(SST, bs = 'tp') + s(SSH, bs = 'tp') + s(CHLA, bs = 'tp')	58.64	240.79	0.77	0.31
6	PBF presence ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(enso, bs = 'tp') + s(SST, bs = 'tp') + s(SSH, bs = 'tp') + s(CHLA, bs = 'tp')	59.10	240.02	0.00	0.45
Gamma GAM: Positive catch rates					
1	nominal_cpue ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds')	15.00	-54848.56	417.09	0.00
2	nominal_cpue ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(PDO, bs = 'tp')	15.71	-55020.90	244.74	0.00
3	nominal_cpue ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(enso, bs = 'tp')	16.06	-55118.82	146.83	0.00
4	nominal_cpue ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(SST, bs = 'tp') + s(SSH, bs = 'tp') + s(CHLA, bs = 'tp')	15.64	-54989.86	275.78	0.00
5	nominal_cpue ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(PDO, bs = 'tp') + s(SST, bs = 'tp') + s(SSH, bs = 'tp') + s(CHLA, bs = 'tp')	16.41	-55177.47	88.17	0.00
6	nominal_cpue ~ s(Year, bs = 'tp') + s(Month, bs = 'cc') + s(vessel_id, bs = 're') + s(Long,Lat, bs = 'ds') + s(enso, bs = 'tp') + s(SST, bs = 'tp') + s(SSH, bs = 'tp') + s(CHLA, bs = 'tp')	16.72	-55265.64	0.00	1.00

Spatio-temporal trends of standardized PBF CPUE

There were minimal differences in the standardized CPUE timeseries with and without the effect of the environment (Figure 1.1a). PBF CPUE generally declined from 1998 to 2006, increased to its highest value in 2011, declined from 2011 to 2018, and thereafter increasing to the end of the timeseries in 2021. Refer to the supplemental (Supplemental Figures 1.1-1.16) for spatio-temporal predictions of PBF CPUE and their associated 90% CIs with and without the effect of the environment for all years.

For most of the modeling duration, relative abundance of PBF was highest offshore U.S. and Mexico waters. Over time, there was a noticeable increase in the spatial footprint of PBF within the EPO (Supplemental Figure 1.17). When PBF CPUE was both at its lowest (2006) and highest (2011) levels, the spatial footprint of PBF (number of unique CDFW fishing blocks with PBF CPUE) was similar (2006: 9; 2011: 7 CDFW blocks) (Figure 1.1b and c). In recent years, when PBF CPUE increased (2018 – 2021), the spatial footprint expanded into waters surrounding the southern Channel Islands (Figure 1.1d; 22 CDFW blocks) and nearshore waters off southern California (Figure 1.1e; 62 CDFW blocks).

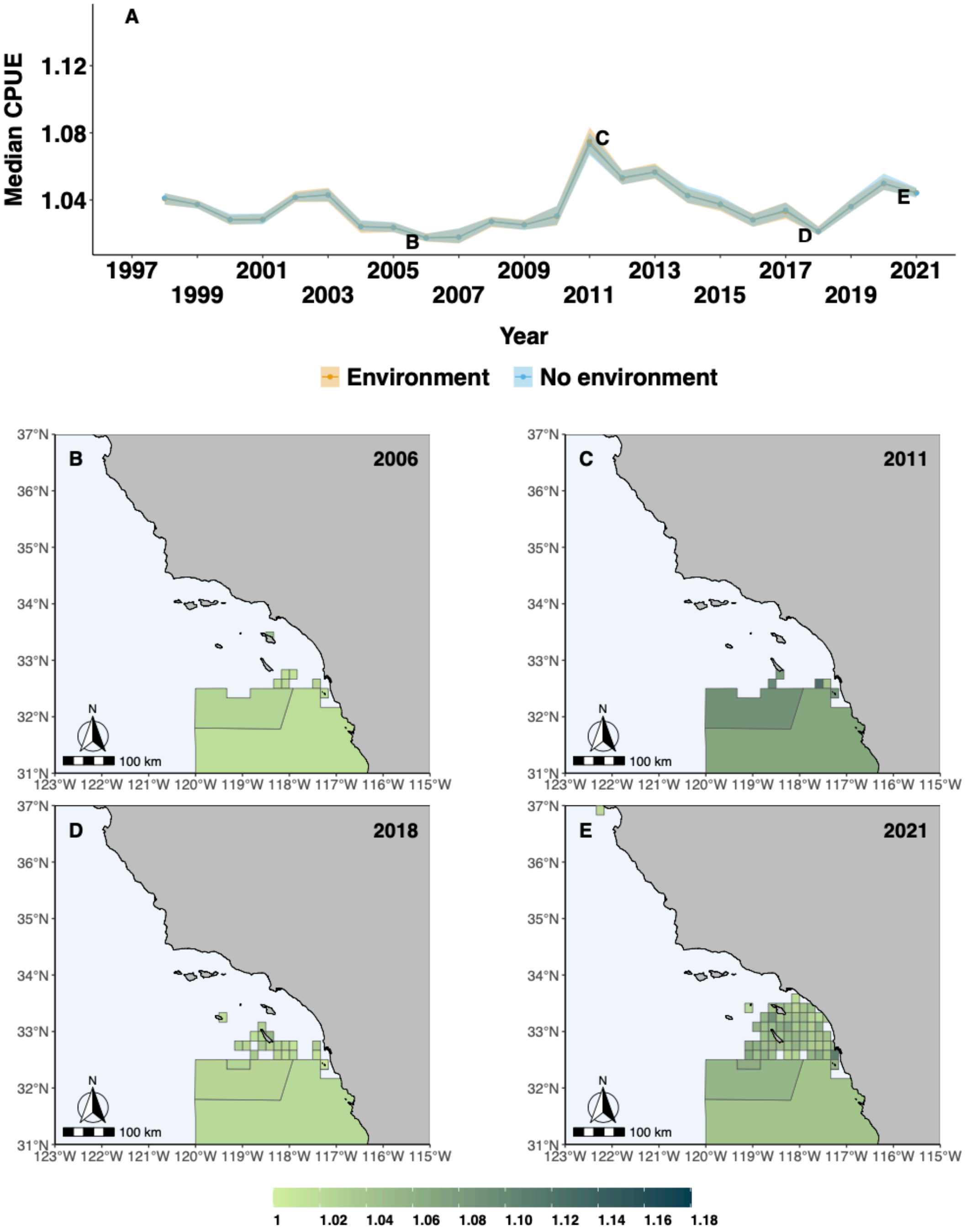


Figure 1.1: Median PBF CPUE from 1997 to 2021 with (orange) and without (blue) the effect of the environment. The timeseries of PBF CPUE (A) is represented by the median CPUE (black line) and 90% CI of model outputs (grey shaded area). Median PBF CPUE by CDFW fishing block throughout southern Californian waters in 2006 (B), 2011 (C), 2018 (D) and 2021 (E) without the effect of the environment. Darker colors correspond to higher CPUE.

Influence of environmental variables on PBF CPUE

The response curves of the environmental variables included in the binomial GAMs exhibited a substantial amount of variability (Figure 1.2a-e). However, the probability of PBF presence tends to be highest in warmer SST (> 20°C), lower SSH (<0.63 m), higher CHLA (2-5 mg/m^3), and slightly negative ENSO (-1.5-0) and PDO indices (-2 to 0). The response curves of the environmental variables included in the gamma GAM indicated PBF CPUE was highest in a wide range of SST (17.10° to 22.57° C) and SSH (0.52 to 0.73 m) (Figure 1.2f-i). Higher PBF CPUE also occurred in multiple ranges of CHLA (0.75 to 1.60 mg/m^3 and 3.15 to 5.50 mg/m^3) and ENSO indices (-2.10 to -0.70, -0.13 to 0.76, and 1.04 to 2.21).

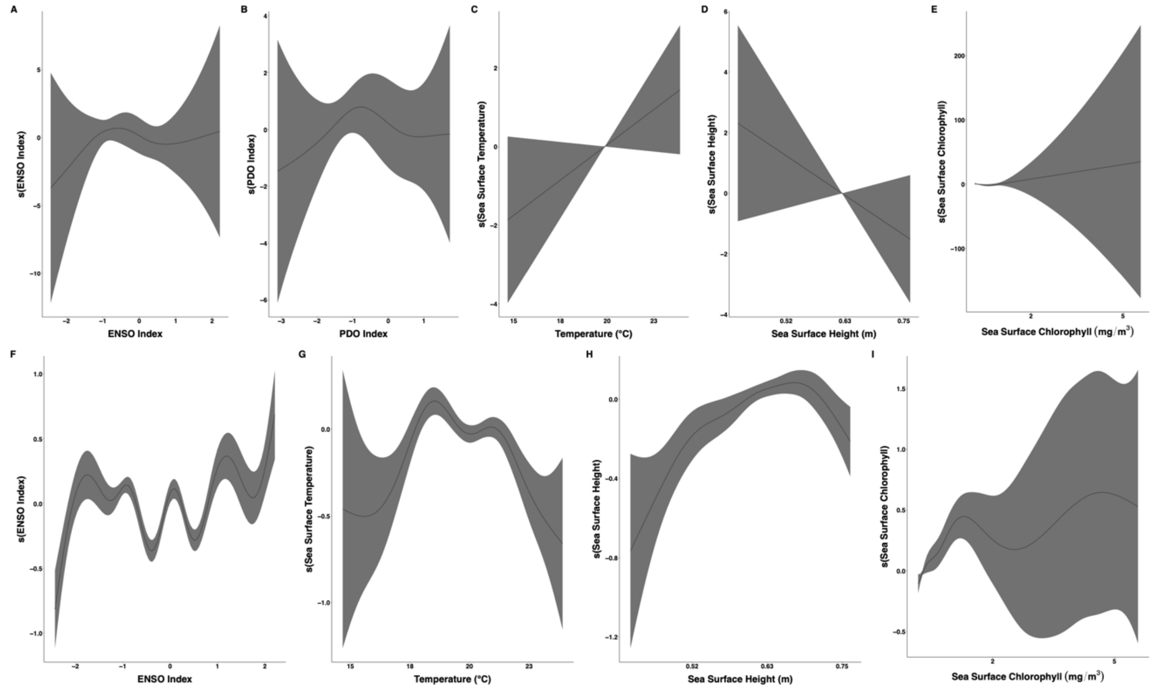


Figure 1.2: Response curves for environmental variables included in the binomial (A-E) and gamma (F-I) GAMs used to standardize PBF CPUE when present. The response curves are developed using the ‘mgcViz’ package in R. The black line represents the mean response, and the grey shaded region represents two SEs of the mean.

Discussion

The recovery in PBF in the WPO is evident in the 2022 ISC stock assessment and harvest strategy projections, however these benefits have yet to be investigated in the EPO. We developed an index of relative abundance for PBF in the EPO based on U.S. CPFV multi-species recreational fishery using a delta-GAM modeling framework. Our findings indicate early signs of recovery in the EPO that can be attributable to favorable environmental conditions and ongoing management efforts.

While there was little to no difference in the PBF CPUE timeseries with and without the influence of the environment, PBF CPUE shows an increasing trend until 2018, well after initial international (2011) and state (2015) management changes. The delayed signal of the recovery reaching the EPO may be due to the inherent lag in population dynamics resulting from the trans-pacific migration of PBF. Additionally, as a multispecies recreational fishery, the U.S. fishery has relatively low fishing intensity compared to fisheries targeting PBF in the WPO and consequently may explain the minimal recovery of PBF observed in the EPO. Despite the subtle indications of PBF recovering in the EPO, various lines of evidence substantiate the recovery in the EPO, including a spatial expansion, extension of the fishing season, increases in the ‘catch to catch limit’ ratio and number of fish released per trip, increases in size of PBF, and recruits from the WPO reaching the EPO. For many years, PBF CPUE was concentrated in offshore U.S. and Mexican waters, even in 2011 when PBF CPUE peaked in the EPO (Figure 1.1b and c). However, in recent years, the spatial footprint of PBF expanded throughout the southern California bight (Supplemental Figure 1.17) as far north as San Pedro and into coastal waters (Figure 1.1d and e). While not explicitly captured in this analysis, PBF have been observed north of San Pedro, in waters off of northern California (Moss Landing and Santa Cruz) as well as in Washington and Alaska (personal communication with Elizabeth Hellmers). Captains of CPFV vessels continue to report large schools of PBF in offshore waters typically targeted by the fleet, and now report large schools along the coast, some within just a few miles offshore of southern California ports (Hendricks, 2017; Horizonadmin, 2023). While the PBF fishing season tends to peak during summer and fall in southern California, the fishing season has recently extended, with significant PBF catches occurring earlier (e.g., April) and later (e.g., November) in the year than was previously considered normal. Similar observations have been made in the commercial PBF fisheries operating in the EPO. Mexico's commercial fishery, which historically spanned a few months, has been meeting their annual PBF catch limit sooner into their fishing season than in the past. In fact, in 2021 the Mexican commercial fishery reached their catch limit in less than two weeks (personal communication with Dreyfus, M; IATTC 2021). Additionally, the annual median catch to catch limit of PBF and number of PBF released per trip in the CPFV fishery has increased (Supplemental Figure 1.18).

In recent years, CPFV anglers landed larger-sized PBF. The size of trophy (i.e., largest) PBF caught in the CPFV fleet has been documented in the "Whoppers of the Week" section of the Western Outdoor News, a California fishing and hunting newspaper, since 1953 (Bellquist et al., 2016). Bellquist et al. (2016) noted PBF exhibited a long-term increase in mean trophy size (weight in kg) through time. This upward trend continued to persist when the timeseries was extended, with a particularly steep increase in more recent years (2010-2019), which cannot be explained by growth alone (Supplemental Figure 1.19). Additionally, length frequency data collected by the Sportfishing Association of California (SAC) indicated median size of PBF generally increased since 2016 (Supplemental Figure 1.20) and pulses of recruits from the WPO are now apparent in the EPO, especially in 2017 and 2019, which is promising evidence of the recovery.

The environment has a notable influence on fish stock productivity by changing habitat suitability in the ecosystem, predator-prey encounter rates, and population dynamics (e.g., growth, stock-recruitment relationships, etc.) (Farley et al., 2016; Hare et al., 2010; Karp et al., 2019). We were unable to distinguish variability in PBF CPUE attributable to the changes in the environment from variability attributable to changes in management. This is most likely due to data limitations. The response curves for each of the environmental variables included in the binomial and gamma GAMs reveal negligible ranges in environmental variables, especially for SSH and CHLA. Additionally, within these ranges, response curves were matched with considerable uncertainty suggesting the data was noisy, making it challenging to discern clear signals in environmental preference. These data limitations may be an artifact of modeling PBF CPUE at the spatial resolution of CDFW fishing blocks.

Although there was considerable uncertainty in the response curves for each of the environmental variables included in the binomial and gamma GAMs, they still demonstrate associations between PBF presence and relative abundance with the environment that align with past studies. Climatic variables, such as ENSO and PDO, have notable effects on a wide range of physical and biotic processes, including temperature, stratification, winds, upwelling, and primary and secondary production, all of which have ensuing effects on marine species (Koslow et al., 2014; Tommasi et al., 2017). It is understood that positive ENSO and PDO indices are generally correlated with higher SST values, lower CHLA concentrations, moderate winds, a deeper thermocline, and a decrease in upwelling (García-Morales et al., 2017), all of which are unfavorable conditions for PBF. In our study, PBF presence was highest in slightly negative PDO and ENSO indices, yet PBF CPUE was highest in a range of negative and positive ENSO indices. Consequently, our results from the gamma GAM may be an artifact of incongruous spatial resolutions between the climatic phenomena and the spatio-temporal population dynamics of PBF modeled from the U.S. recreational fishery. Another reason for the preference for both positive and negative ENSO indices could be due to the species' broad thermal tolerance. The results from this study, as well as past tagging (Boustany et al., 2010; Domeier et al., 2005; Fujioka et al., 2018; Itoh et al., 2003; Kitagawa et al., 2006, 2007), experimental (Blank et al., 2004) and habitat studies (Hazen et al., 2013; Runcie et al., 2019) documented PBF in a range of temperatures. Even with this broad thermal tolerance, it is thought that endothermic capabilities of PBF increase with size, where smaller PBF can tolerate SST of 14 – 22 °C (Boustany et al., 2010; Fujioka et al., 2018; Kitagawa et al., 2006) and larger PBF can tolerate SSTs >25 °C (Ashida et al., 2015). Within this broad range of SST, higher CPUE in PBF was associated with SST of ~18-22 °C, which agrees with the associated preference in SST with PBF presence in the current and past (Runcie et al., 2019) studies. In view of this tolerance for such a wide range of SST, Runcie et al (2019) suggested other fine-scale environmental variables may be more influential on PBF distribution, such as prey availability.

Studies identified seasonal variability in PBF migration patterns and distribution in the EPO was tightly correlated to peaks in temperature and coastal primary productivity, which have a notable influence on forage species (Boustany et al., 2010; Domeier et al., 2005; Fujioka et al., 2015; Kitagawa et al., 2007). Key forage species in the EPO include Pacific sardine (sardine), market squid, *Doryteuthis opalescens*, Northern Anchovy (anchovy), *Engraulis mordax,* and Pacific Herring, *Clupea pallasii* (Thompson, Harvey, et al., 2019), some of which PBF rely on as prey in the EPO (Portner et al., 2022). This study supports these findings through higher CPUE of PBF during summer and fall months in areas of higher CHLA concentration (>3 mg/m^3) and SSH levels (0.5m), both of which are proxies for prey availability (Figure 2 h and i).

The EPO, particularly the California Current System (CCS), is a highly dynamic and productive marine ecosystem that has experienced major oceanographic changes in recent years that have proven favorable for PBF. In 2013, surface waters rose to record high temperatures, resulting in a large mass of warm water that occurred in the southern portion of the CCS from 2014 to 2015, known as the "Warm Blob" (Bond et al., 2015). Subsequently, the strongest El Nino event ever recorded occurred through 2016 (Jacox et al., 2016). Since then, conditions returned to near average with relatively weak El Nino conditions recurring in the winter of 2018 to 2019 (Thompson, Schroeder, et al., 2019). Increased water temperatures greatly influenced coastal upwelling in the CCS by diminishing upwelling strength. Upwelling strength recovered in 2016 after the El Nino, increased in strength in 2017 (Jiménez-Quiroz et al., 2019), and remained relatively strong through 2020 (Weber et al., 2021). These oceanographic conditions in the CCS impacted the distribution and abundance of many local marine species throughout the food web, including major prey species of PBF (Bond et al., 2015; Jacox et al., 2016; Leising et al., 2016; Weber et al., 2021). Anchovy and sardine are important forage fishes in the CCS and their dynamics are greatly influenced by upwelling conditions, and thus their population dynamics coincide with environmental variability (Baumgartner et al., 1992; Lluch-Belda et al., 1992; McClatchie et al., 2017; Weber et al., 2021). The sardine populations in the CCS has been low since 2015 (Hill et al., 2015; Weber et al., 2021), and there are no signs of recovery. The anchovy population, which was low from 2013 to 2015 (Sydeman et al., 2020), recovered in recent years, reaching a record high in 2019 (Thompson et al., 2022; Thompson, Harvey, et al., 2019; Thompson, Schroeder, et al., 2019; Weber et al., 2021). Since PBF are opportunistic feeders (Craig et al., 2017; Portner et al., 2022), the resurgence of northern anchovy in the CCS likely contributed to the initial signs of PBF recovery in the EPO.

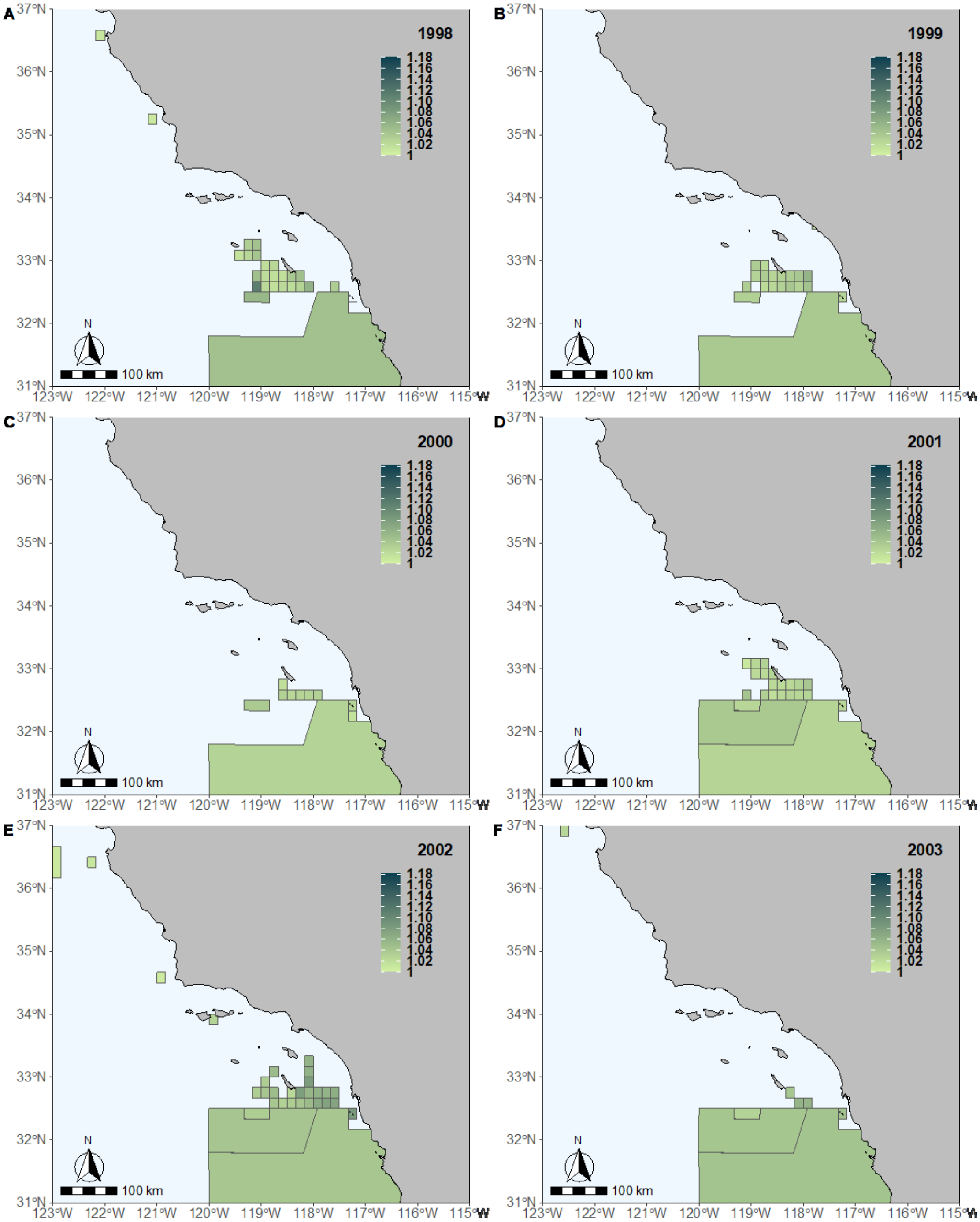
Given the consistent signs of PBF recovery in the WPO as noted in the 2022 stock assessment (ISC, 2022), the IATTC and WCPFC increased their catch limits for large (>30kg fish) fish in both ocean basins (IATTC, 2021; WCPFC, 2021). To support this decision, a suite of harvest scenarios (catch increases) was tested to identify which, if any, level of increased catch would still result in PBF reaching the second rebuilding target by 2034. Outcomes of the harvest scenarios indicated the stock is likely to meet the second rebuilding target by 2029 with probabilities > 90% under proposed increases in catch limits (ISC, 2022). While these outcomes and projections are encouraging, the stock is still considered overfished (10.2% of SSB_0) relative to the potential biomass-based reference points (20%SSB_0) adopted for other tuna species by the two RFMOs.

Owing to the preliminary recovery, Japan recently submitted a proposal requesting further increases in catch limits as many Japanese fishers are forced to release PBF to comply with current catch limits (IATTC WCPFC JWG, 2023). As this proposal could result in an increase in catch limits basin-wide, risk to the recovery of PBF should clearly be understood before progressing. Particularly noting that further changes in environmental conditions due to climate change will impact the stock dynamics, fishery interactions, and thus its progress towards recovery, key elements of the current harvest strategy are lacking (e.g., no agreed biological reference points to determine stock status) and its robustness has not been fully tested. Harvest strategy simulation testing is generally conducted using a management strategy evaluation framework which has not yet been completed. Given these circumstances, a precautionary approach to catch limits would be prudent.

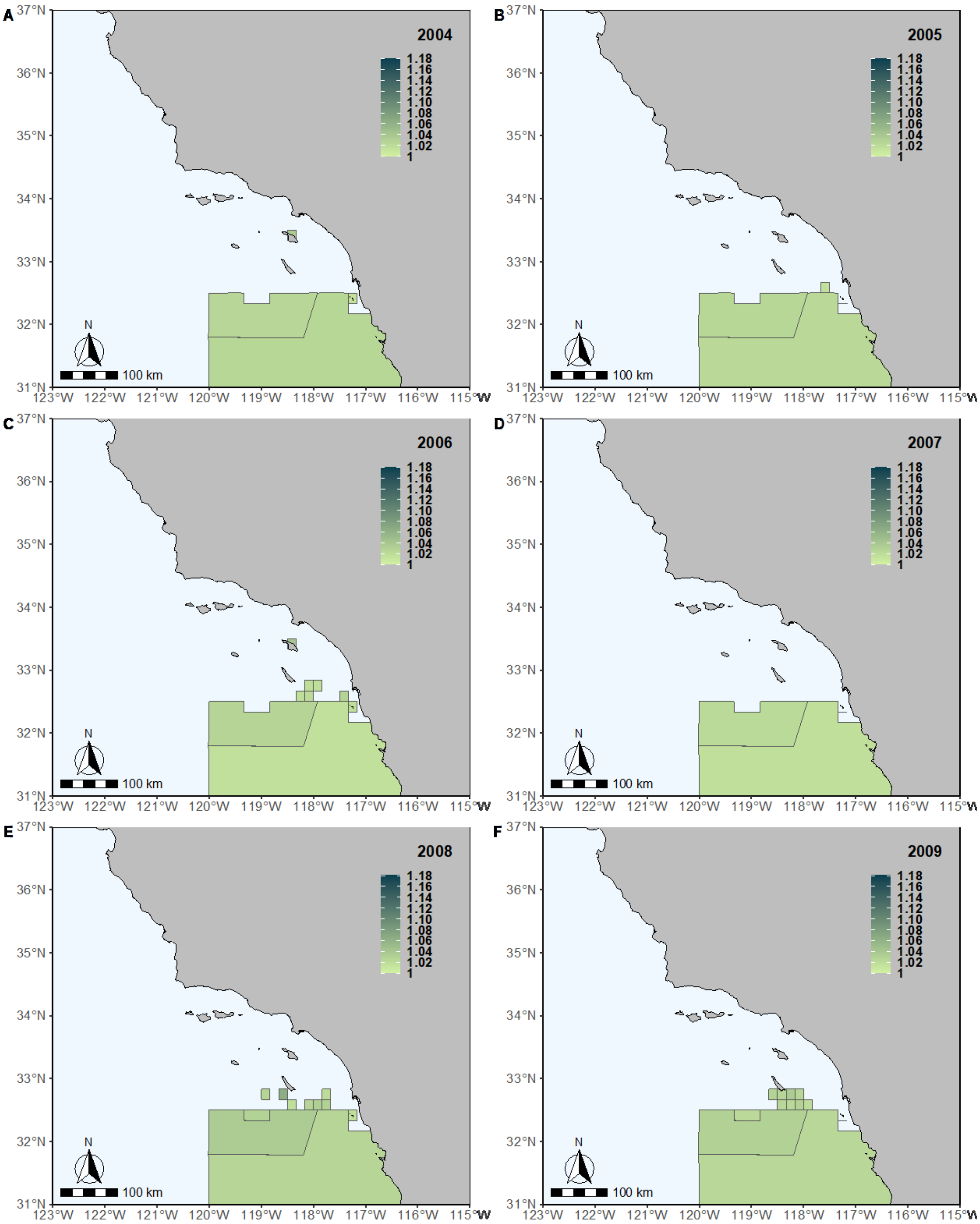
This study develops the first standardized index of abundance index for PBF from the EPO based on the U.S. recreational fishery in effort to track the recovery of PBF in the EPO. The results from this study indicate the recovery of PBF in the WPO is showing early signs in the EPO, which is contributable to favorable environmental conditions and proactive conservation measures. Collectively, this work enables a Pacific-wide assessment of PBF recovery efforts by supporting the development of stock-wide assessments and harvest strategy projections, implementation of future management measures, and investigation of climate change impacts on PBF in the EPO.

Acknowledgements

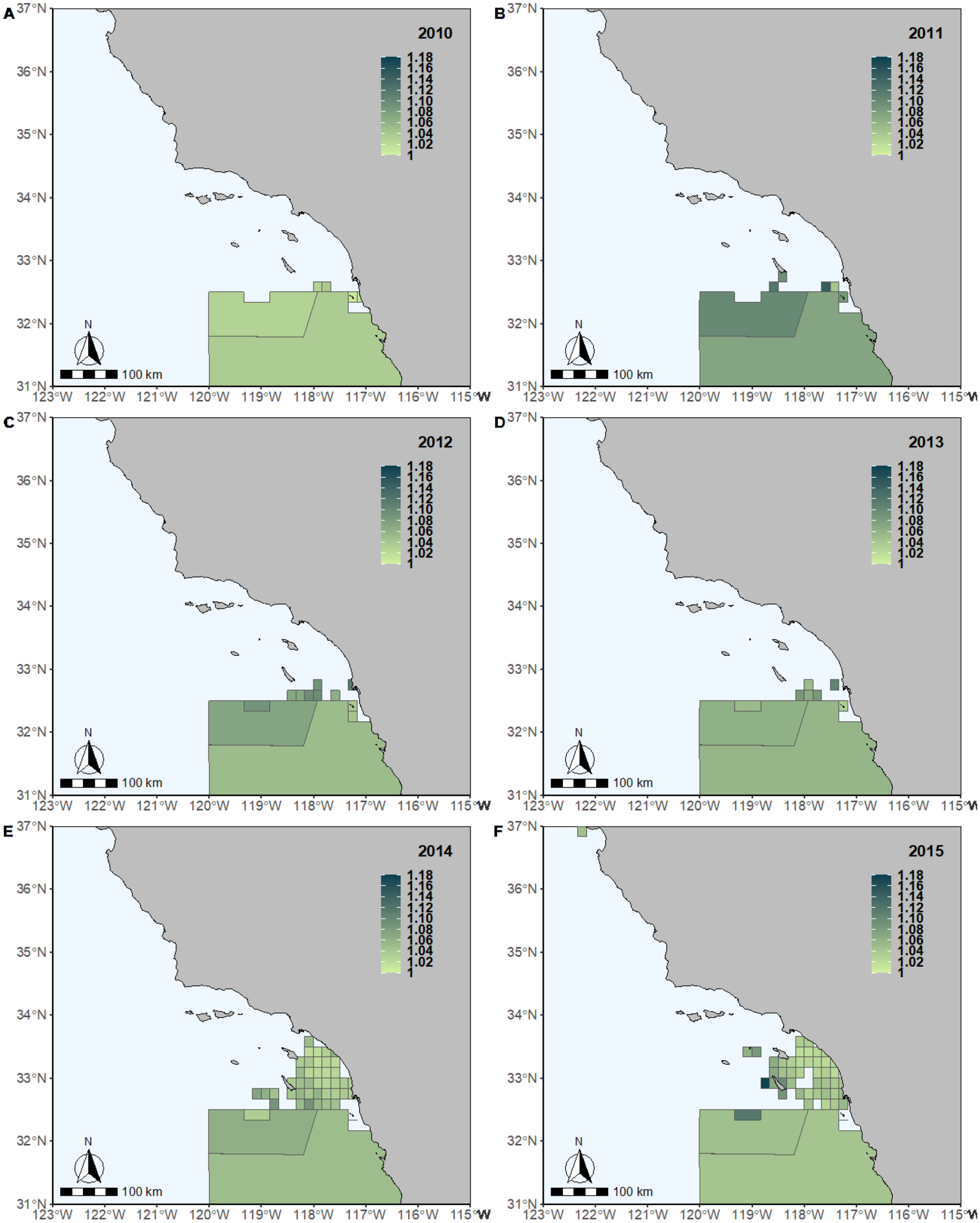
Chapter 1, in full is currently being prepared for submission for publication of the material. Coates, Julia, Semmens; Brice X. The dissertation author was the primary researcher and author of this material.



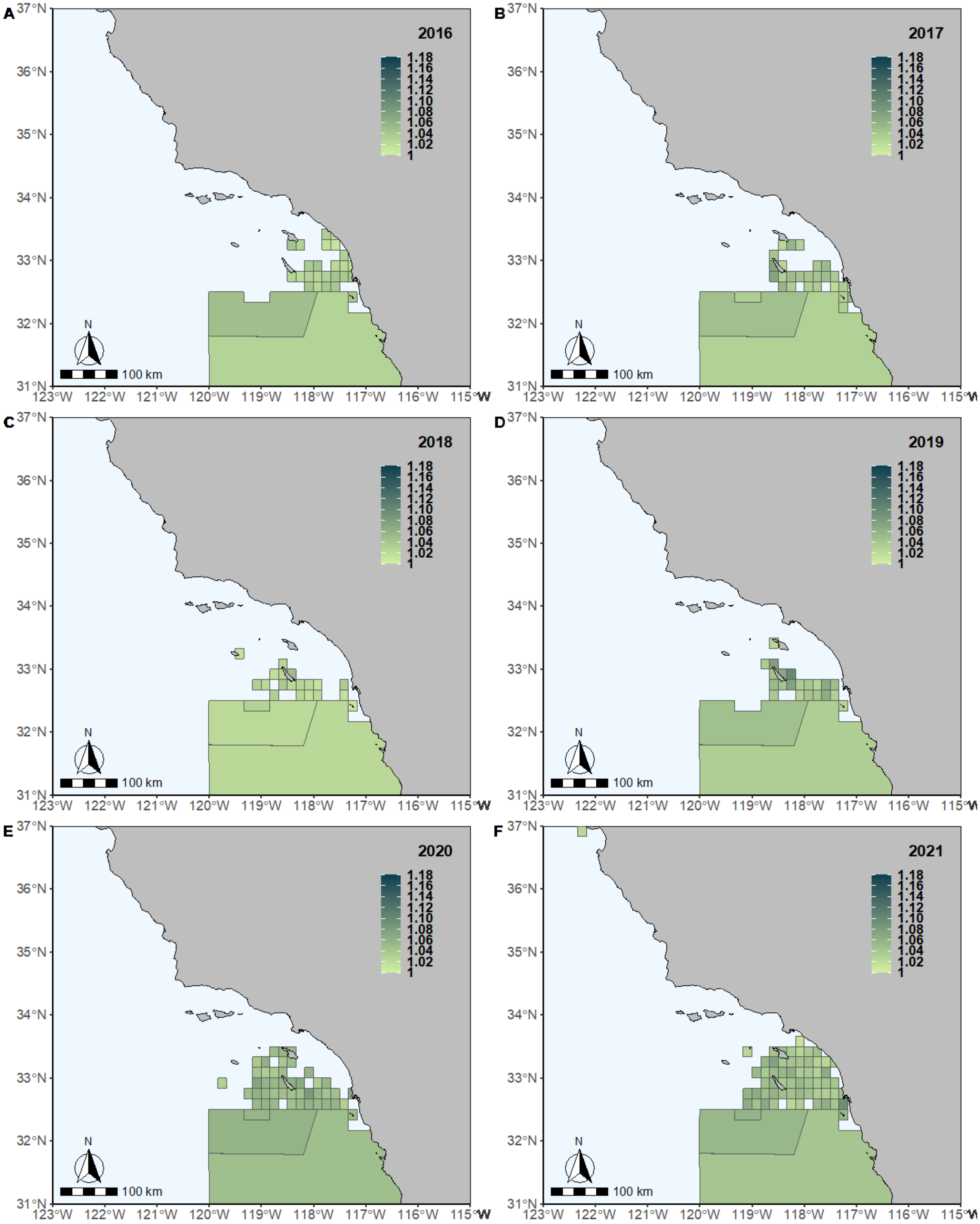
Supplemental Figure 1.1: Median PBF CPUE by CDFW fishing block throughout southern California waters in 1998 (A), 1999 (B), 2000 (C), 2001 (D), 2002 (E), and 2003 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



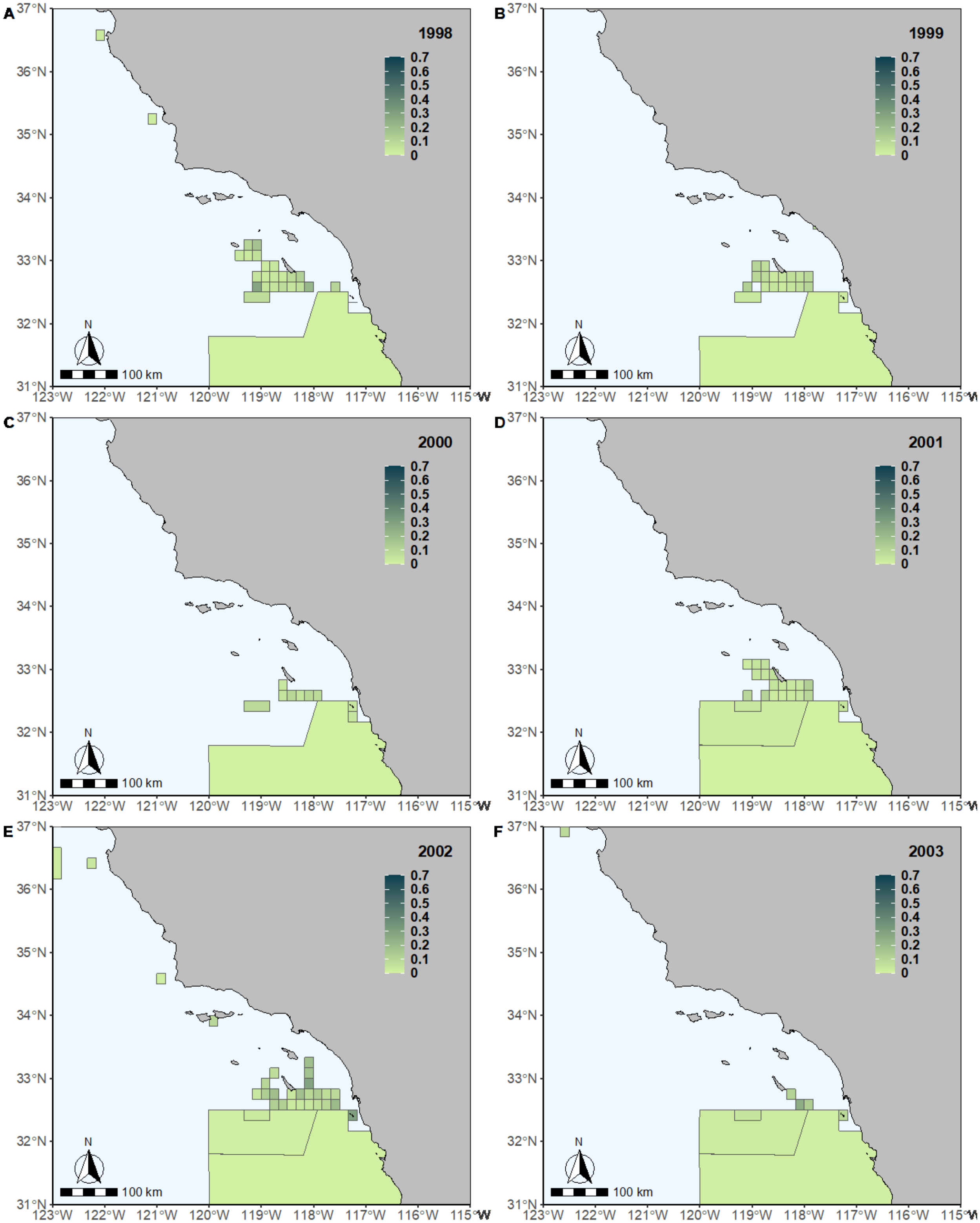
Supplemental Figure 1.2: Median PBF CPUE by CDFW fishing block throughout southern California waters in 2004 (A), 2005 (B), 2006 (C), 2007 (D), 2008 (E), and 2009 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



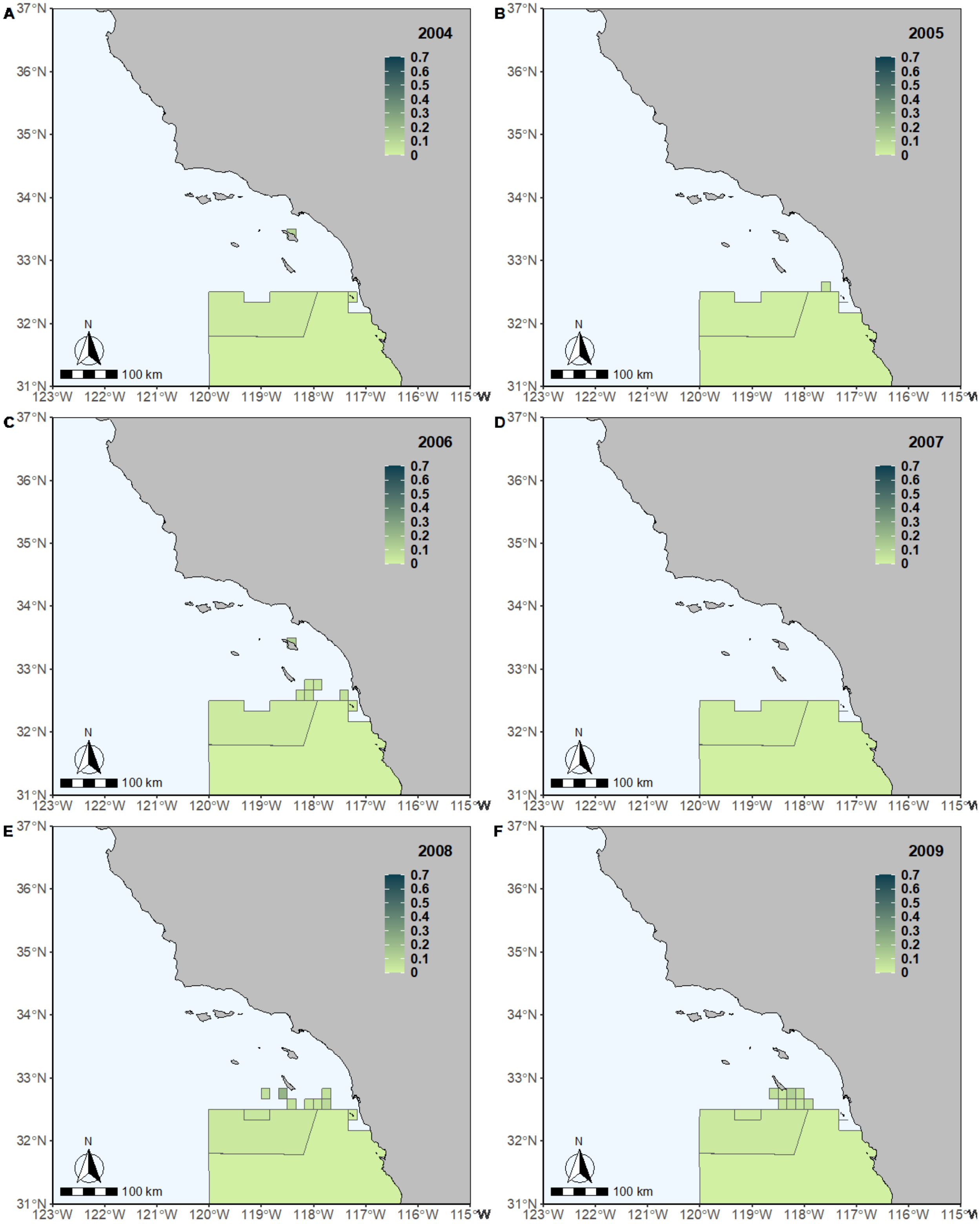
Supplemental Figure 1.3: Median PBF CPUE by CDFW fishing block throughout southern California waters in 2010 (A), 2011 (B), 2012 (C), 2013 (D), 2014 (E), and 2015 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



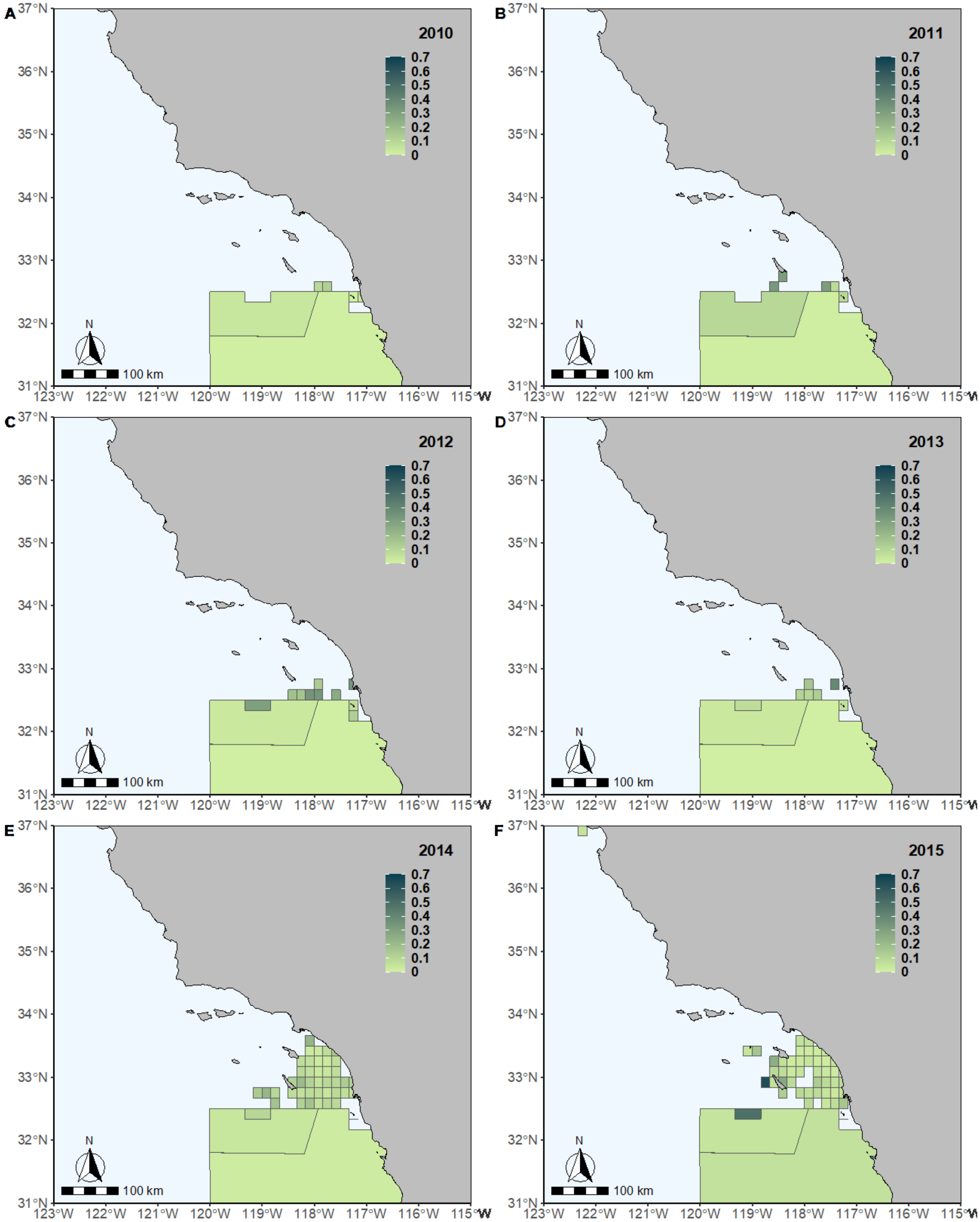
Supplemental Figure 1.4: Median PBF CPUE by CDFW fishing block throughout southern California waters in 2016 (A), 2017 (B), 2018 (C), 2019 (D), 2020 (E), and 2021 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



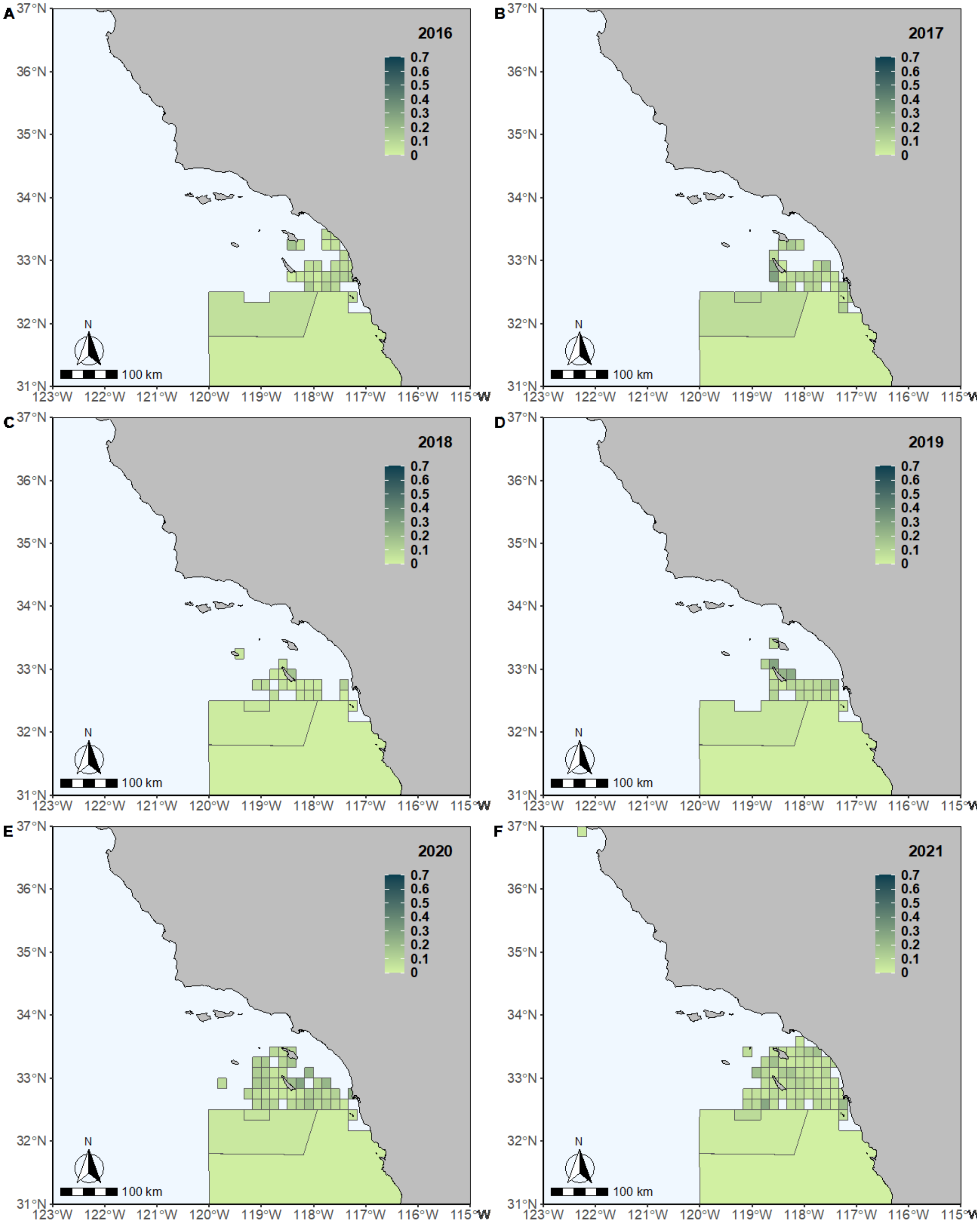
Supplemental Figure 1.5: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 1998 (A), 1999 (B), 2000 (C), 2001 (D), 2002 (E), and 2003 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



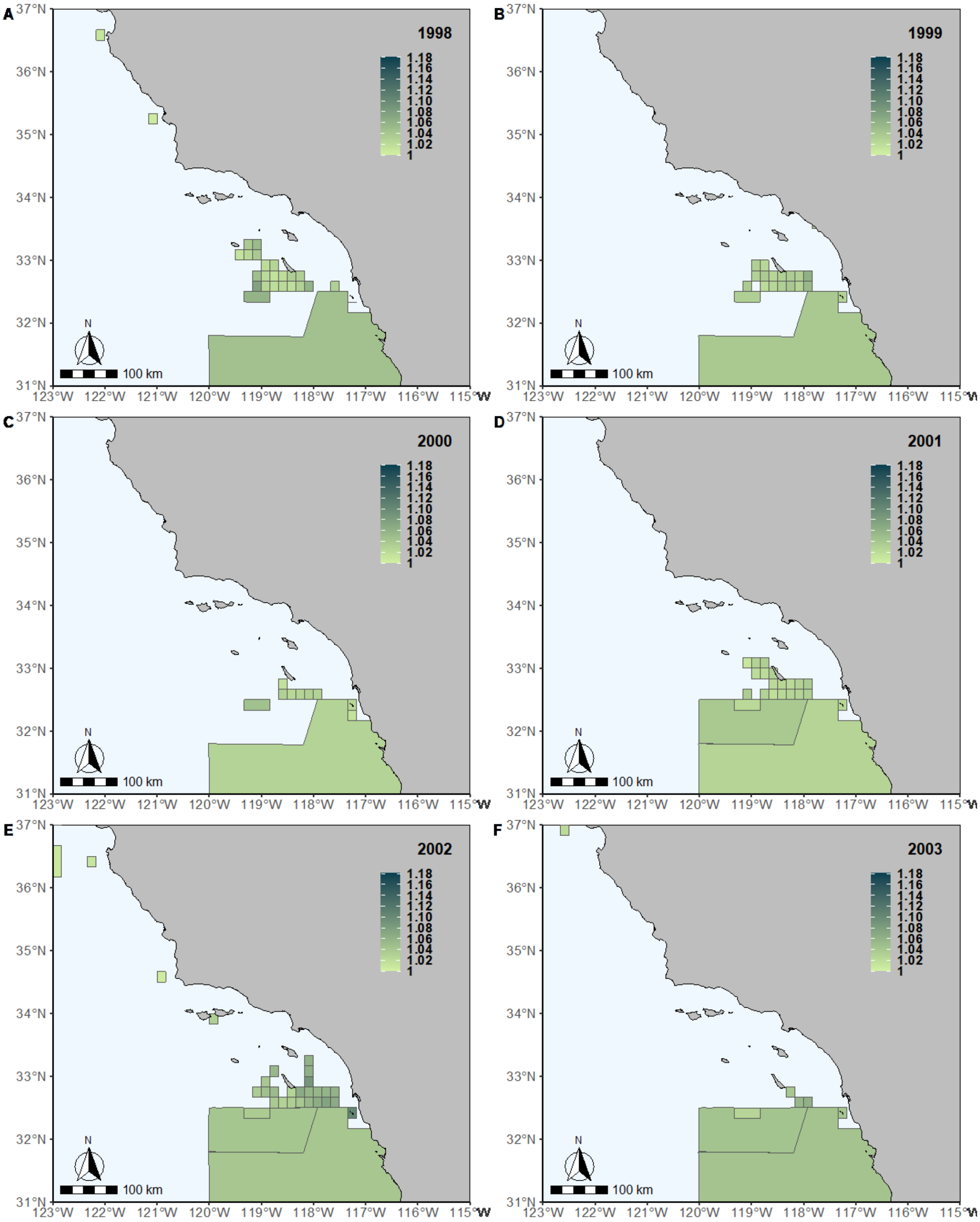
Supplemental Figure 1.6: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 2004 (A), 2005 (B), 2006 (C), 2007 (D), 2008 (E), and 2009 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



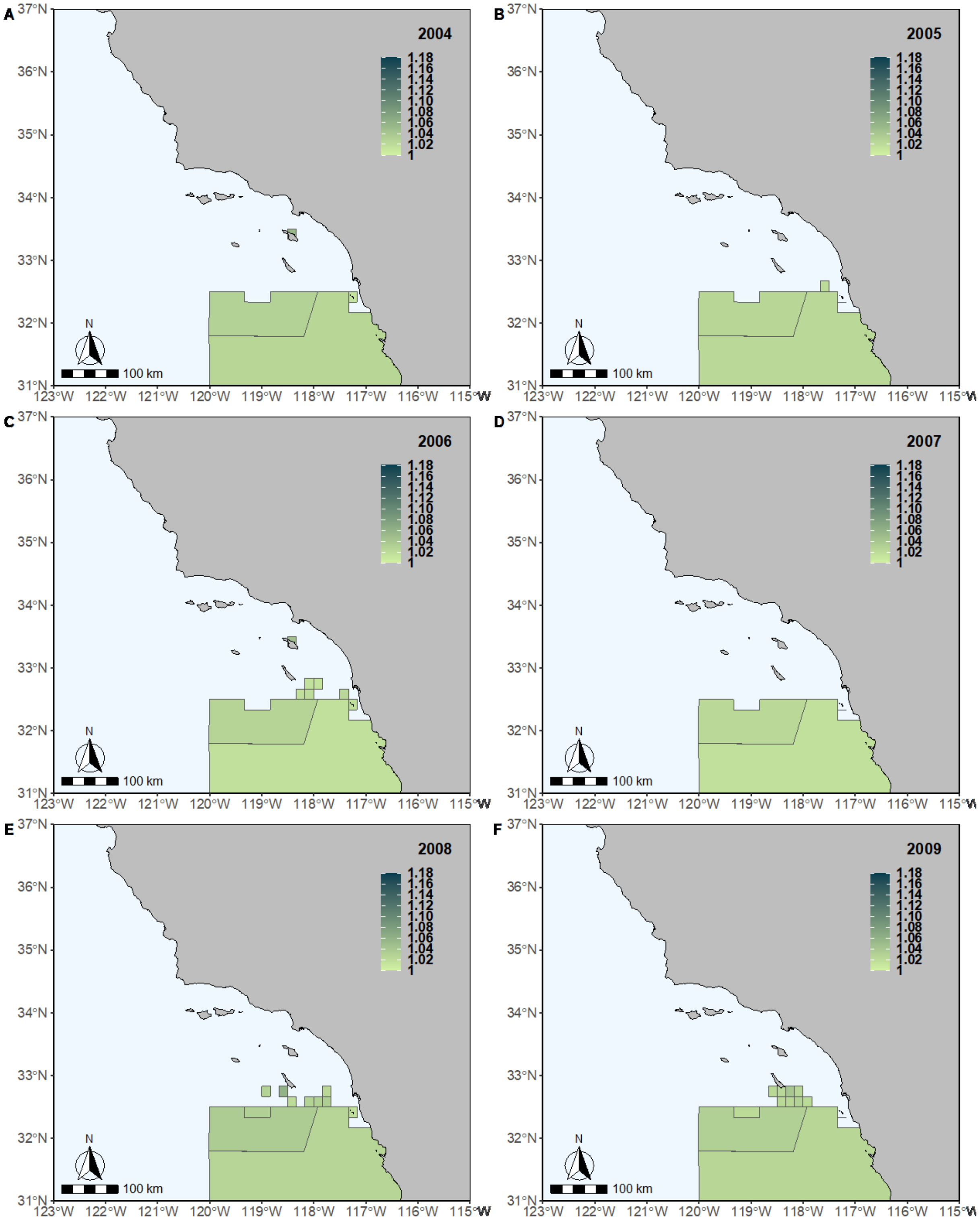
Supplemental Figure 1.7: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 2010 (A), 2011 (B), 2012 (C), 2013 (D), 2014 (E), and 2015 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



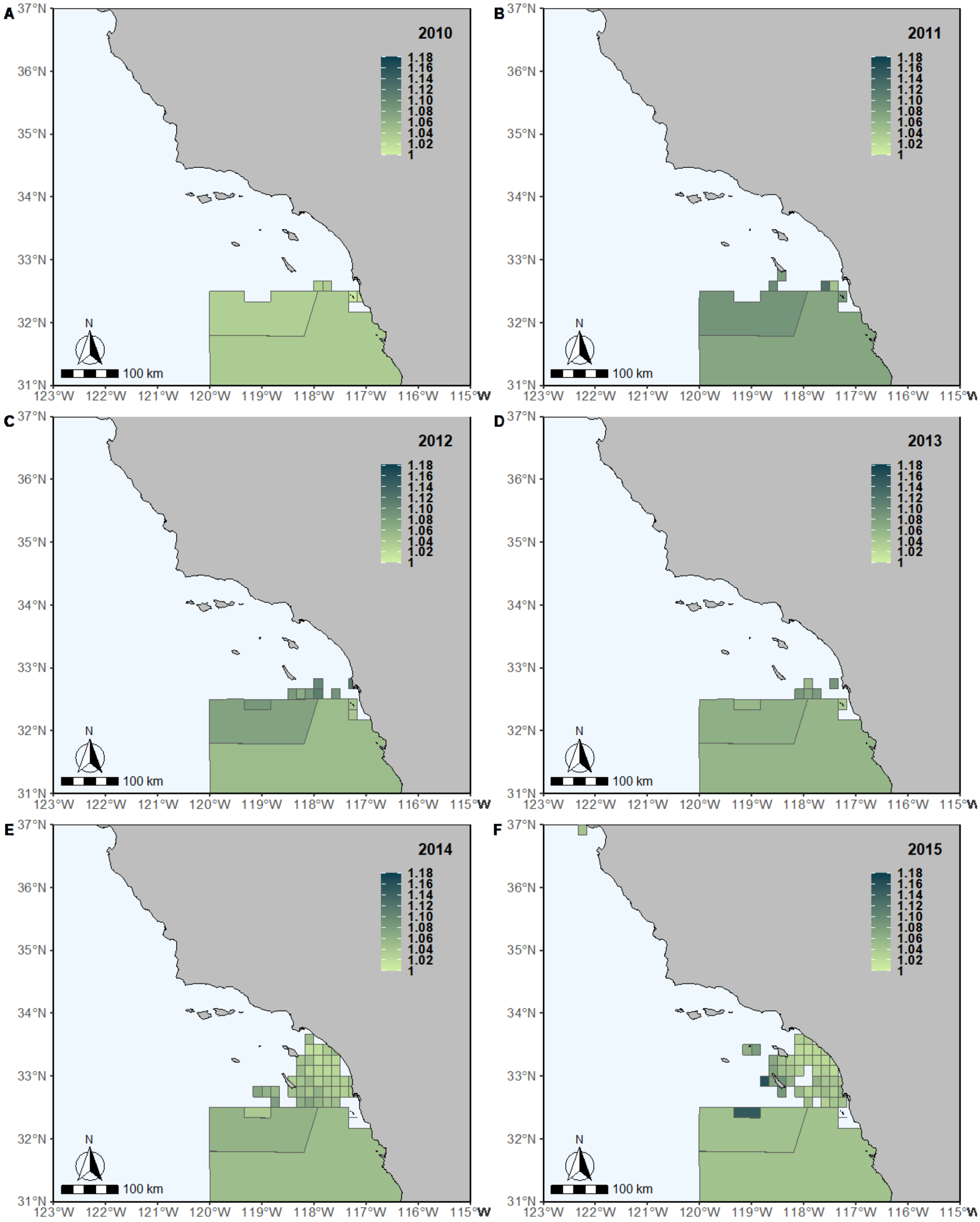
Supplemental Figure 1.8: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 2016 (A), 2017 (B), 2018 (C), 2019 (D), 2020 (E), and 2021 (F) with the effect of the environment included. Darker colors correspond to higher CPUE.



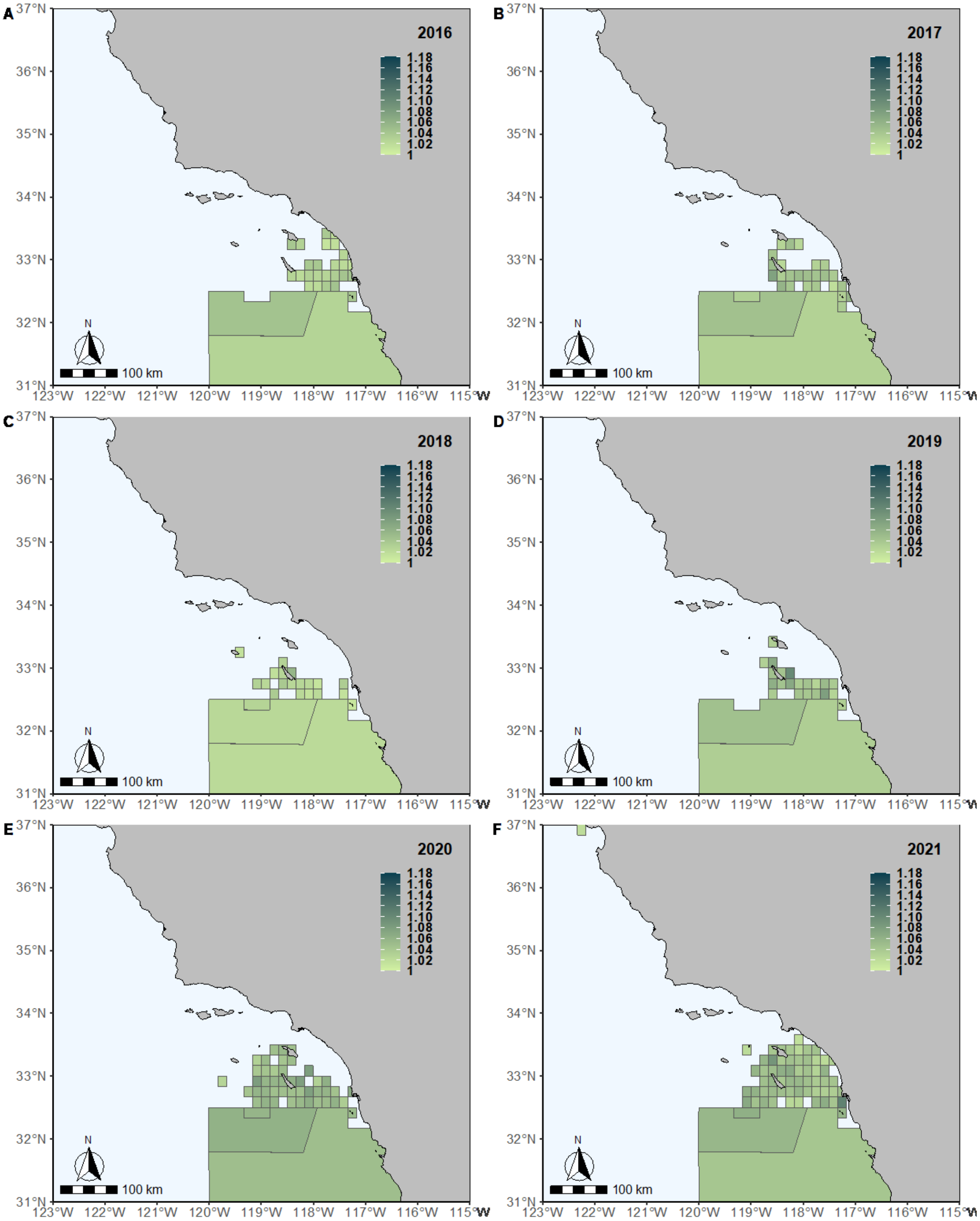
Supplemental Figure 1.9: Median PBF CPUE by CDFW fishing block throughout southern California waters in 1998 (A), 1999 (B), 2000 (C), 2001 (D), 2002 (E), and 2003 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



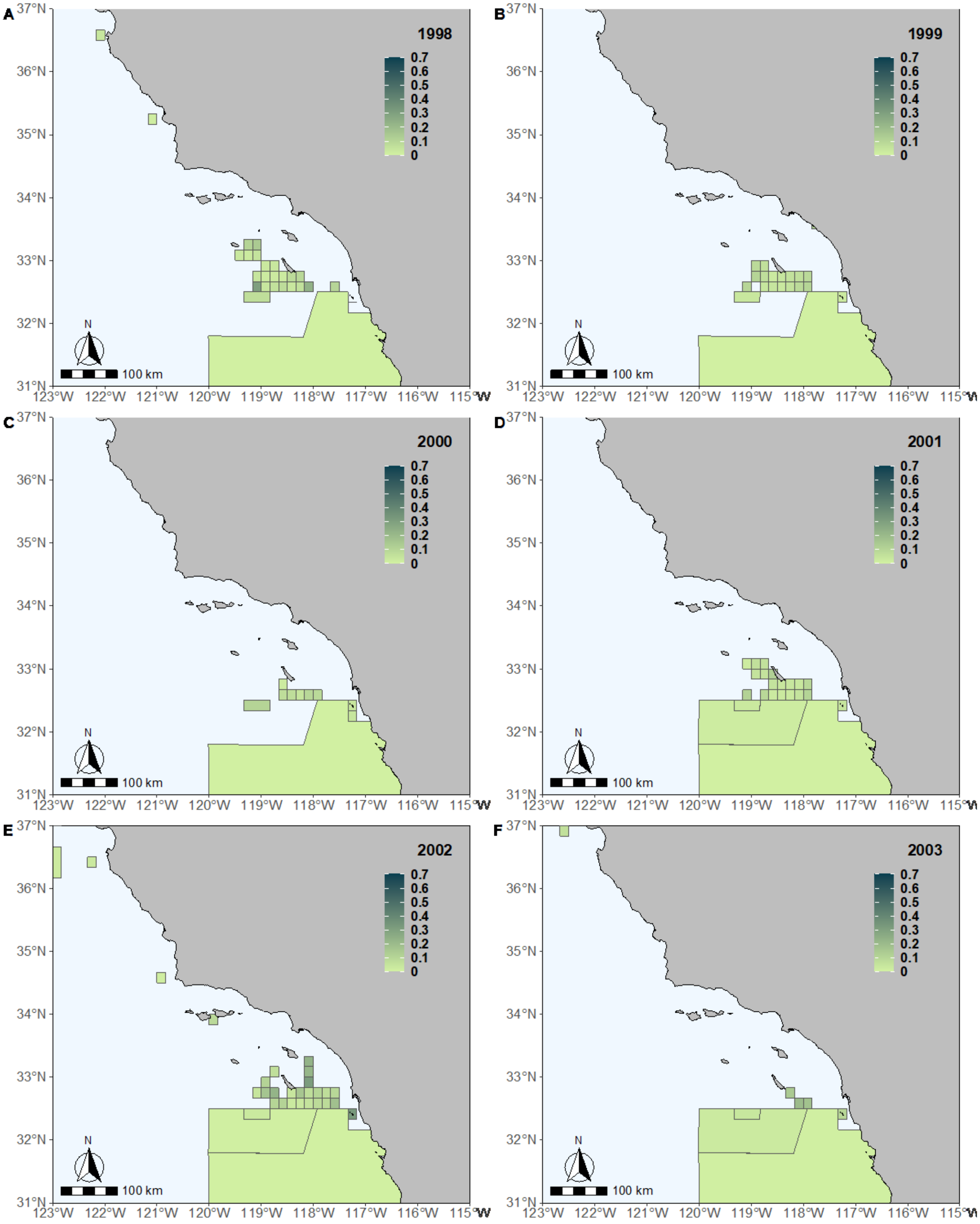
Supplemental Figure 1.10: Median PBF CPUE by CDFW fishing block throughout southern California waters in 2004 (A), 2005 (B), 2006 (C), 2007 (D), 2008 (E), and 2009 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



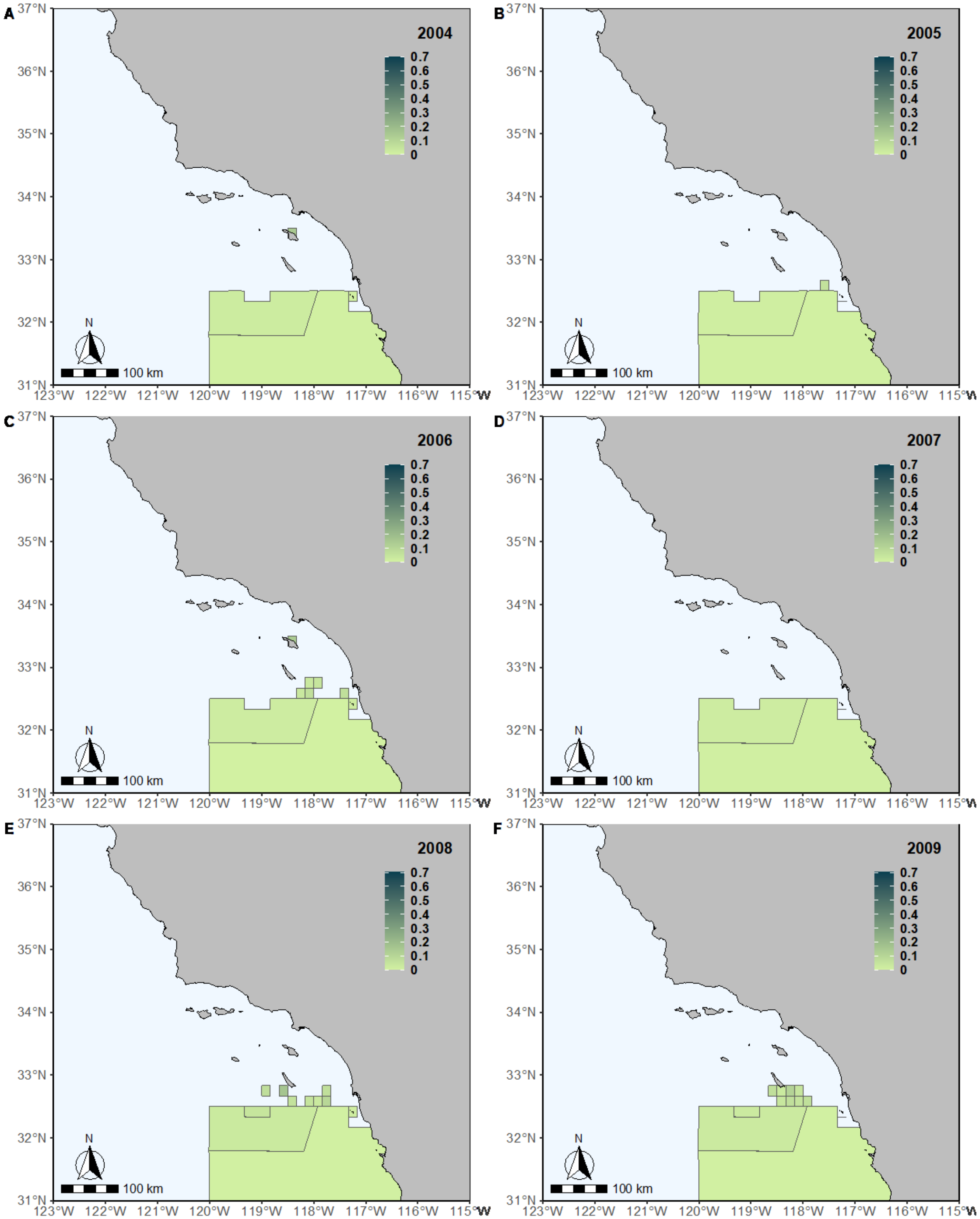
Supplemental Figure 1.11: Median PBF CPUE by CDFW fishing block throughout southern California waters in 2010 (A), 2011 (B), 2012 (C), 2013 (D), 2014 (E), and 2015 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



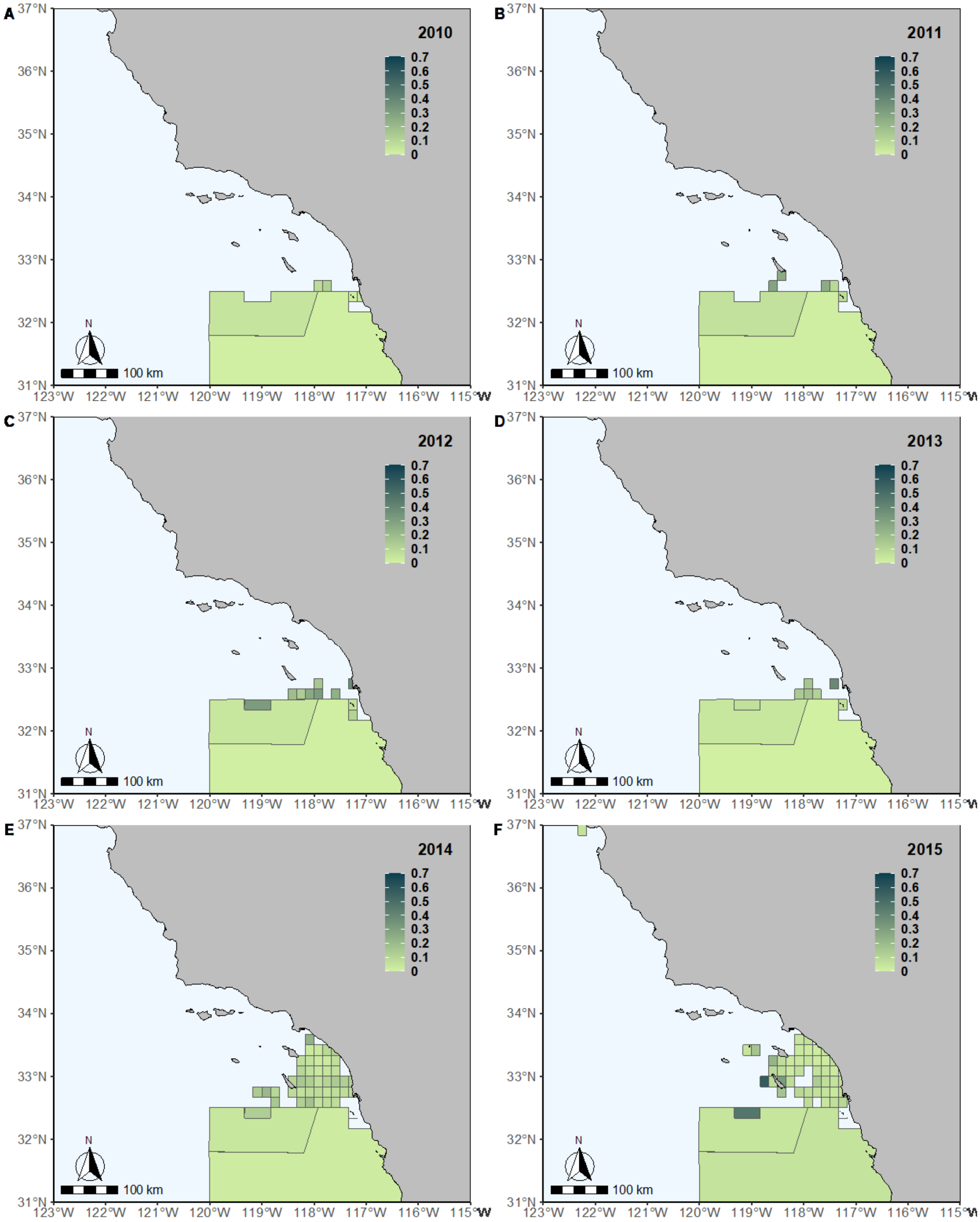
Supplemental Figure 1.12: Median PBF CPUE by CDFW fishing block throughout southern California waters in 2016 (A), 2017 (B), 2018 (C), 2019 (D), 2020 (E), and 2021 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



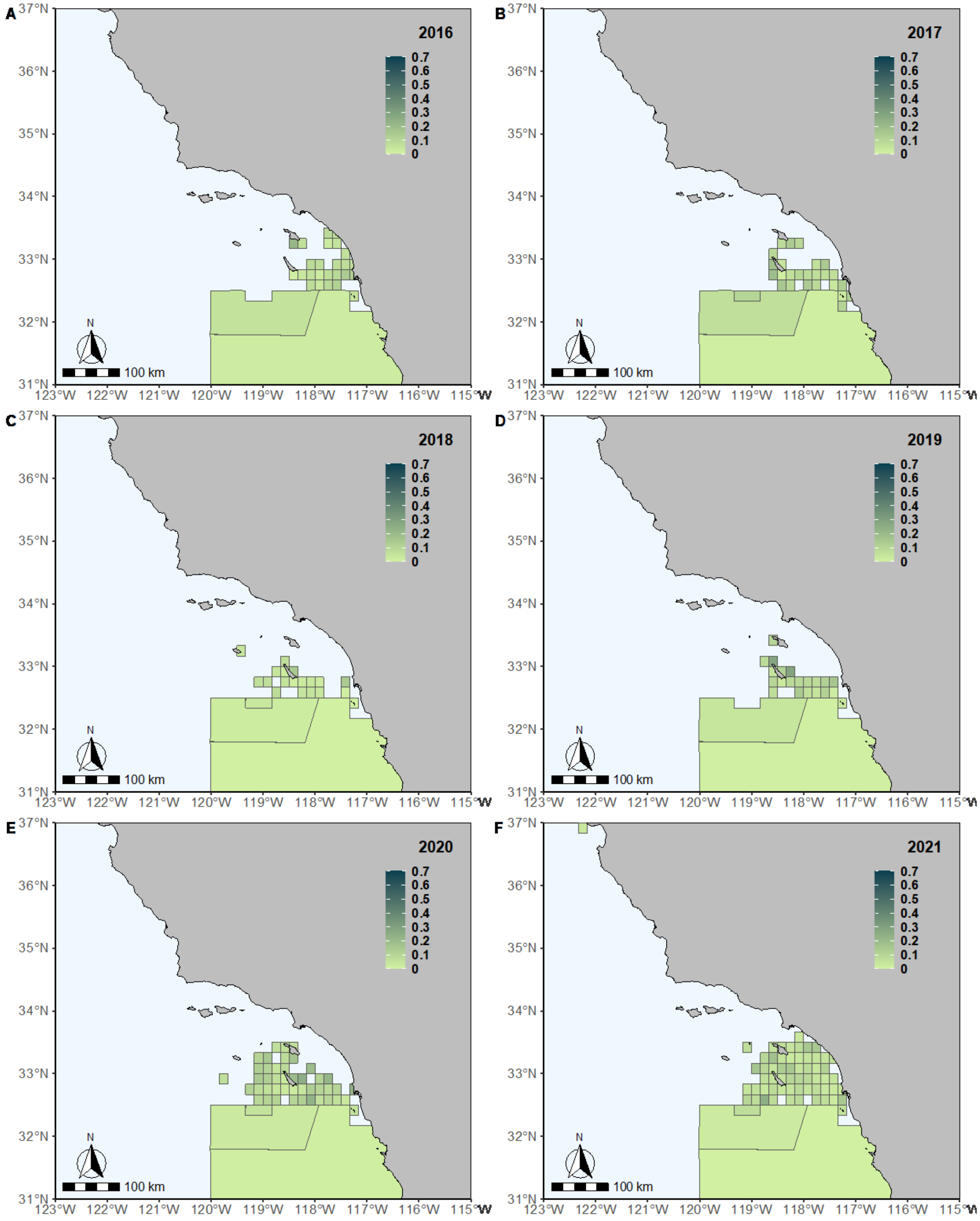
Supplemental Figure 1.13: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 1998 (A), 1999 (B), 2000 (C), 2001 (D), 2002 (E), and 2003 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



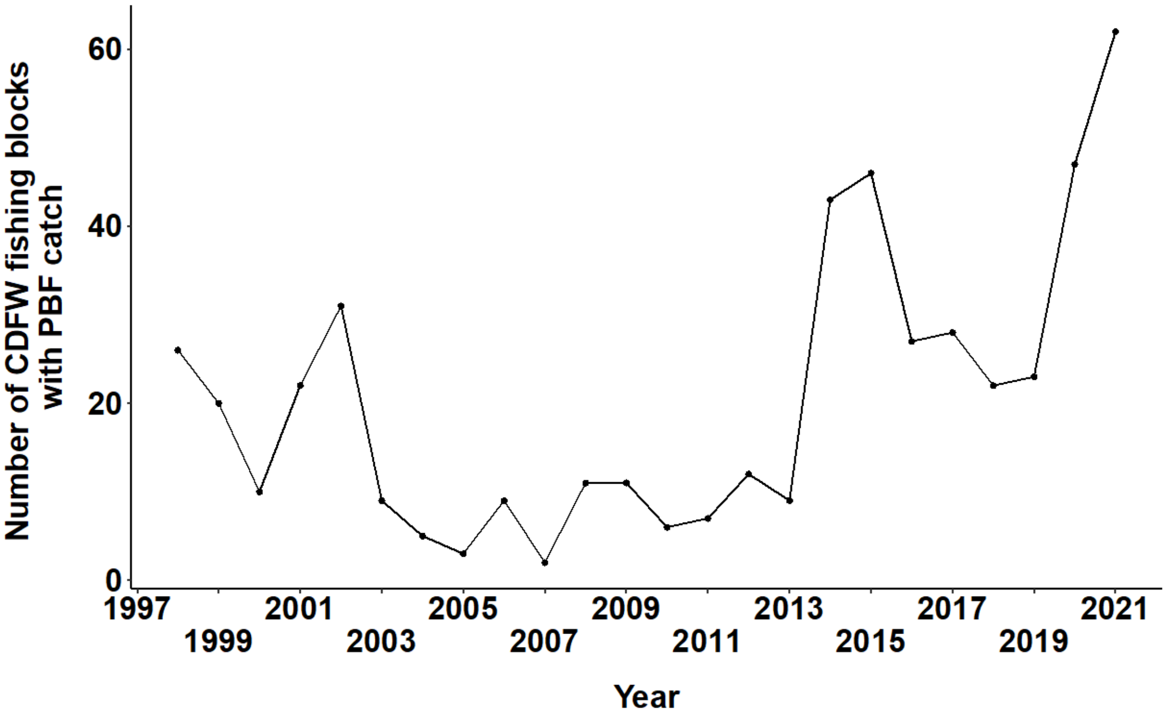
Supplemental Figure 1.14: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 2004 (A), 2005 (B), 2006 (C), 2007 (D), 2008 (E), and 2009 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



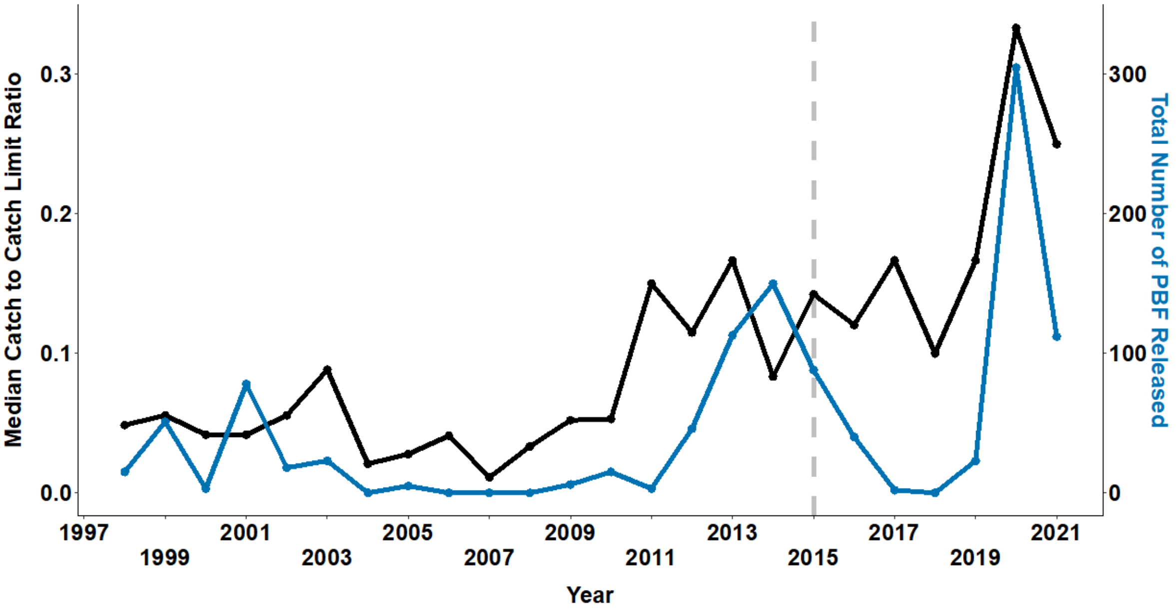
Supplemental Figure 1.15: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 2010 (A), 2011 (B), 2012 (C), 2013 (D), 2014 (E), and 2015 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



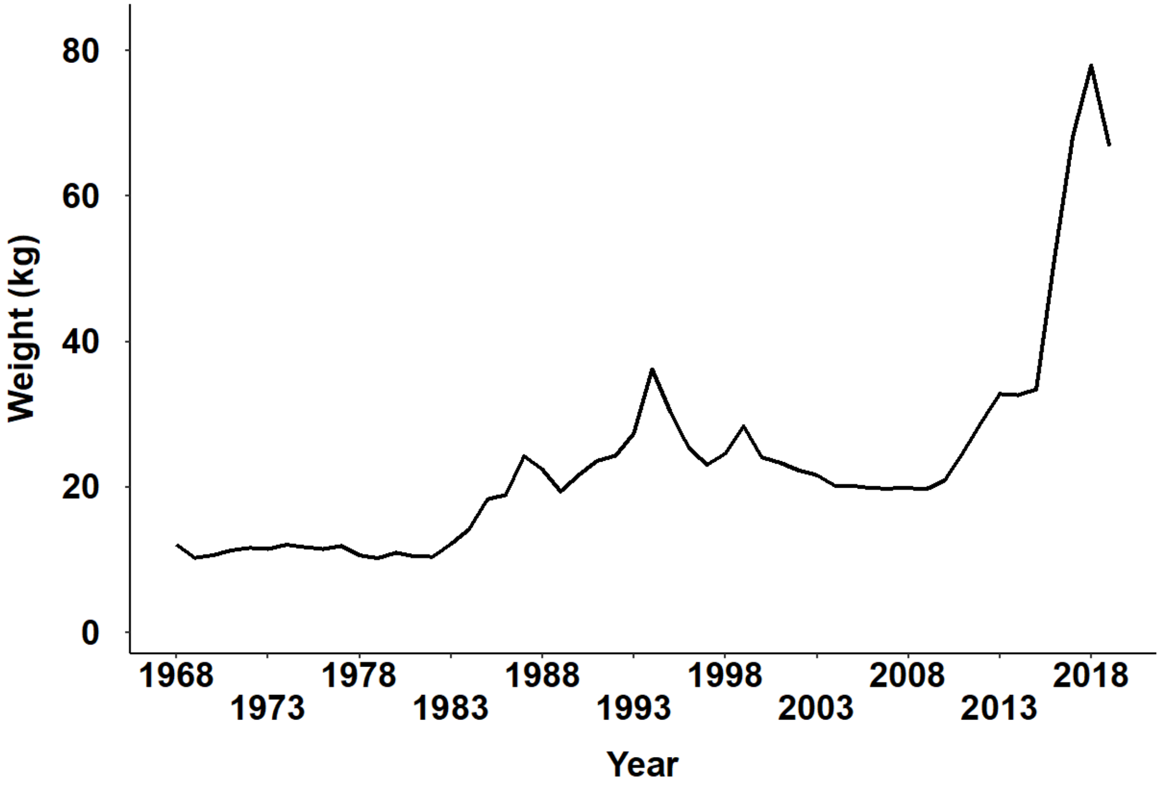
Supplemental Figure 1.16: 90% CI in PBF CPUE by CDFW fishing block throughout southern California waters in 2016 (A), 2017 (B), 2018 (C), 2019 (D), 2020 (E), and 2021 (F) with the effect of the environment excluded. Environmental variables are held constant at their respective means. Darker colors correspond to higher CPUE.



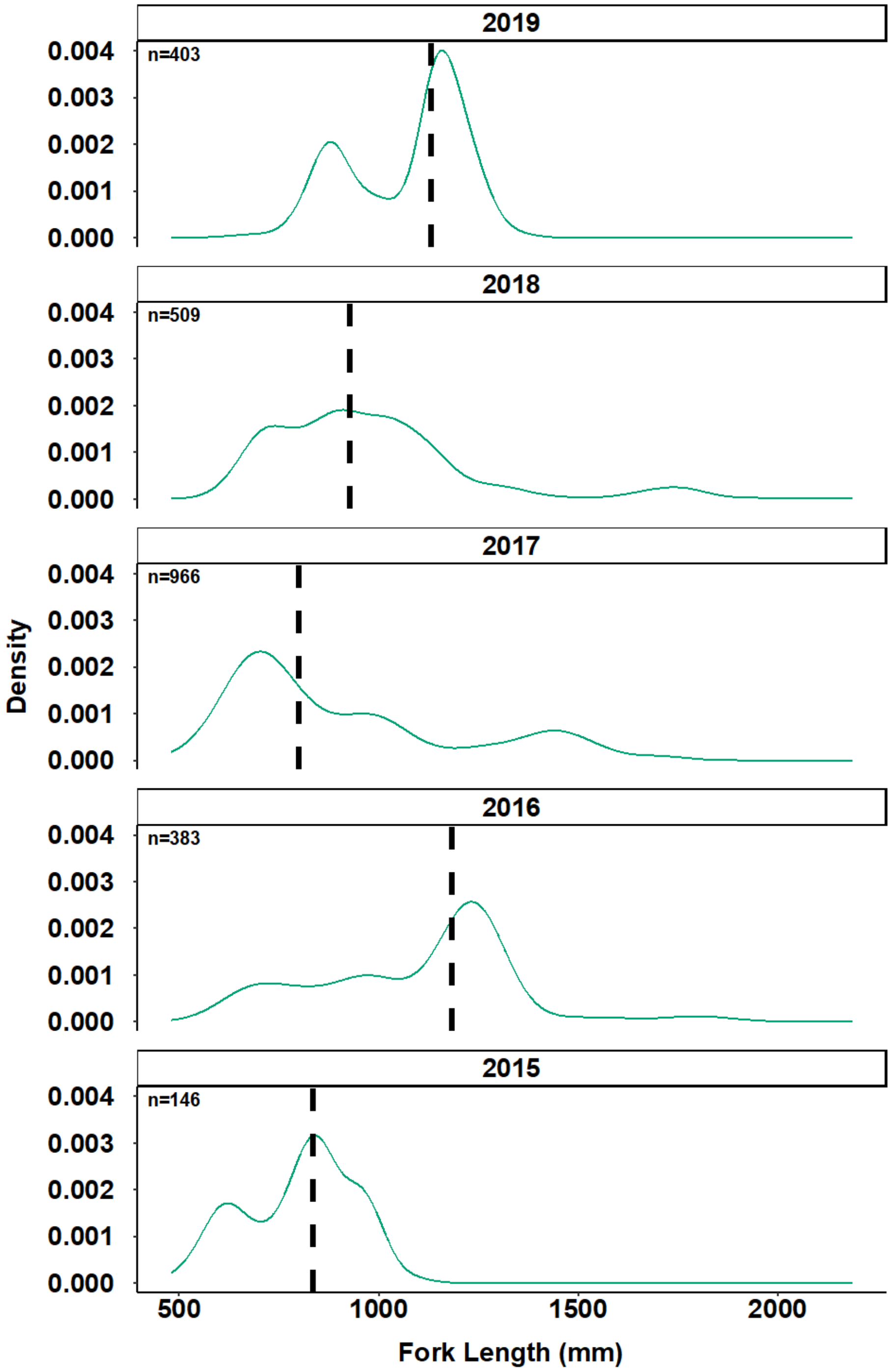
Supplemental Figure 1.17: Number of CDFW fishing blocks with PBF catch through time.



Supplemental Figure 1.18: Annual median catch to catch limit ratio per trip and total number of PBF released. The black line denotes the median catch to catch limit ratio per trip, the blue line denotes the total number of PBF released on all trips, and the grey dashed vertical line denotes the change in management (bag limit).



Supplemental Figure 1.19: Timeseries of trophy-sized PBF weight (kg) modeled from Bellquist et al (2016) Bayesian state-space time series model. The blackline represents mean and green shaded area represents 90% range in model outputs. For modeling methods, refer to Bellquist et al. (2016).



Supplemental Figure 1.20: Length frequency of PBF caught on-board CPFVs operating out of southern California ports and sampled by SAC from 2015 to 2019. Black dashed vertical lines represent median annual fork length of PBF.

Chapter 2

Long-term relative abundance trends of Caribbean parrotfishes reveal need for improved regional fisheries management

Jordan H. DiNardo, Dan A. Greenberg, Brice X. Semmens

Abstract

As a trophically diverse group of herbivores, parrotfishes play a role in competitive interactions between corals and algae, and thus likely an important part in supporting the resilience of Caribbean coral reefs. However, human impacts such as fishing have led to a substantial reduction in parrotfish abundance across the wider Caribbean region (WCR). Despite their depleted status and ecological significance, there has been no comprehensive assessment of the long-term population dynamics of parrotfish species across the WCR to date, due in large part to a lack of traditional fisheries dependent and independent data from the region. This study seeks to address this knowledge gap by modeling long-term citizen science data using multivariate autoregressive state-space models to investigate trends and synchrony in parrotfish relative abundance within and across functional groups (browsers, excavators, scrapers) throughout the WCR. Trends of parrotfishes were surprisingly variable through time, with most regions having more than double the relative abundance of parrotfishes between high and low abundance time periods; in general, there was no evidence for decline or recovery of parrotfishes across the WCR over the last three decades. Parrotfish population trends within functional groups were generally in synchrony, indicating the impacts of regional processes (top-down, bottom-up, and recruitment) are consistent across species. Synchrony in parrotfish relative abundance trends across functional groups was region specific, recognizing regional processes can have variable impacts across functional groups. These results collectively support the notion that protecting herbivore abundance without regard to functional groups may not be an effective strategy for conserving parrotfish in the WCR. Instead, implementing management measures that account for functional groups of parrotfishes and their regional synchrony may more effectively promote parrotfish conservation and more accurately acknowledge the distinctive roles parrotfish play in the recovery of Caribbean coral reefs.

Introduction

Understanding the dynamics of populations and ecosystems, including species interactions, is crucial for the efficient management of marine resources (Paine, 1988). This is particularly evident for coral reef ecosystems, which are among the most biodiverse, complex (D. R. Bellwood et al., 2012; Eddy et al., 2018; Hughes et al., 2002; Roberts et al., 2002) and threatened marine ecosystems (D. R. Bellwood et al., 2004; Brandl et al., 2019; Hughes et al., 2017; MacNeil et al., 2015). The wider Caribbean region (WCR) hosts some of the most degraded reefs in the world, driven in part by intense human impacts such as fishing, coastal development, and anthropogenic climate change (Hughes, 1994; E. J. Jackson et al., 2014; Pandolfi et al., 2003). As such, the survival and recovery of these reefs depends heavily on mitigating negative impacts of local and global stressors on essential ecological processes that support coral reef ecosystem functioning, such as herbivory (Brandl et al., 2019).

Herbivory supports coral reefs by limiting the distribution, abundance, and growth of algae (Carpenter, 1988; Hixon & Brostoff, 1996; Lewis, 1986; Morrison, 1988). In the WCR, the long-spined sea urchin *(Diadema antillarum)* was the dominant herbivore (Carpenter, 1986, 1988) until it experienced a disease-induced die-off in 1983 (Carpenter, 1988; Lessios, 1988). The mass-mortality of *D. antillarum* significantly reduced population densities (Hughes et al., 2010; E. J. Jackson et al., 2014; Ruttenberg et al., 2019; Schutte et al., 2010; Williams & Polunin, 2001), resulting in a phase shift from coral to macroalgae dominance across the WCR (D. R. Bellwood et al., 2004; Hughes et al., 2007, 2010). A wide variety of other anthropogenic stressors contributed to this rapid phase shift, including overfishing, increasing ocean temperatures, coral disease and predator outbreaks, and coastal pollution (D. R. Bellwood et al., 2004; E. J. Jackson et al., 2014; Schutte et al., 2010). Herbivorous fishes, such as parrotfishes (family Scaridae), have since been the dominant herbivores in Caribbean coral reef ecosystems, potentially filling the ecological role of *D. antillarum* in controlling algal growth and coverage (D. R. Bellwood et al., 2004; Hughes et al., 2010; E. J. Jackson et al., 2014; P. J. Mumby et al., 2006; P. J. Mumby & Steneck, 2008; Schutte et al., 2010; Williams & Polunin, 2001).

The parrotfishes are a diverse group of herbivorous reef fishes known for their fused beak-like jaws, which allows them to graze algae from reef substrate (D. Bellwood & Choat, 1990). The presence and abundance of parrotfishes facilitate corals by curbing algal growth that can have negative impacts on coral recruitment, growth, and survivorship (Adam et al., 2015; Hughes, 1994; Hughes et al., 2007; Ruttenberg et al., 2019). Parrotfish in the WCR fall into three broad functional groups, all of which play an essential role in the Caribbean coral reef ecosystem: 1) browsers (also known as croppers or grazers) graze on erect macroalgae and ultimately reducing macroalgae overgrowth and shading (Adam et al., 2018; D. R. Bellwood et al., 2004; Cardoso et al., 2009); 2) excavators feed by excavating crustose, endolithic algae, and microbes from within the reef framework, exposing the carbonate framework and assisting in sediment production on reefs as well as the settlement of corals and coralline algae (Adam et al., 2018); 3) scrapers feed by scraping the carbonate reef framework to remove turf algae and associated detritus, microbes, and infauna, creating space for coral settlement as well as assisting in the growth and survival of coralline algae and corals (D. Bellwood & Choat, 1990).

Though parrotfishes play several key roles in the WCR coral reef ecosystem, they also have a long history of overexploitation in the region (Hardt, 2008; Hughes, 1994; E. J. Jackson et al., 2014; Pandolfi et al., 2003). Although parrotfishes were typically not the target of choice for Caribbean artisanal fisheries, the overexploitation of higher trophic-level species, such as grouper and snappers, resulted in many fisheries shifting to more reliable and plentiful herbivorous fishes instead (McClenachan, 2009; P. Mumby et al., 2012). While parrotfishes are vulnerable to most fishing techniques, trap-fishing and spearfishing largely contributed to their steep decline throughout the WCR (E. J. Jackson et al., 2014; Johnson, 2010; P. Mumby et al., 2012) and consequently many parrotfish populations were classified as overfished by the mid-20th century (E. J. Jackson et al., 2014). The first quantitative underwater surveys of abundance were conducted in the late 1970s and 1980s well after parrotfish were deemed overfished (E. J. Jackson et al., 2014). Only three surveys of parrotfish biomass that span longer than ten years have been conducted (St.John, Bonaire, and Guadalupe), all of which depicted the depleted state of parrotfishes by the first decade of the 21st century. Of these, only Guadeloupe showed a slight gradual increase in biomass, though the drivers to this increase were unclear (Steneck et al., 2009, 2011). Developing management strategies that promote the ecological functioning of parrotfishes, requires a comprehensive understanding of status and trends of parrotfish species across the WCR.

Noting the extensive history of overfishing in the WCR and its ensuing effects on coral reefs, the Cartagena Convention, as a legal regional framework for the protection of the Caribbean Sea, was adopted in 1983 and entered into force in 1986. To date, 26 United Nations (UN) member states in the WCR have ratified the Cartagena Convention (United Nations Environment Programme, 2021). The regional Convention is divided into three technical agreements, one of which is the Specially Protected Areas and Wildlife (SPAW) Protocol that came into effect in 2000. Eighteen member states have ratified the SPAW Protocol to provide a legal framework for the conservation of regional biodiversity by obligating contracting states to protect vulnerable ecosystems and species in the Caribbean, including parrotfishes (United Nations Environment Programme, 2021). Eleven of the eighteen states that endorsed the protocol have developed regulatory measures on herbivorous fish, including species- and family-specific fishing bans, size limits, and protected areas (AIDA, 2021). While some states currently have or are working towards establishing fishing restrictions, many states throughout the WCR remain without any identified regulatory measures (AIDA, 2021). In 2018, the Scientific and Technical Advisory Committee of the SPAW Protocol prioritized the evaluation of herbivorous fishes and developed a Species Working Group to evaluate parrotfish populations (AIDA, 2021). This task has proven to be challenging as little is known about stock status and population dynamics, either for individual parrotfish species or collectively. Improving our understanding of the population dynamics of parrotfish is critical for determining and implementing best practices for conservation efforts to recover these species and the coral reefs they reside in.

In this study we investigated population trends of ten parrotfish species across three decades, and throughout the WCR, to inform management and conservation efforts. To do so, we use citizen science observations from the Reef Environmental Education Foundation (REEF) Volunteer Fish Survey Project to derive region-specific relative annual abundances of Caribbean parrotfishes. Using these species-specific relative abundances, we next calculated the extent of synchrony in population trends within and across functional groups, and generated aggregate functional group and cumulative parrotfish (all species combined) population trends. Collectively, this study supports the notion that efforts to protect and restore herbivore abundance will be aided by broadly similar responses across parrotfish species, particularly within functional groups, however given that partial decoupling across functional groups, managers should carefully consider how proposed management actions may act independently on each functional group.

Materials and methods

Parrotfish abundance data and site selection

We obtained parrotfish abundance data from the Reef Environmental Education Foundation (REEF) Volunteer Fish Survey Project, a citizen science program that has generated over 200,000 surveys in the WCR since its inception in 1993 (REEF, 2023). REEF surveyors use a roving diver technique to survey a wide variety of habitats within a particular site and report all fish, including parrotfish, observed throughout the water column during regular dive activities. Surveyors vary in experience in fish identification (novice to expert), which is determined by the number of REEF surveys completed and examination scores. Each species observed and positively identified during a survey is assigned to an abundance category based on how many were seen throughout the dive: single [1], few [2-10], many [11-100], or abundant [>100].

Because REEF surveys are conducted opportunistically by volunteer divers, survey effort has been highly variable through time across the WCR. While many locations are surveyed infrequently, a few regions have had fairly consistent survey effort throughout the duration of the Volunteer Fish Survey Project, due largely to robust dive tourism infrastructure. From this subset of highly-surveyed locations, we selected Bonaire, Cozumel, Key Largo, Little Cayman, Roatan, and Tortola to develop a comprehensive representation of parrotfish population dynamics across the WCR and its implication on fisheries management (Supplemental Figure 2.1 Supplemental Table 2.1, Supplemental Table 2.2).

We limited our analysis to ten large-bodied parrotfish species in the WCR (Supplemental Table 2.3): *Scarus coelestinus* (Midnight Parrotfish), *Sc. coeruleus* (Blue Parrotfish), *Sc. guacamaia* (Rainbow Parrotfish), *Sc. iseri* (Striped Parrotfish), *Sc. taeniopterus* (Princess Parrotfish), *Sc. vetula* (Queen Parrotfish), *Sparisoma aurofrenatum* (Redband Parrotfish), *Sp. chrysopterum* (Redtail parrotfish), *Sp. rubripinne* (Yellowtail Parrotfish), and *Sp. viride*

(Stoplight Parrotfish). These parrotfish species are important herbivores of the WCR reefs and have been categorized into the three general functional groups: Redband Parrotfish, Yellowtail Parrotfish, and Redtail Parrotfish comprise the browsers; Midnight Parrotfish, Rainbow Parrotfish, and Stoplight Parrotfish comprise the excavators; and Blue Parrotfish, Princess Parrotfish, Queen Parrotfish, and Striped Parrotfish comprise the scrapers (Adam et al., 2018; Alfaro & Westneat, 1999; Bruggemann et al., 1996; Cardoso et al., 2009; Harborne & Mumby, 2018; Ruttenberg et al., 2019). Finally, we limited our analysis to surveys conducted between 1994 and 2019, ranging in duration from 20 minutes to 120 minutes and conducted by expert surveyors (surveyors who conducted at least 35 surveys and passed the REEF experience level exam with at least 90% accuracy) during the day with valid geolocation documentation.

Bayesian multivariate autoregressive state-space modeling

We used Bayesian multivariate autoregressive state-space (MARSS) modeling to analyze long-term trends of parrotfish relative abundance across the WCR. MARSS modeling is widely used in ecology to model species' population trends because of its ability to separate process (state) and observation variability (Hinrichsen & Holmes, 2009; Holmes et al., 2012). Decoupling these two sources of variability helps to discern temporal changes in population size due to environmental and demographic stochasticity (process variability) from changes due to variation in sampling, measurement, and detection error (observation variability) (Ward et al., 2010).

The general approach of state-space modeling assumes the true population abundance (on a log scale) varies through time following a first-order autoregressive process with yearly log-normal deviations ($\boldsymbol{w_t}$) influenced by demographic processes (process variance,$\boldsymbol{\sigma^2}$).

$$\boldsymbol{x_t = x_{t-1} + w_t \qquad w_t \sim N(0,\sigma^2)}$$

This differs from the annual abundance observed in surveys (***y_t***), which is derived from the true, unobserved population state for a given year and additional yearly stochastic processes (***v_t***) influenced by observation variance ($\boldsymbol{\tau^2}$).

$$\boldsymbol{y_t = x_t + v_t \qquad v_t \sim N(0, \tau^2)}$$

Expanding this state-space modeling approach to MARSS modeling of multiple species, each with their own time series of abundance, we altered the above equations to estimate the synchrony between their underlying states (***$x_{1,t}, \ldots, x_{m,t}$***) by jointly deriving their population deviations from correlated stochastic process. While the number of underlying (***m***) and observed (***n***) states can vary (Ward et al. 2010), for our analysis we treated each species as a unique underlying and observed state (***n*** = ***m*** = 10) in each region. The variance-covariance matrix (***Q***) is described by within species variance ($\sigma_1^2, \ldots, \sigma_m^2$) and their correlation ($\rho_{1,m}$). The observation variance associated with each species as independent sampling errors in each species (***$v_{1,t}, \ldots, v_{m,t}$***).

$$\begin{bmatrix} x_{1,t} \\ \cdots \\ x_{m,t} \end{bmatrix} = \begin{bmatrix} x_{1,t-1} + w_{1,t} \\ \cdots \\ x_{m,t-1} + w_{m,t} \end{bmatrix} \quad \begin{bmatrix} w_{1,t} \\ \cdots \\ w_{m,t} \end{bmatrix} \sim MVNormal\left(\begin{bmatrix} 0 \\ 0 \\ 0 \end{bmatrix}, Q\right) \qquad Q = \begin{bmatrix} \sigma_1^2 & \cdots & \rho_{1,m} \\ \vdots & \ddots & \vdots \\ \rho_{1,m} & \cdots & \sigma_m^2 \end{bmatrix}$$

$$\begin{bmatrix} \alpha_{1,t} \\ \cdots \\ \alpha_{n,t} \end{bmatrix} = \begin{bmatrix} x_{1,t} + v_{1,t} \\ \cdots \\ x_{n,t} + v_{n,t} \end{bmatrix} \qquad \begin{matrix} v_{1,t} \sim N(0, \tau_1^2) \\ \cdots \\ v_{n,t} \sim N(0, \tau_n^2) \end{matrix}$$

Since species' observations in each REEF survey (***i***) are reported as one of five (***K***=5) abundance categories (***Y: 0*, 1, 2-10, 11-100, or >100; 0* is derived from non-sightings***), we used an ordinal logistic regression developed by Greenberg et al. (Greenberg et al., 2024) to model the probability of observing each abundance category for a parrotfish species on a survey. Cut points (***$c_1, \ldots, c_{K-1}$***) estimated from an induced Dirichlet prior, categorize the latent abundance ($\boldsymbol{\eta_i}$) into probability intervals based on a set of linear predictors thought to influence diver observations including, year ($\boldsymbol{\alpha_t}$), site ($\boldsymbol{\alpha_s}$), diver ($\boldsymbol{\alpha_{dv}}$), and day of year ($\boldsymbol{\alpha_{dmy}}$) in design matrix ($\boldsymbol{X_i}$).

$$\boldsymbol{Ordered\ Logistic(y_i|\eta_{t,s,dv,dmy,i}, c_{1-K-1})}$$

$$= \begin{cases} \boldsymbol{logit^{-1}(c_1 - \eta_i)} & \boldsymbol{Y = 0} \\ \boldsymbol{logit^{-1}(c_2 - \eta_i) - logit^{-1}(c_1 - \eta_i)} & \boldsymbol{Y = 1} \\ \boldsymbol{logit^{-1}(c_3 - \eta_i) - logit^{-1}(c_2 - \eta_i)} & \boldsymbol{Y = 2 - 10} \\ \boldsymbol{logit^{-1}(c_4 - \eta_i) - logit^{-1}(c_3 - \eta_i)} & \boldsymbol{Y = 11 - 100} \\ \boldsymbol{1 - logit^{-1}(c_4 - \eta_i)} & \boldsymbol{Y > 100} \end{cases}$$

$$\boldsymbol{\eta_{t,s,dv,dmy,i} = \alpha_t + \alpha_s + \alpha_{dv} + \alpha_{dmy} + X_i}$$

We performed all modeling work in R (v4.0.2) using the ‘rstan’ package (v2.21.5; Guo et al., 2023). Posterior estimates were approximated from four chains via the No-U-Turn sampler variant of the Hamiltonian Monte Carlo algorithm (Betancourt, 2018; Hoffman & Gelman, 2011) in Stan. For each of the chains, we set the number of iterations to 2,000, the number of warmups to 1,000, and thinning to 1, resulting in an output of 4,000 total posterior samples. We checked that all $\hat{R}$ ratio <1.05 and $N_{eff} > 400$ (Gelman & Rubin, 1992) using the ‘coda’ package (v0.19-4; Plummer et al., 2020) in R. Refer to Supplemental Information for the full model with prior specifications (Supplemental Table 2.4).

Modeling framework

We modeled the relative abundance of all ten parrotfish species, representative of the three functional groups, in the six regions across the WCR separately, equating to six models, one for each region. We transformed posterior outputs of the model (probability of observing each abundance category for a parrotfish species on a survey) to abundance values by assigning each abundance category a corresponding minimum number of individuals (***Y: 0*, 1, 2, 11, and 101; 0* is derived from non-sightings***). The summation of this equates the abundance of a given parrotfish on a survey. These transformed posterior outputs of species-specific relative abundance of all ten parrotfish in each region were aggregated to investigate relative abundance of parrotfishes in space and time cumulatively and by functional group. Relative abundances trends were derived by summing the total number of parrotfishes per region (for cumulative relative abundance trends of parrotfishes) and per region and functional group (for relative abundance trends by functional groups) then dividing by the annul number of surveys for each region. For the regional cumulative relative abundance trends, we derived the regional slope estimates from a simple linear regression on the median yearly values to assess the general trends of parrotfishes across the WCR over the three decades. We also derived a median cumulative relative abundance of parrotfishes (aggregated over years) for each region to compare with their overfishing threat, to explore the association between regional parrotfish abundance and overfishing threat, which was estimated by the World Resource Institute (World Resources Institute, 2004). For regions that had area deemed have both medium and high threat of overfishing, we created a new overfishing category (Medium/High).

We used the posterior distribution of the correlation coefficient ($\rho_{1,m}$) in abundance trends between each parrotfish species pair (interspecies comparison) as a measure of species-specific synchrony in each region. Within each functional group (e.g., browsers, which comprises 3 interspecies comparisons) and across functional group comparison (e.g., scraper-browser, which comprises of 12 interspecies comparisons), we derived a posterior median group correlation. Additionally, we estimated the correlation coefficient across functional groups (functional group comparison) by first aggregating the annual posterior median estimates of relative abundance for each species in each functional group, and then subsequently calculating correlations between the yearly aggregated relative abundances of pair-wise functional groups using the ‘cor.test’ function in base R. When making inferences about synchrony across functional groups, we drew from the correlation coefficients derived from functional group comparisons, rather than that derived from interspecies comparisons. While interspecies comparisons are valuable in assessing synchrony within functional groups, the number of interspecies comparisons across functional groups makes it challenging to make meaningful inferences.

Owing to the complexity in investigating the synchrony in relative abundance trends between ten parrotfish species (equating 45 interspecies comparisons) that make up three functional groups in six regions across the WCR, we establish an interpretive framework representing four synchronous states of nature, and the potential ecological processes driving these states (Table 2.1). Each synchronous state of nature corresponds to the sign of correlation between taxa (positive or negative, rows) and system (within or across functional groups, columns) and describes the underlying suite of mechanisms that may produce such correlations (see text within each quadrant). For instance, if the relative abundance of species within a functional group are positively correlated (upper left quadrant, Table 2.1), it is likely that top down (e.g., fishing) or bottom up (e.g., productivity) processes, or recruitment processes, are driving synchrony. Additionally, these processes are masking any signal of within functional group competitive interactions. For the sake of simplicity, we only focus on the sign of correlation within and across functional groups: whether it is positive (synchronous) or negative (asynchronous). The interpretive framework allows us to place our results in the context of these synchronous states of nature, and the potential underlying processes driving parrotfish dynamics in each region and system.

Table 2.1: The four synchronous states of nature that correspond to the sign of correlation (positive or negative rows) and system (within or across functional groups columns). Each synchronous state of nature describes the underlying processes (text within each quadrant) that may be contributing to the corresponding sign of correlation and system for a given region.

	Within Functional Groups	**Across Functional Groups**
+	**A** Top-down or bottom-up processes that are trophically related to the population dynamics of parrotfish have a similar impact on parrotfish species within a functional group. Recruitment processes coupled across species within functional groups. Competition in negligible on the population dynamics of parrotfish species within a functional group.	**D** Top-down or bottom-up processes that are trophically related to the population dynamics of parrotfish have a similar impact on parrotfish species across functional groups. Recruitment processes coupled across functional groups.
−	**B** Top-down or bottom-up processes that are trophically related to the population dynamics of parrotfish have a varying impact on parrotfish species within a functional group. Recruitment processes decoupled across species within functional groups. Competition is influential on the population dynamics of parrotfish species within a functional group.	**C** Top-down or bottom-up processes that are trophically related to the population dynamics of parrotfish have a varying impact on parrotfish species across functional groups. Recruitment processes decoupled across functional groups.

Results

Relative abundance trends of parrotfish across the WCR

The cumulative relative abundance of all ten parrotfishes was highly variable through time across the WCR, in some regions more than others (Figure 2.1). Little Cayman exhibited the most variability through time while Tortola had the least variability in cumulative relative abundance of parrotfish through time. In some regions the relative abundance of parrotfish exhibited an increasing trend since 1994, most notably in Bonaire, Roatan, and Little Cayman. In these regions, the relative abundance of parrotfishes peaked in 2000, 2008, 2012, respectively, thereafter declining to lower levels. In the remaining regions across the WCR, parrotfish relative abundance fluctuated at low levels through time. Since 2012, the relative abundance of parrotfishes generally exhibited an increasing trend in most regions across the WCR. Despite this variability in relative abundance through time, the overall trends in parrotfish relative abundance are negligible across the WCR (Bonaire: -0.322; Key Largo: -0.008837; Cozumel: 0.432; Little Cayman: 0.563: Roatan: 0.1896; 0.0513). Across regions, the relative abundance of parrotfishes is largely driven by a subset of five parrotfish species, Princess Parrotfish, Queen Parrotfish, Redband Parrotfish, Stoplight Parrotfish, and Striped Parrotfish. For species-specific relative abundance trends for all ten parrotfishes in each region refer to Supplemental information (Supplemental Figure 2.2-Supplemental Figure 2.7). Overfishing threat exhibited a negative impact on median cumulative relative abundance across regions (Figure 2.1B). Regions with lower overfishing threats, including Little Cayman and Bonaire, had higher relative abundances than regions with higher overfishing threat (Cozumel and Key Largo).

The relative abundance of parrotfishes aggregated by functional groups were also highly variable in time across the WCR (Figure 2.2). In Bonaire, Roatan, and Little Cayman, scrapers tended to have the highest relative abundance through time, followed by excavators and browsers. The peaks in cumulative relative abundance trends observed in 2000 and 2012 in Bonaire and Little Cayman, respectively, were most attributable to the scrapers' relative abundance. In Key Largo and Tortola, where the cumulative relative abundance trends of parrotfish were relatively low through time, excavators generally tended to have the highest relative abundance through time compared to the other functional groups. In Cozumel, the relative abundance of parrotfish was similar across all three functional group early in the time series; however, in 2007, the relative abundance of browsers noticeably increased. The relative abundance trends of the other functional groups started to increase in 2012 but to a lesser extent.

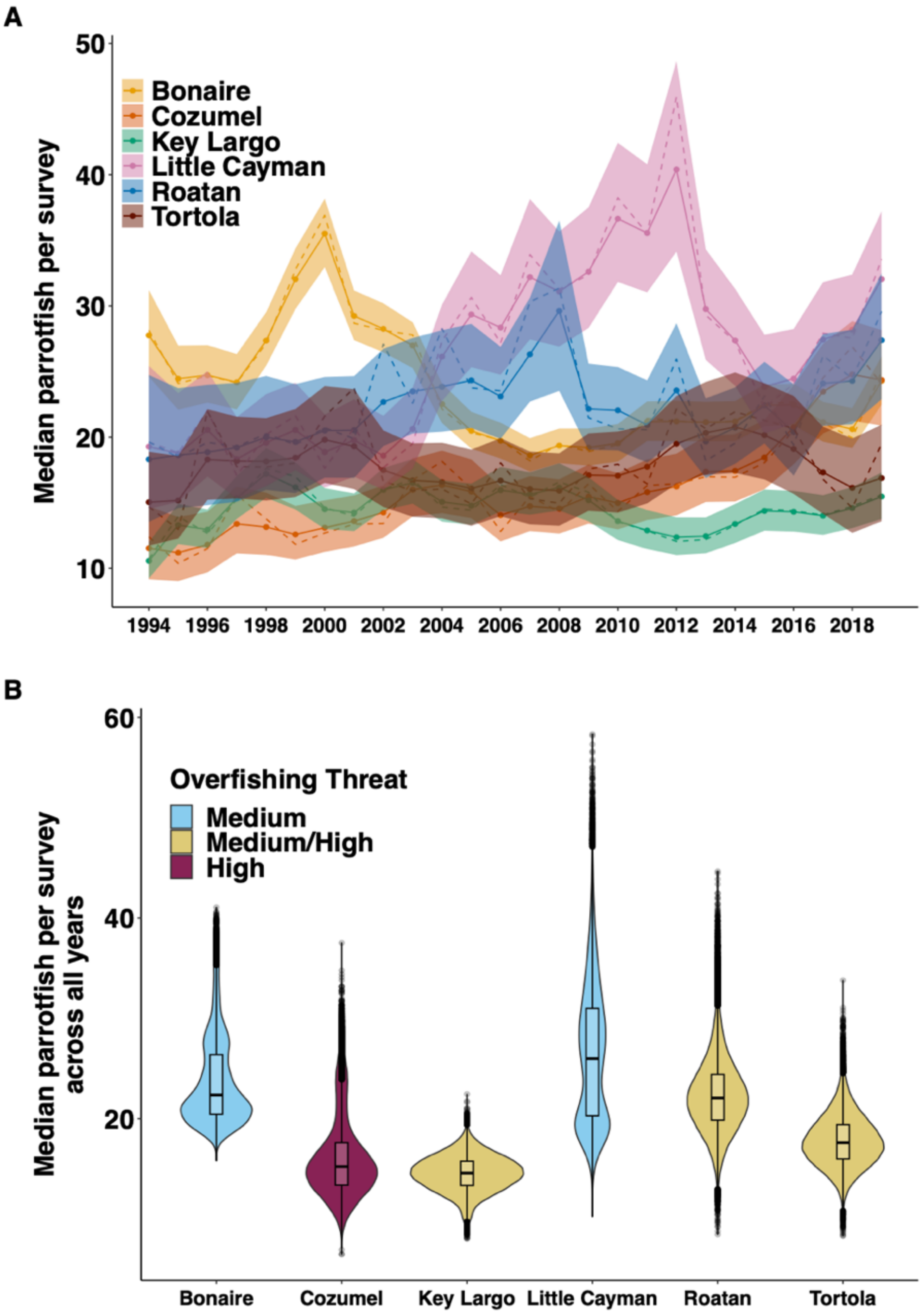


Figure 2.1: Cumulative relative abundance for the ten parrotfish species from 1994 to 2019 by region (A). Median cumulative relative abundance of all ten parrotfish across all years by region (B). In panel A, regional trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated over the ten parrotfish species. The solid line represents state process while the dashed line represents observation process for each region. In panel B, colors denote overfishing threat where blue corresponds to medium, yellow corresponds to medium/high and red corresponds to high overfishing threat. Level of overfishing threat was derived by the World Resource Institute (WRI).

Synchrony in relative abundance trends of parrotfish across functional groups

Synchrony in relative abundance trends of parrotfish functional groups varied across regions (Figure 2.2; Supplemental Table 2.5). In Bonaire, Little Cayman, and Tortola, all three functional group comparisons were positive, implying recruitment, top-down, bottom-up, or recruitment processes are impacting these functional groups similarly in these regions (quadrant D). For the remaining regions that exhibit at least one negative functional group comparison, the impact of these synchronous processes varies across functional group (quadrant C). In Key Largo and Roatan, the median correlation for the scraper-browser functional group comparison were positive while the median correlation for the excavator-browser and scraper-excavator functional group comparisons were negative. In Cozumel, the median correlation for the excavators-browsers was negative while the median correlation for the scraper-browser and scraper-excavator functional group comparisons were positive. Interestingly, across all regions, the median correlation for the scraper-browser functional group was positive, suggesting recruitment, top-down, bottom-up, or recruitment processes are impacting these functional groups similarly across the WCR (quadrant D).

Synchrony in relative abundance trends of parrotfish across functional groups in regards to interspecies comparisons varied considerably across regions as well (Supplemental Table 2.7-Supplemental Table 2.14). In all regions, except Roatan, the majority of interspecies comparisons were positive. In Bonaire, Key Largo, and Cozumel, the median group correlations were positive for the scraper-browser and scraper-excavator functional group comparisons and negative for the excavator-browser functional group comparison (Supplemental Table 2.8). In Little Cayman and Tortola, the median group correlation for all three functional group comparisons were positive. In Roatan, the median group correlations were negative for the excavator-browser and scraper-browser functional group comparisons and positive for the scraper-excavator functional group comparison.

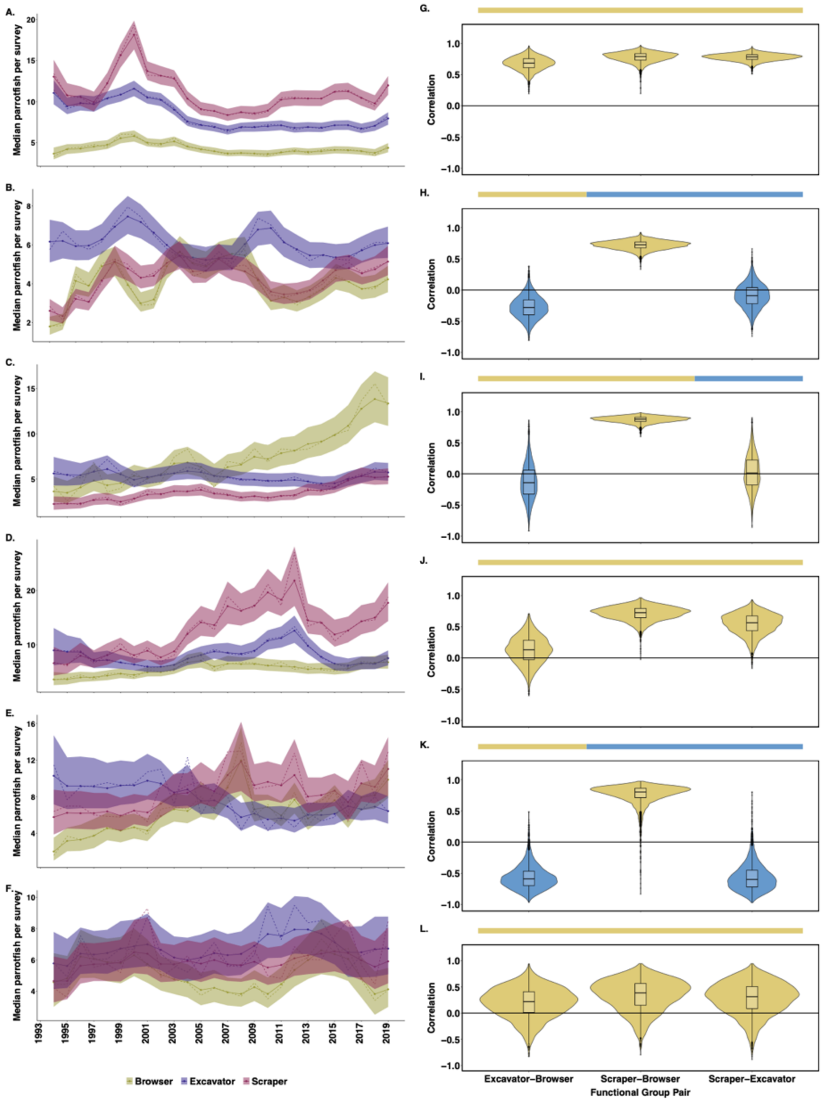


Figure 2.2: Cumulative relative abundance of parrotfish species from 1994 to 2019 by functional group (A-F) and their corresponding correlations between each functional group comparison for Bonaire (A and G), Cozumel (B and H), Key Largo (C and I), Little Cayman (D and J), Roatan (E and K), and Tortola (F and L). In panels A-F, the relative abundance trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated over parrotfish species corresponding to each functional group. The solid trend line represents state process while the dashed trend line represents observation process for each functional group in each region. In panels G-L, the violin plots represent the distribution of correlation between the cumulative relative abundance trends of each functional group comparison. The colors denote the sign of the median correlation for a given functional group comparison, where yellow denotes positive, and Blue denotes negative correlations. The horizontal bars at the top of each panel denote the proportion of positive and negative functional group comparisons for each region.

Synchrony in relative abundance trends of parrotfish within functional groups

In all six regions across the WCR, most interspecies comparisons within functional groups were positive (Figure 2.3, Supplemental Table 2.7) implying that the effects of competitive interactions between parrotfish species within each functional group are swamped by recruitment and/or top-down/bottom-up processes (quadrant A). For browsers, all regions had a positive median group correlation across all interspecies comparisons, except for Little Cayman. This patten is reflected in the specific interspecies comparisons, in that all three interspecies comparisons amongst the browsers exhibited a positive median correlation except for Little Cayman, where the median correlation for one interspecies comparison was negative (Redband Parrotfish-Yellowtail Parrotfish). Within the excavators, all regions had positive median group correlation across all interspecies comparisons, except for Cozumel. In Bonaire, Key Largo, Little Cayman, and Tortola, all three interspecies comparisons amongst the excavators exhibited a positive median correlation. Whereas in Cozumel and Roatan, two (Midnight Parrotfish -Stoplight Parrotfish and Rainbow Parrotfish -Stoplight Parrotfish) and one (Rainbow Parrotfish -Stoplight Parrotfish) interspecies comparisons had a negative median correlation, respectively. Within the scrapers, all regions, except Roatan and Tortola, had positive median group correlations across all interspecies comparisons. While all six interspecies comparisons amongst the scrapers had positive median correlations in Bonaire and Little Cayman, the interspecies comparisons amongst the scrapers exhibited varying signs in median correlation across the other four regions. For median estimates for each interspecies comparison within functional groups by region refer to Supplemental Information (Supplemental Table 2.15-Supplemental Table 2.20).

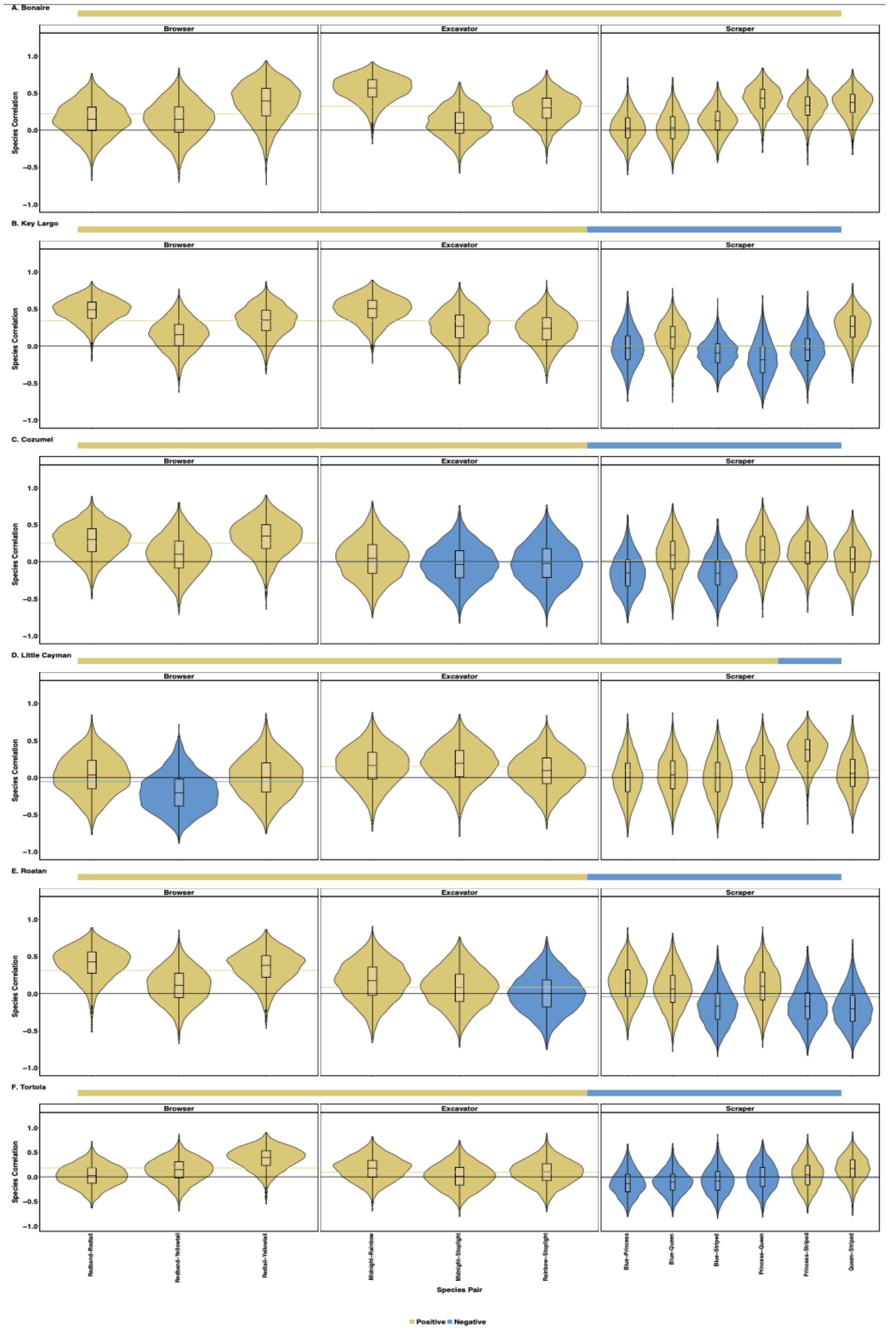


Figure 2.3: Correlation between modeled parrotfish abundance trends representative of each interspecies comparison within each functional group in Bonaire (A), Cozumel (B), Key Largo (C), Little Cayman (D), Roatan (E), Tortola (F). The correlation of each interspecies comparison is represented as violin plots depicting distribution of posterior estimates from the variance-covariance matrix. The colors denote the sign of the median correlation for a given interspecies comparison, where yellow denotes positive, and Blue denotes negative correlations. The horizontal lines in each panel represent median group correlation across all interspecies comparisons and the horizontal bars at the top of each panel denote the proportion of positive and negative interspecies comparisons within each functional group.

Discussion

Establishing management actions to preserve or rebuild herbivores on coral reefs requires an understanding of how management mechanisms (e.g., protected areas, gear restrictions) are likely to act on the diverse community of herbivorous species that collectively act to mediate coral-algal interactions. If the dynamics of herbivore populations are largely decoupled from each other, managers will be challenged to assess the likely impact of management for each member of the herbivore community separately. On the other hand, if herbivores synchronously respond to the ecology and management of a reef, the task of a manager becomes one of generality – effective management for some herbivores are likely to benefit most/all. In this study, we modeled the relative abundance of parrotfish across the WCR and evaluated the synchrony in their populations within and across functional groups using a uniquely long timeseries of observations from citizen science scuba divers participating in REEF's Volunteer Fish Survey Project. The results from this work indicate the relative abundance trends of parrotfish throughout the WCR were variable through time, in some regions more than others and contingent on regional processes. However, synchrony in parrotfish species was generally positive within functional groups, suggesting that trophically similar parrotfish species are responding similarly to any or all ecological processes, fishing pressure and management actions. On the other hand, synchrony across functional groups was less consistent, suggesting that, at least in some regions, the population dynamics of scrapers, browsers and excavators are partially decoupled, and may be responding differently to ecological and anthropogenic drivers. Our findings collectively support the notion that efforts to protect and restore herbivore abundance will be aided by broadly similar responses across parrotfish species, particularly within functional feeding guilds. That said, given that partial decoupling across functional groups, managers should carefully consider how proposed management actions may act independently on each functional group.

The value of citizen science data

Given the nature of a volunteer dataset, there are a few limitations. A common issue with volunteer-based data collection efforts is the potential sampling bias. Like analyses using catch-per-unit-effort to derive a standardized index of abundance, our study is susceptible to non-random sampling in space and time. In addition, the roving diver technique performed by REEF surveyors yields ‘counts per unit time’ rather than ‘abundance per unit space’. Consequently, estimates generated are not directly relatable to other standard survey methods that focus on numbers of individuals per unit of space. Finally, size is not a part of the roving diving survey, so estimation of biomass is not possible from the REEF data. Consequently, variability in count data may be influenced by the recruitment of young individuals.

Despite these limitations, the REEF citizen science data used in this study offer a unique and valuable perspective, particularly considering the absence of species-specific trend data from fisheries monitoring programs across the WCR. Leveraging citizen contributions has enabled us to gain insights into parrotfish populations and dynamics that would otherwise be unavailable.

Cumulative relative abundance trends of parrotfish across the WCR

Our results suggest that over the past three decades there has not been substantial evidence of pervasive declines or increases of parrotfish populations across the WCR, however parrotfish populations exhibited variability in space and time. This variability in parrotfish relative abundance can be at least partially attributed to the variability in anthropogenic factors (fisheries, habitat degradation, human-induced climate change, conservation efforts) across the WCR. The WCR is a large and complex maritime region that exhibits an array of ecoregions that vary in coral reef benthic communities and associated fish assemblages (Francisco-Ramos &

Arias-González, 2013). Furthermore, anthropogenic stressors, including but not limited to human population growth, overfishing, and environmental changes have been negatively impacting coral reefs across the WCR to varying degrees.

Historical fishing pressure varied across the WCR due to a combination of fishing practices (e.g., fishing gears), economic well-being, and cultural traditions (E. J. Jackson et al., 2014). By the 1970s, the Greater and Lesser Antilles (West Indies) were densely populated regions that had developed labor intensive artisanal fisheries to exploit local reef fishes. Consequently local fish populations were deemed overfished by the early 20th century (Hawkins & Roberts, 2004). In contrast areas along the coast of Florida, Mexico, the Mesoamerican Barrier Reef, and northern South America were less densely populated and thus exhibited less fishing pressure until the 1970s to 1990s (J. B. C. Jackson, 1997; McClenachan, 2009). This variation in fishing pressure in space and time across the WCR has resulted in similar disparities in fisheries management, which partly contributes to the variability in parrotfish relative abundance across regions. Bonaire, in particular, implemented conservation measures to protect reef fishes in 1971 with the banning of spearfishing. This spearfishing ban was followed by the establishment of the Bonaire National Marine Park in 1979 (E. J. Jackson et al., 2014). These management actions were the first of their kind in the WCR and may explain Bonaire's relatively higher parrotfish relative abundances in the early years. Among the regions included in this analysis, Bonaire and Roatan implemented a fishing ban on all parrotfishes in 2010 (Harms-Tuohy, 2021), which could explain slight upward trends in parrotfish relative abundance in these regions. Conversely, regions that implemented resource management later in time or have yet to implement resource management show relatively lower parrotfish relative abundances and minimal rebuilding, such as in Tortola. This variability in fishing pressure and fisheries management across the WCR is also effectively demonstrated in the regional relationship between parrotfish relative abundance and overfishing threat, where regions that have lower threat to overfishing exhibited a higher median relative abundance compared to regions with higher threats to overfishing (Figure 2.1B).

Global warming has resulted in several climatic variations, including increased sea surface temperatures (gradual increase in SST and increased frequency and intensity of marine heat waves), and hurricane intensity, and decreased dissolved oxygen and pH levels worldwide (IPCC, 2023). These changes significantly impact coral reefs health and resiliency (Hughes et al., 2017) and the fish communities that inhabit them (Alva-Basurto & Arias-González, 2014; Hughes et al., 2017). Moreover, the spatio-temporal variability in these environmental changes across the WCR (Dixon et al., 2022; P. J. Mumby, 2006; The Climate Studies Group Mona, 2020), likely contributes to the observed variations in relative abundance of parrotfishes across the various regions included in our study.

In addition to the above-mentioned anthropogenic stressors, parrotfish movement and larval dispersal distances are very limited (Cowen et al., 2006; Green et al., 2015). Consequently, connectivity in parrotfish populations across the WCR is restricted or effectively nonexistent (Cowen et al., 2006). This lack of connectivity in parrotfish populations across the WCR upholds the variability observed in relative abundance trends of parrotfishes across the WCR.

Relative abundance trends of parrotfish across the WCR by functional group

Like the cumulative parrotfish relative abundance trends, the relative abundance of parrotfishes aggregated by functional groups varied considerably within and across regions throughout the WCR. Irrespective of which paradigm is assumed to be true to explain the parrotfish-coral relationship (top-down regulation of benthos by parrotfish vs bottom-up regulation of parrotfish by benthos), the relative abundance trends of each functional group and how they relate to one another (winners and losers) reflect the state of the coral reef in each region (Russ et al., 2015). In fact, the co-occurrence of specific functional groups of herbivorous fishes (not only parrotfishes), in addition to functional diversity, has been shown to correlate with an improved (healthy) coral reef benthic state (Sheppard et al., 2023).

Based on the intermediate disturbance hypothesis, Sheppard et al. (2023) suggested excavating herbivores to be beneficial to coral richness and coral cover when at an intermediate proportional abundance. Excavators play a key role in bioerosion on coral reefs, exposing the carbonate framework and assisting in sediment production on reefs. Additionally, excavators aid in settlement of corals and coralline algae from the removal of crustose, endolithic algae, and microbes from within the reef framework (Adam et al., 2018; D. Bellwood & Choat, 1990). The pattern of intermediate abundances of excavators associated with healthier reefs was observed in Bonaire and Little Cayman, which are less degraded than other reefs across the WCR (Jackson et al. 2014; Slattery and Lesser 2019).

Conversely, Sheppard et al. (2023) revealed reefs dominated by one functional group were predicted to have lower levels of coral-related benthic metrics (coral richness, recruitment, cover, and calcification rate). In our study, Cozumel was the only region to exhibit a considerable long-term dominance of a single functional group – browsers - since 2007, suggesting the benthos habitat is dominated by macroalgae (Sheppard et al., 2023). Browsers graze on erect macroalgae and ultimately reducing macroalgae overgrowth and shading and consequently competition between corals and algae (Adam et al., 2018; D. R. Bellwood et al., 2004; Cardoso et al., 2009). The pattern exhibited between the dominance of a single functional group and lower coral richness and cover agrees with long-term benthic cover studies in Cozumel that note the overall decline in hard coral cover and increase in macroalgae cover through time (Contreras-Silva et al., 2020). In turn, this shift towards an algae-dominant reef creates favorable conditions for browsers to thrive.

In the face of climate change, the health and resiliency of coral reefs are under threat (Hughes et al., 2017). Anthropogenic stress has been shown to reduce coral coverage worldwide promote the ensuing phase shift from a coral-dominated to algae-dominated system (Good & Bahr, 2021). These environmental changes have been shown to alter fish communities (Wilson et al., 2006) and from our study it's clear winners and losers can emerge when environmental conditions are favorable to specific functional groups. In algal-dominated coral reef ecosystems, browsers may begin to dominate across coral reefs, like that seen in Cozumel.

Synchrony within and across functional groups

Our results indicated synchrony between individual parrotfish species as generally positive (proportion of interspecies comparisons and median group correlation across interspecies comparisons) within the same functional group in most regions across the WCR. These findings support the notion that regional ecological processes, whether that be recruitment and/or top-down/bottom-up processes, have a similar impact on parrotfish species within a functional group (quadrant A). Consequently, any signal of competition in relative abundance trends is negligible and swamped by these ecological processes. Synchrony across functional groups was less prevalent than synchrony within functional groups and varied considerably across regions. For example, in Key Largo where the scraper-browser comparison was the only positive functional group comparison, clear winners and losers may ensue depending on the underlying ecological processes occurring in the region and which functional group they favor (quadrant C). This differs from regions like Bonaire, Little Cayman, and Tortola where all three functional group comparisons exhibited a positive median correlation and thus the impact of shifts in habitat and ecology is unform across functional groups (quadrant D).

Distinguishing the primary ecological process most influential in shaping synchrony among parrotfish populations across the WCR is challenging. Rather, it's likely a combination of top-down, bottom-up processes and recruitment processes, that collectively shape parrotfish population dynamics within and across functional groups. The predominant top-down forcing on parrotfish abundance is fishing. While some variations in fishing pressure exist (World Resources Institute, 2004), it has generally led to the depletion of parrotfishes across the WCR. Conversely, bottom-up processes have a considerable influence on the coral reef health and the parrotfishes that inhabit them, and this impact has been shown to vary in time and space. While Caribbean coral reefs are generally thought to be less oligotrophic than other coral reef ecosystems (Miloslavich et al., 2010), human population growth and coastal development across the WCR has increased nutrition pollution, disrupting coral reef function and growth (Silbiger et al., 2018). Climate change impacts further exacerbate these changes in bottom-up processes in varying degrees across the WCR, which can have variable effects on parrotfishes, decoupling their dynamics when conditions prove more favorable for particular functional groups.

Finally, recruitment contributes to parrotfish population dynamics within and across functional groups. While recruitment is influenced by both top-down and bottom-up processes, many of which discussed above (Hare, 2014), Valles et al. (2009) revealed *Scarus* spp. recruitment and settlement were influenced by differing environmental factors than that of *Sparisoma* spp., suggesting preferential environmental conditions to promote recruitment may exist across parrotfishes which could also explain the decoupling of parrotfish dynamics in particular regions, such as Cozumel.

Parrotfishes play an important role in maintaining coral-dominated reef habitats. Consequently, much of the literature asserts their decline potentially threatens coral reef resiliency and their recovery is needed to recovery coral reefs. However, our study clearly indicates bottom-up processes drive parrotfish population dynamics in some regions more than others. While studies have advocated for protecting herbivorous reef fish to enhance coral resilience to climate change (Hughes et al., 2007; Kennedy et al., 2013; P. J. Mumby et al., 2006; Rasher et al., 2012), Carassou et al. (2013) suggested that the exclusive protection of herbivores may not effectively increase coral resilience especially when facing large-scale climatic disturbances. Our findings align more closely with this suggestion as we have demonstrated that bottom-up processes have a strong influence on parrotfish population dynamics. While our results don't conclusively prove this theory, they are certainly consistent with it. Therefore, management of herbivorous reef fishes may not necessarily promote a top-down control between parrotfish and Caribbean coral reefs.

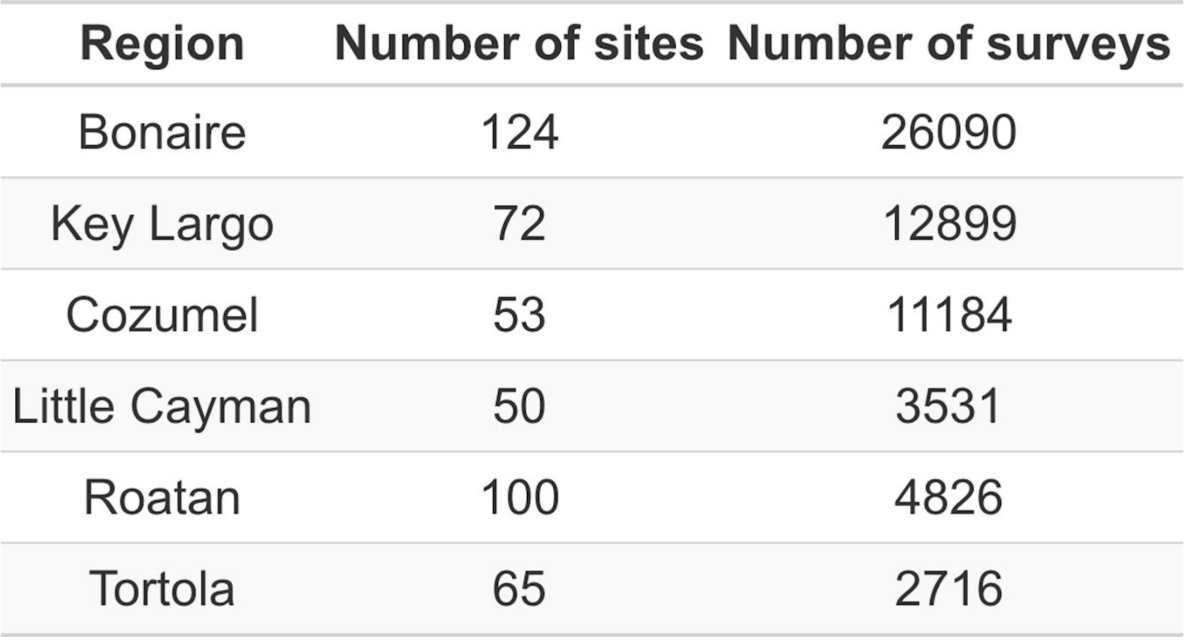
Conservation implications

The functional diversity within herbivorous reef fishes is widely understood to support coral reef ecosystem function (Adam et al., 2018; Brandl et al., 2019; Cardoso et al., 2009; Cheal et al., 2013; Cramer et al., 2017). While the exclusive protection of Caribbean parrotfishes may not effectively recover coral reefs in the WCR in the face of large-scale climatic disturbances, it is still essential. Historically fisheries management aimed at protecting parrotfish across the WCR was typically implemented at the family level (AIDA, 2021). However, in some cases such an approach may be overly simplistic, given the functional diversity across parrotfishes (Supplemental Table 2.3) within a region. Our results demonstrate that while there may be strong positively correlated synchrony within a given functional group, the synchrony across functional groups varied in sign across functional group comparisons and regions. Consequently, generic conservation efforts (e.g., ban on all parrotfishes) may not benefit parrotfish equally across functional groups within a given region, especially when variable recruitment and/or trophic processes impact functional groups differently. Recognizing that the synchrony in parrotfish across functional groups is region-specific, the implementation of regional fisheries management measures at the functional group level, rather than the family level, may be a more effective approach. Adopting a more precise fisheries management approach that corresponds to community level dynamics (e.g., species- or functional group-specific bag limits or bans) will encourage recovery across all functional groups (avoiding 'winners' and 'losers'). This approach has been adopted by some UN member states, including Columbia and Cuba (AIDA, 2021). Conversely, this resolution of management may be challenging, especially for countries that have inadequate enforcement and compliance of management measures and challenges in parrotfish species identification amongst fishers (Harms-Tuohy, 2021). In this case, more generic management approaches may be adopted (e.g., banning of all parrotfishes), however, it's important to recognize that these generic conservation measure may not uniformly benefit all parrotfishes. Consequently, managers should anticipate some functional groups may respond more readily to these conservation efforts than others.

Acknowledgements

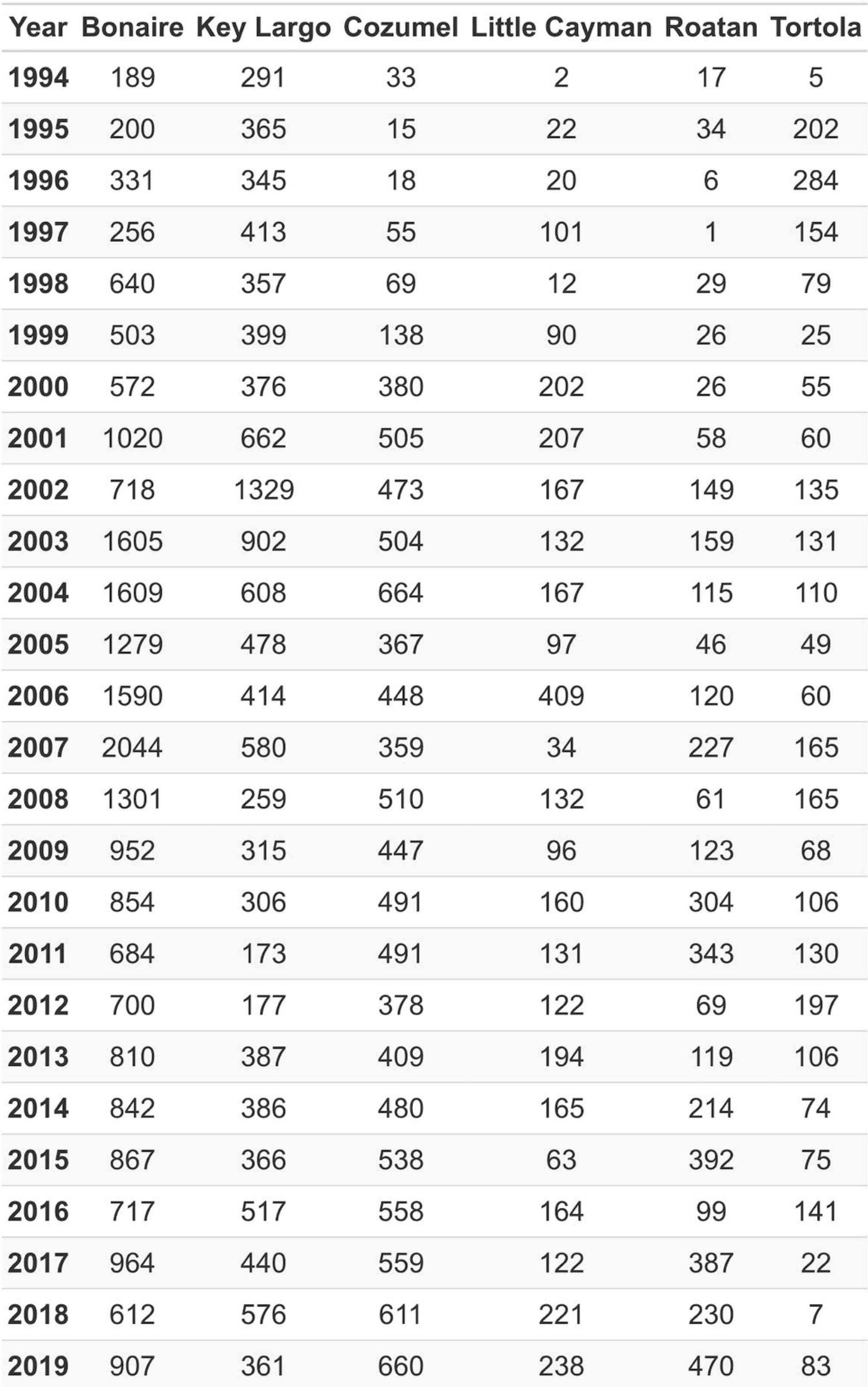
Chapter 2, in part is currently being prepared for submission for publication of the material. Greenberg, Dan A.; Semmens, Brice X. The dissertation author was the primary researcher and author of this material.

Supplemental Table 2.1: Total number of REEF sites and surveys associated with each region across the WCR.

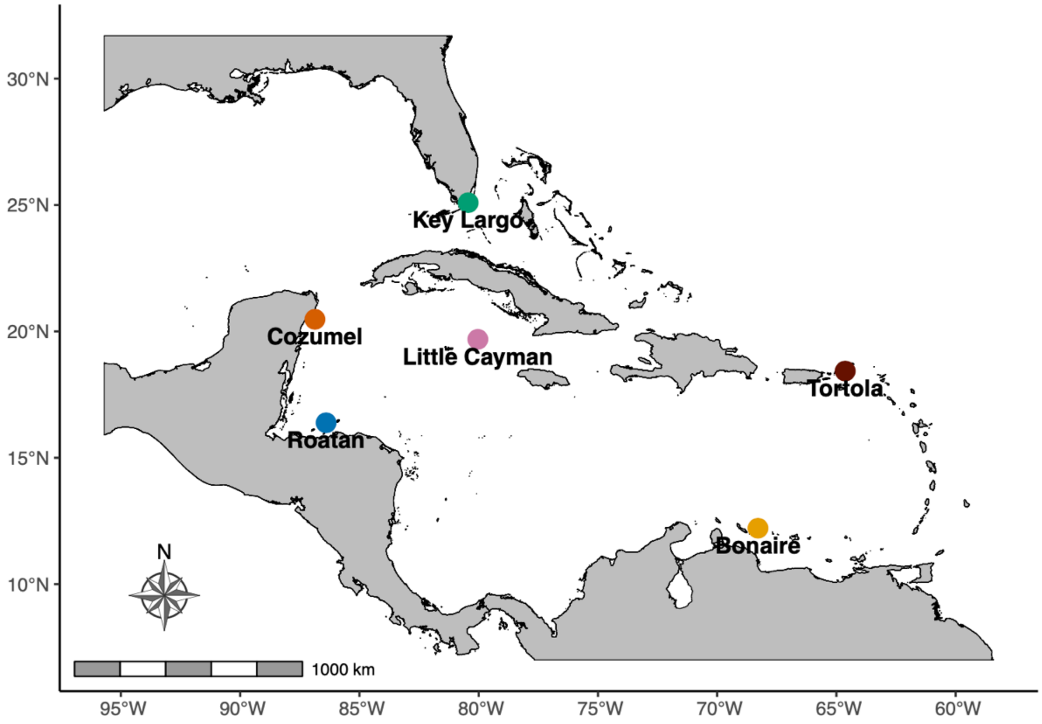


Region	Number of sites	Number of surveys
Bonaire	124	26090
Key Largo	72	12899
Cozumel	53	11184
Little Cayman	50	3531
Roatan	100	4826
Tortola	65	2716

Supplemental Table 2.2: Annual number of REEF surveys conducted in each region across the WCR.

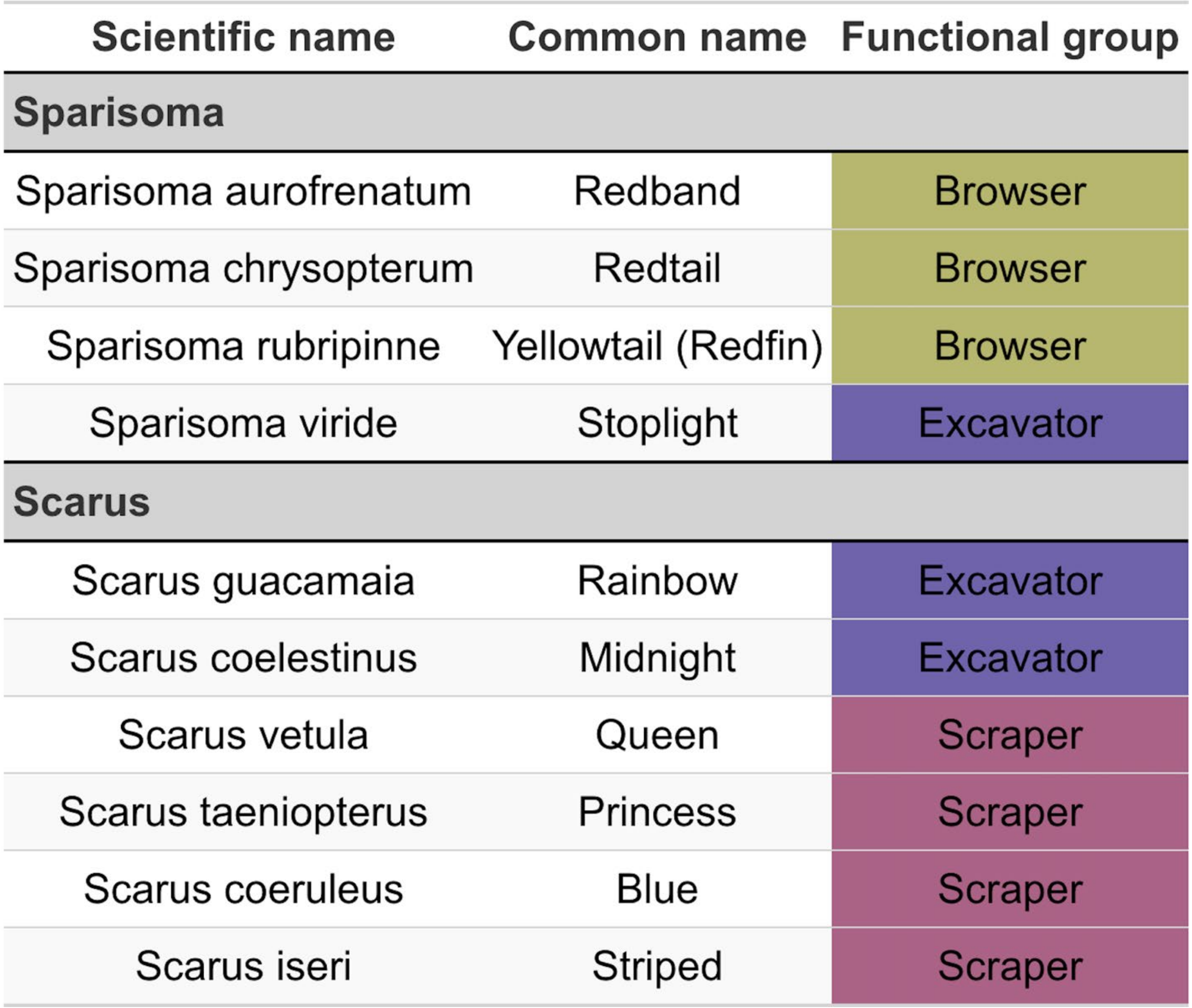


Year	Bonaire	Key Largo	Cozumel	Little Cayman	Roatan	Tortola
1994	189	291	33	2	17	5
1995	200	365	15	22	34	202
1996	331	345	18	20	6	284
1997	256	413	55	101	1	154
1998	640	357	69	12	29	79
1999	503	399	138	90	26	25
2000	572	376	380	202	26	55
2001	1020	662	505	207	58	60
2002	718	1329	473	167	149	135
2003	1605	902	504	132	159	131
2004	1609	608	664	167	115	110
2005	1279	478	367	97	46	49
2006	1590	414	448	409	120	60
2007	2044	580	359	34	227	165
2008	1301	259	510	132	61	165
2009	952	315	447	96	123	68
2010	854	306	491	160	304	106
2011	684	173	491	131	343	130
2012	700	177	378	122	69	197
2013	810	387	409	194	119	106
2014	842	386	480	165	214	74
2015	867	366	538	63	392	75
2016	717	517	558	164	99	141
2017	964	440	559	122	387	22
2018	612	576	611	221	230	7
2019	907	361	660	238	470	83



Supplemental Figure 2.1: Map of the wider Caribbean region (WCR). REEF surveys conducted in each of the six regions were included in the analysis.

Supplemental Table 2.3: Ten parrotfish species included in the analysis identified by scientific name, common name, and functional group.



Scientific name	Common name	Functional group
Sparisoma		
Sparisoma aurofrenatum	Redband	Browser
Sparisoma chrysopterum	Redtail	Browser
Sparisoma rubripinne	Yellowtail (Redfin)	Browser
Sparisoma viride	Stoplight	Excavator
Scarus		
Scarus guacamaia	Rainbow	Excavator
Scarus coelestinus	Midnight	Excavator
Scarus vetula	Queen	Scraper
Scarus taeniopterus	Princess	Scraper
Scarus coeruleus	Blue	Scraper
Scarus iseri	Striped	Scraper

Supplemental Table 2.4: Full model specifications.

$$\boldsymbol{Ordered\ Logistic(y_i|\eta_{t,s,dv,dmy,i}, c_{1-K-1}}$$

$$\begin{cases} \boldsymbol{logit^{-1}(c_1 - \eta_i)} & \boldsymbol{Y = 0} \\ \boldsymbol{logit^{-1}(c_2 - \eta_i) - logit^{-1}(c_1 - \eta_i)} & \boldsymbol{Y = 1} \\ \boldsymbol{logit^{-1}(c_3 - \eta_i) - logit^{-1}(c_2 - \eta_i)} & \boldsymbol{Y = 2 - 10} \\ \boldsymbol{logit^{-1}(c_4 - \eta_i) - logit^{-1}(c_3 - \eta_i)} & \boldsymbol{Y = 11 - 100} \\ \boldsymbol{1 - logit^{-1}(c_4 - \eta_i)} & \boldsymbol{Y > 100} \end{cases}$$

$$c_{1:K-1} \sim InducedDirichlet(\alpha_{1:K} = 1)$$

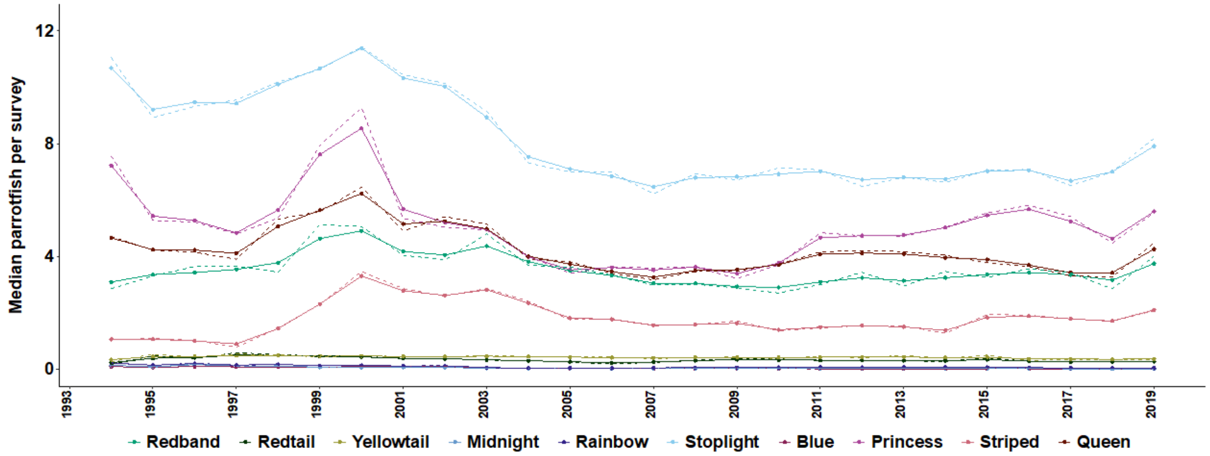
$$\boldsymbol{\eta_{t,s,dv,dmy,i} = \alpha_{n,t} + z_s * \sigma_s + z_{dv} * \sigma_{dv} + z_{dmy} * \sigma_{dmy} + X_i}$$

$$\begin{bmatrix} x_{1,t} \\ \cdots \\ x_{m,t} \end{bmatrix} = \begin{bmatrix} x_{1,t-1} + w_{1,t} \\ \cdots \\ x_{m,t-1} + w_{m,t} \end{bmatrix} \qquad \begin{bmatrix} w_{1,t} \\ \cdots \\ w_{m,t} \end{bmatrix} \sim MVNormal\left(\begin{bmatrix} 0 \\ 0 \\ 0 \end{bmatrix}, Q \right) \qquad Q = \begin{bmatrix} \sigma_1^2 & \cdots & \rho_{1,m} \\ \vdots & \ddots & \vdots \\ \rho_{1,m} & \cdots & \sigma_m^2 \end{bmatrix}\dagger$$

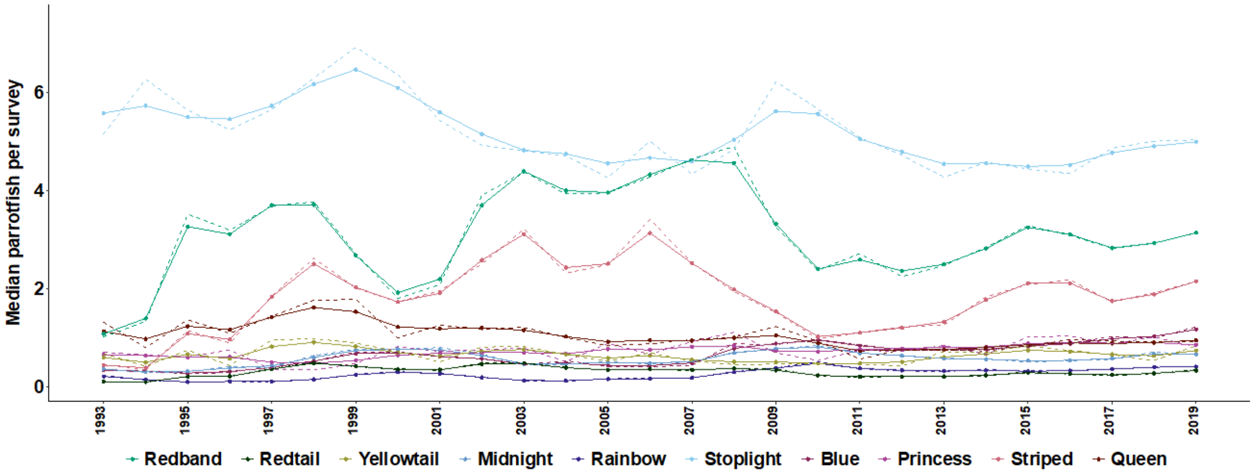
$$\begin{bmatrix} \alpha_{1,t} \\ \cdots \\ \alpha_{n,t} \end{bmatrix} = \begin{bmatrix} x_{1,t} + v_{1,t} \\ \cdots \\ x_{n,t} + v_{n,t} \end{bmatrix} \qquad \begin{matrix} v_{1,t} \sim N(0, \tau_1^2) \\ \cdots \\ v_{n,t} \sim N(0, \tau_n^2) \end{matrix}$$

$\boldsymbol{\sigma_{1:m}} \sim \boldsymbol{Gamma}(\alpha = 3, \beta = 0.5)$
$\boldsymbol{\tau_{1:m}} \sim \boldsymbol{Gamma}(\alpha = 3, \beta = 0.5)$

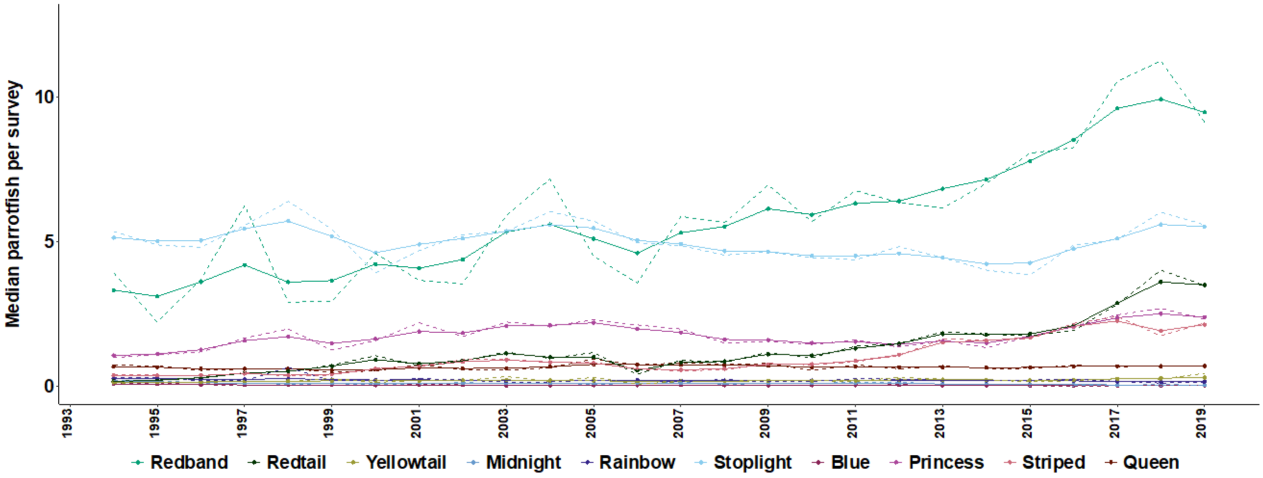
$z_s \sim N(0,1)$ [site effect deviate]
$z_{dv} \sim N(0,1)$ [diver effect deviate]
$z_{dmy} \sim N(0,1)$ [survey day of year effect deviate]
$\boldsymbol{\sigma_s, \sigma_{dv}, \sigma_{dmy} \sim Gamma(\alpha = 4, \beta = 2)}$
$\boldsymbol{x_{1:m,0} \sim N(0, 5)}$
†correlation matrix estimated from lower-triangular Cholesky factor, L ~ LKJ(1.0)



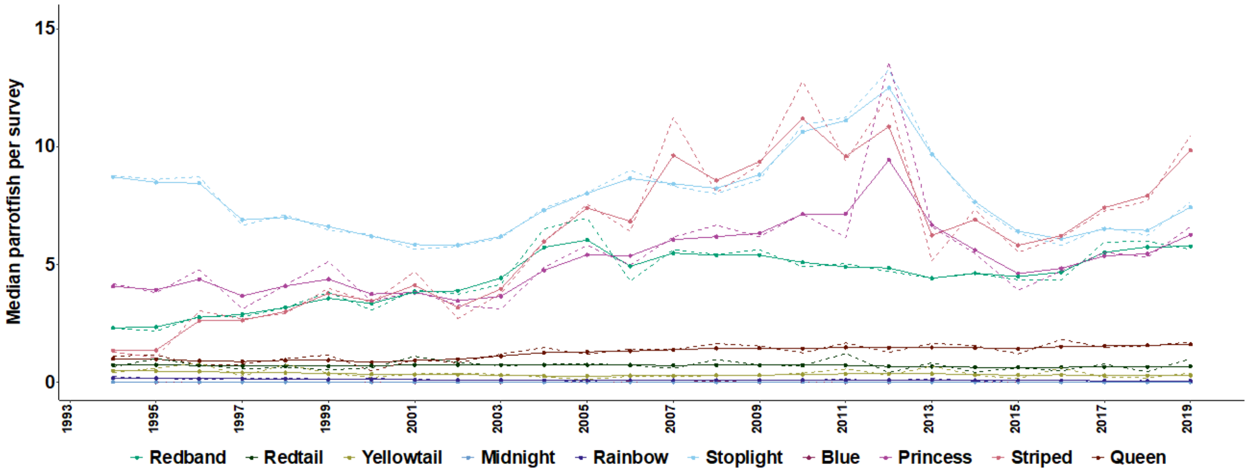
Supplemental Figure 2.2: Species-specific relative abundance of parrotfishes from 1994 to 2019 in Bonaire. Trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated model iterations for a given species. The solid line represents state process while the dashed line represents observation process for each species.



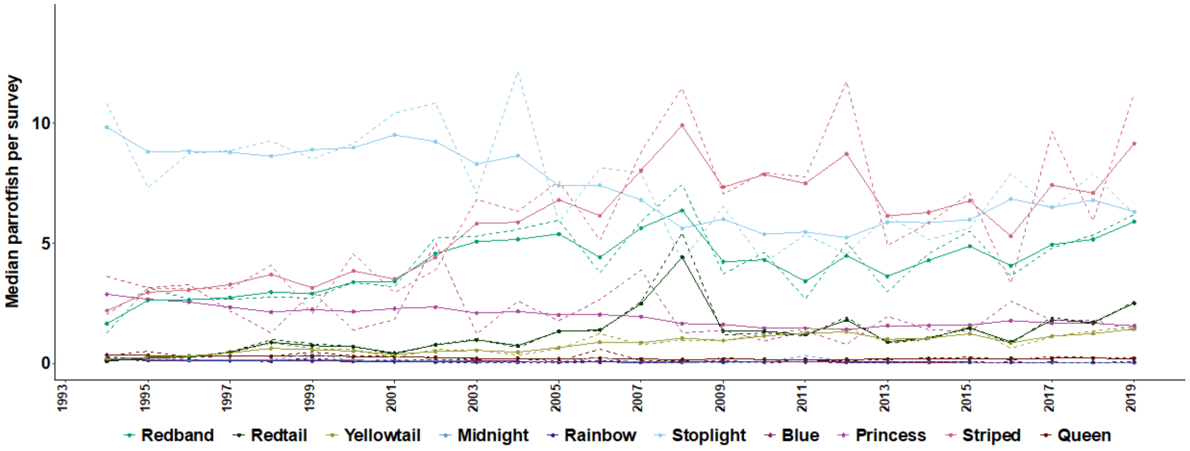
Supplemental Figure 2.3: Species-specific relative abundance of parrotfishes from 1994 to 2019 in Key Largo. Trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated model iterations for a given species. The solid line represents state process while the dashed line represents observation process for each species.



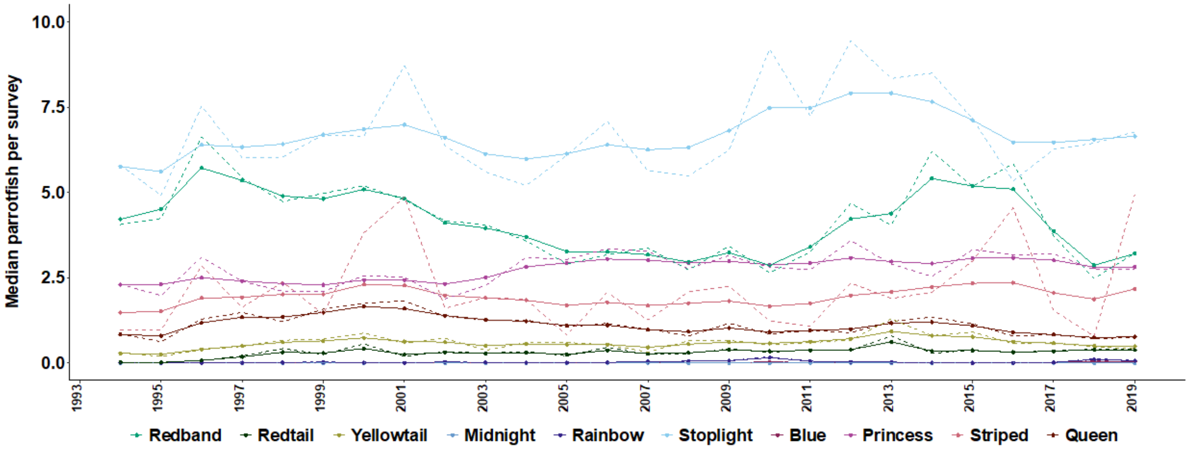
Supplemental Figure 2.4: Species-specific relative abundance of parrotfishes from 1994 to 2019 in Cozumel. Trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated model iterations for a given species. The solid line represents state process while the dashed line represents observation process for each species.



Supplemental Figure 2.5: Species-specific relative abundance of parrotfishes from 1994 to 2019 in Little Cayman. Trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated model iterations for a given species. The solid line represents state process while the dashed line represents observation process for each species.

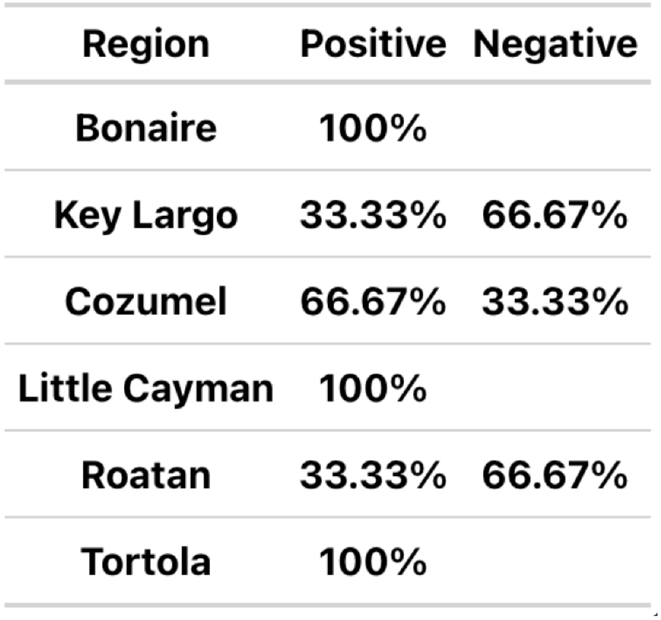


Supplemental Figure 2.6: Species-specific relative abundance of parrotfishes from 1994 to 2019 in Roatan. Trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated model iterations for a given species. The solid line represents state process while the dashed line represents observation process for each species.



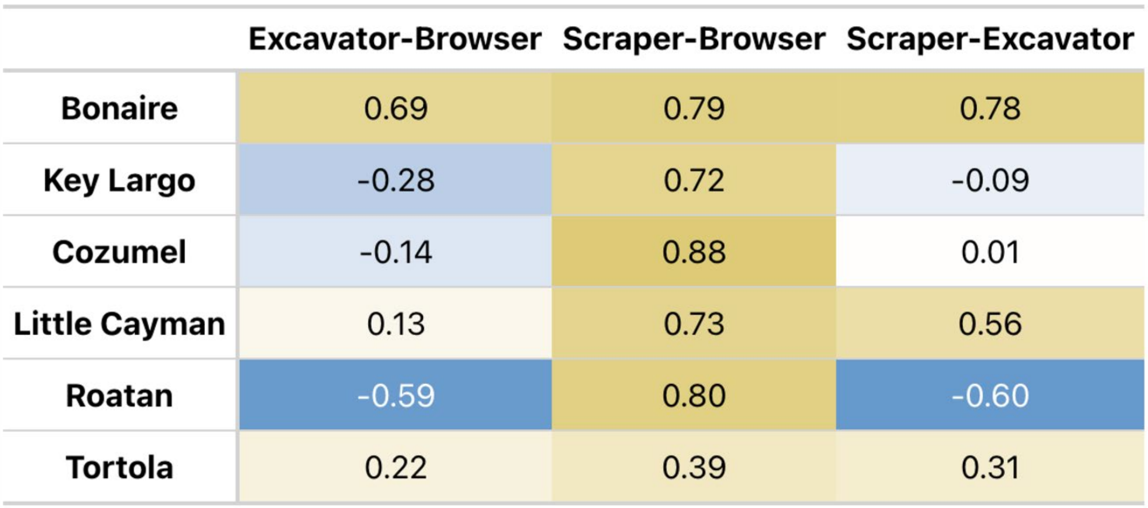
Supplemental Figure 2.7: Species-specific relative abundance of parrotfishes from 1994 to 2019 in Tortola. Trends represent median posterior estimates and shading represent 90% credible intervals of posterior estimates aggregated model iterations for a given species. The solid line represents state process while the dashed line represents observation process for each species.

Supplemental Table 2.5: Proportion of positive and negative functional group comparisons for each region. These proportions are calculated using derived correlation between the posterior estimates of the cumulative relative abundance trends of each functional group comparison.

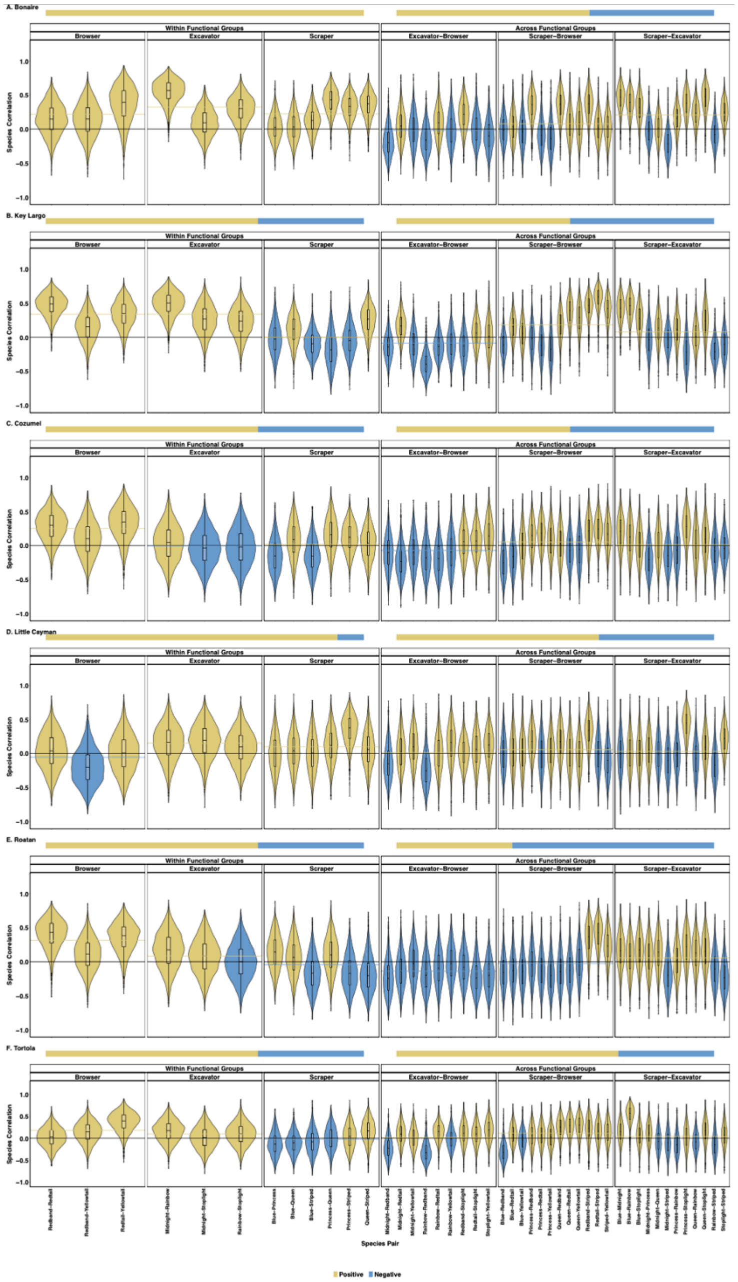


Region	Positive	Negative
Bonaire	100%	
Key Largo	33.33%	66.67%
Cozumel	66.67%	33.33%
Little Cayman	100%	
Roatan	33.33%	66.67%
Tortola	100%	

Supplemental Table 2.6: Median correlation between each functional group comparisons by region. The color of the cell denotes sign and degree of correlations, where yellow hues denote positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.

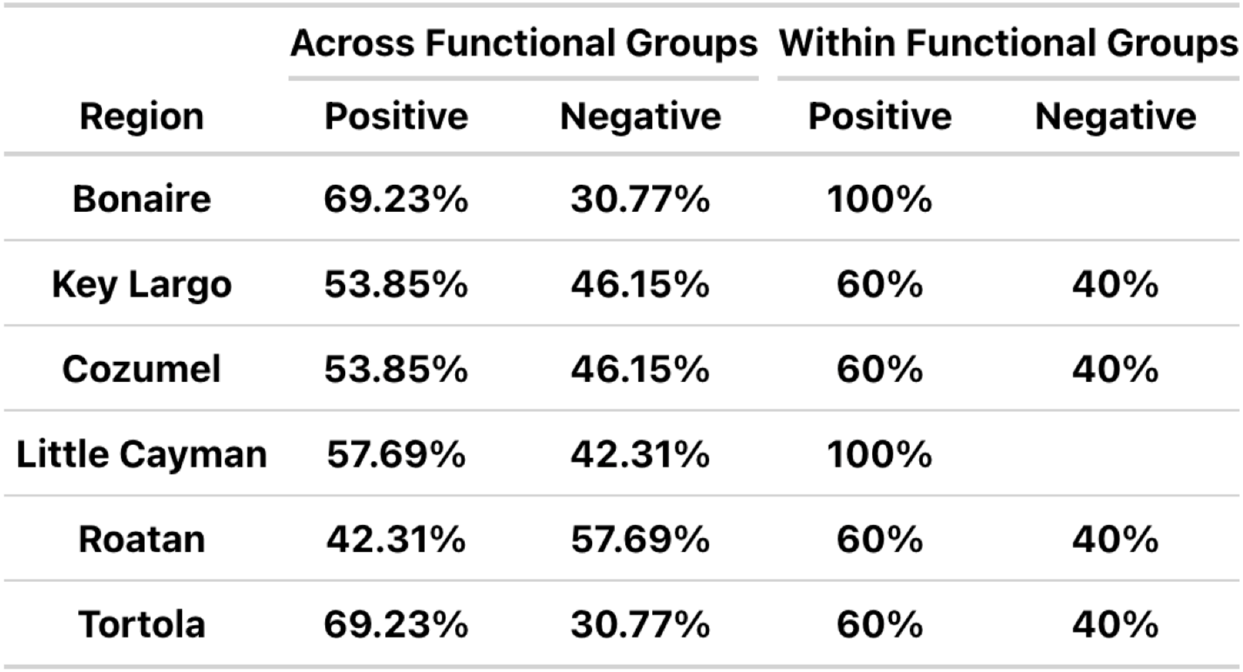


	Excavator-Browser	Scraper-Browser	Scraper-Excavator
Bonaire	0.69	0.79	0.78
Key Largo	-0.28	0.72	-0.09
Cozumel	-0.14	0.88	0.01
Little Cayman	0.13	0.73	0.56
Roatan	-0.59	0.80	-0.60
Tortola	0.22	0.39	0.31



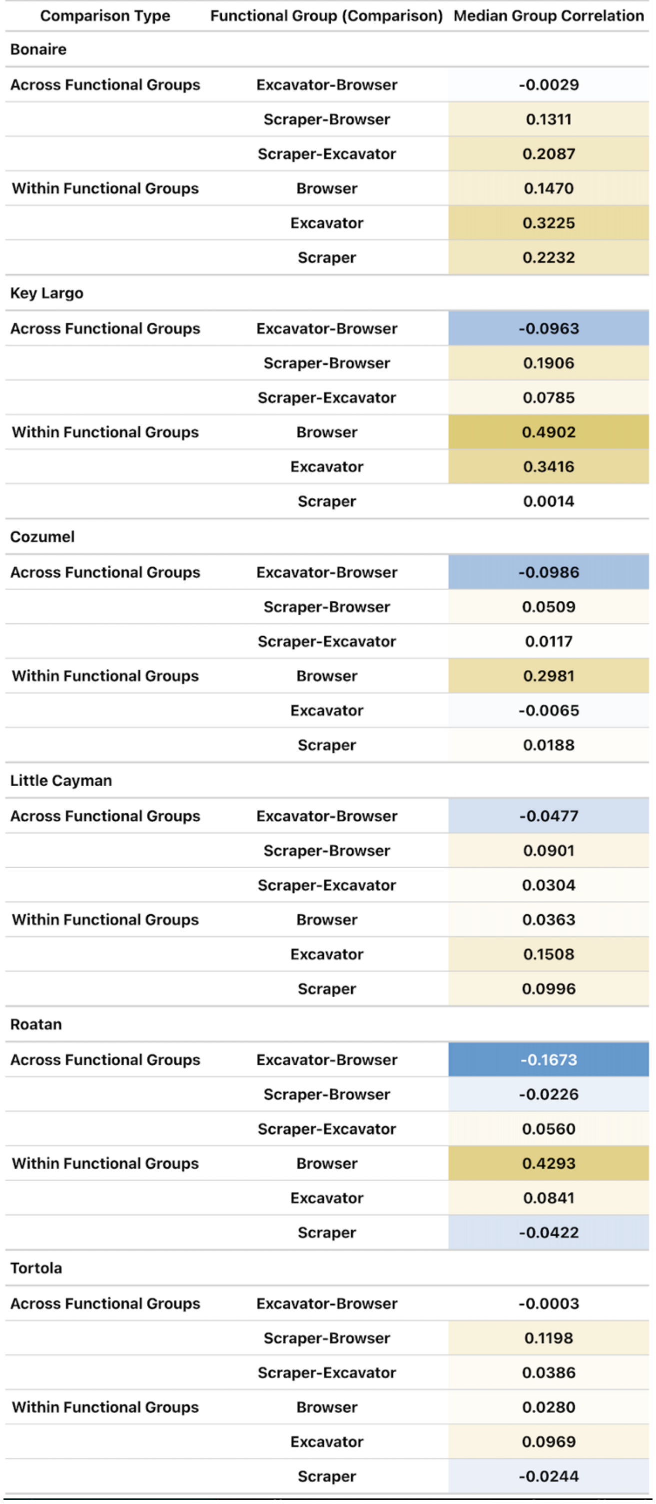
Supplemental Figure 2.8: Correlation between modeled parrotfish abundance trends representative of each interspecies comparison within and across functional groups in Bonaire (A), Cozumel (B), Key Largo (C), Little Cayman (D), Roatan (E), Tortola (F). The correlation of each interspecies comparison is represented as violin plots depicting distribution of posterior estimates from the variance-covariance matrix. The colors denote the sign of the median correlation for a given interspecies comparison, where yellow denotes positive, and blue denotes negative correlations. The horizontal lines in each panel represent median group correlation across all interspecies comparisons and the horizontal bars at the top of each panel denote the proportion of positive and negative interspecies comparisons within and across functional group (comparison).

Supplemental Table 2.7: Proportion of positive and negative interspecies comparisons across and within functional groups for each region. These proportions are calculated using posterior estimates of interspecies correlations.



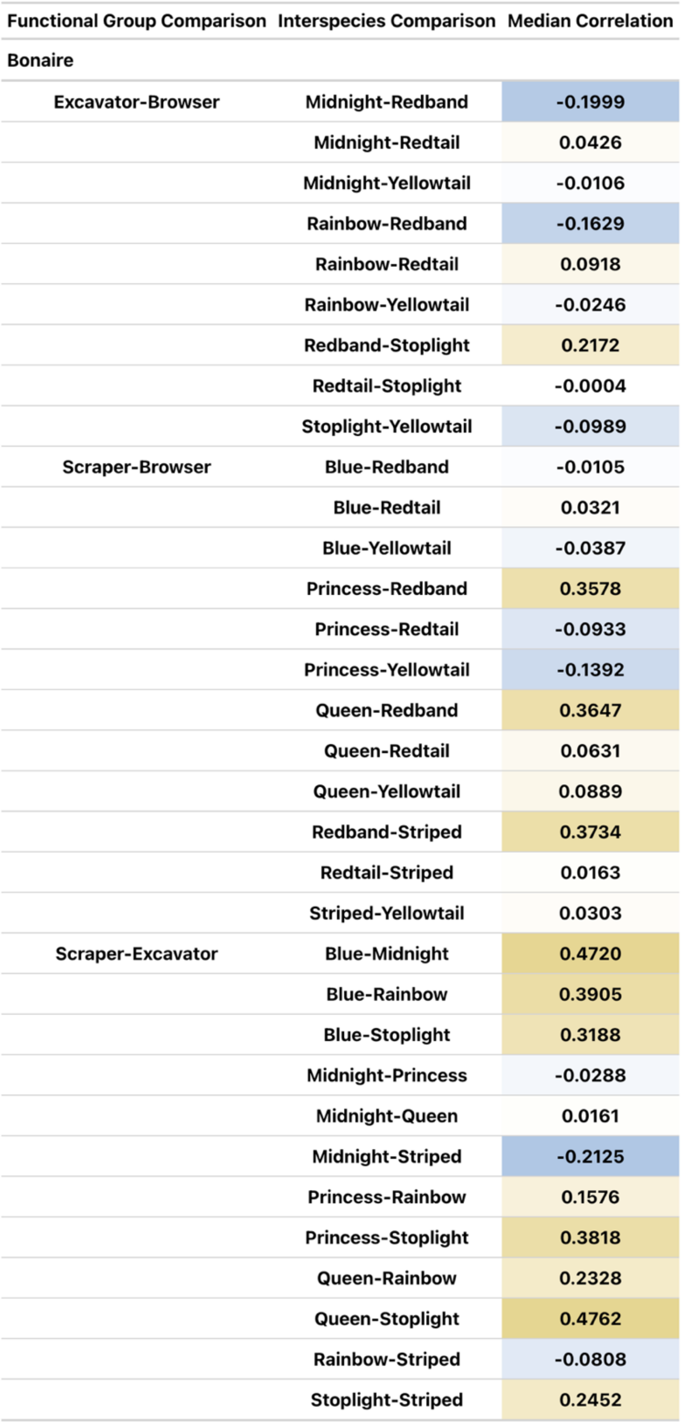
	Across Functional Groups		Within Functional Groups	
Region	**Positive**	**Negative**	**Positive**	**Negative**
Bonaire	**69.23%**	**30.77%**	**100%**	
Key Largo	**53.85%**	**46.15%**	**60%**	**40%**
Cozumel	**53.85%**	**46.15%**	**60%**	**40%**
Little Cayman	**57.69%**	**42.31%**	**100%**	
Roatan	**42.31%**	**57.69%**	**60%**	**40%**
Tortola	**69.23%**	**30.77%**	**60%**	**40%**

Supplemental Table 2.8: Median group correlations within and across functional groups by region. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation. The median group correlation represents the median correlation across all interspecies comparisons within in the given functional group (comparison).



Comparison Type	Functional Group (Comparison)	Median Group Correlation
Bonaire		
Across Functional Groups	Excavator-Browser	-0.0029
	Scraper-Browser	0.1311
	Scraper-Excavator	0.2087
Within Functional Groups	Browser	0.1470
	Excavator	0.3225
	Scraper	0.2232
Key Largo		
Across Functional Groups	Excavator-Browser	-0.0963
	Scraper-Browser	0.1906
	Scraper-Excavator	0.0785
Within Functional Groups	Browser	0.4902
	Excavator	0.3416
	Scraper	0.0014
Cozumel		
Across Functional Groups	Excavator-Browser	-0.0986
	Scraper-Browser	0.0509
	Scraper-Excavator	0.0117
Within Functional Groups	Browser	0.2981
	Excavator	-0.0065
	Scraper	0.0188
Little Cayman		
Across Functional Groups	Excavator-Browser	-0.0477
	Scraper-Browser	0.0901
	Scraper-Excavator	0.0304
Within Functional Groups	Browser	0.0363
	Excavator	0.1508
	Scraper	0.0996
Roatan		
Across Functional Groups	Excavator-Browser	-0.1673
	Scraper-Browser	-0.0226
	Scraper-Excavator	0.0560
Within Functional Groups	Browser	0.4293
	Excavator	0.0841
	Scraper	-0.0422
Tortola		
Across Functional Groups	Excavator-Browser	-0.0003
	Scraper-Browser	0.1198
	Scraper-Excavator	0.0386
Within Functional Groups	Browser	0.0280
	Excavator	0.0969
	Scraper	-0.0244

Supplemental Table 2.9: Median correlation for each interspecies comparison across functional groups in Bonaire. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



Functional Group Comparison	Interspecies Comparison	Median Correlation
Bonaire		
Excavator-Browser	Midnight-Redband	-0.1999
	Midnight-Redtail	0.0426
	Midnight-Yellowtail	-0.0106
	Rainbow-Redband	-0.1629
	Rainbow-Redtail	0.0918
	Rainbow-Yellowtail	-0.0246
	Redband-Stoplight	0.2172
	Redtail-Stoplight	-0.0004
	Stoplight-Yellowtail	-0.0989
Scraper-Browser	Blue-Redband	-0.0105
	Blue-Redtail	0.0321
	Blue-Yellowtail	-0.0387
	Princess-Redband	0.3578
	Princess-Redtail	-0.0933
	Princess-Yellowtail	-0.1392
	Queen-Redband	0.3647
	Queen-Redtail	0.0631
	Queen-Yellowtail	0.0889
	Redband-Striped	0.3734
	Redtail-Striped	0.0163
	Striped-Yellowtail	0.0303
Scraper-Excavator	Blue-Midnight	0.4720
	Blue-Rainbow	0.3905
	Blue-Stoplight	0.3188
	Midnight-Princess	-0.0288
	Midnight-Queen	0.0161
	Midnight-Striped	-0.2125
	Princess-Rainbow	0.1576
	Princess-Stoplight	0.3818
	Queen-Rainbow	0.2328
	Queen-Stoplight	0.4762
	Rainbow-Striped	-0.0808
	Stoplight-Striped	0.2452

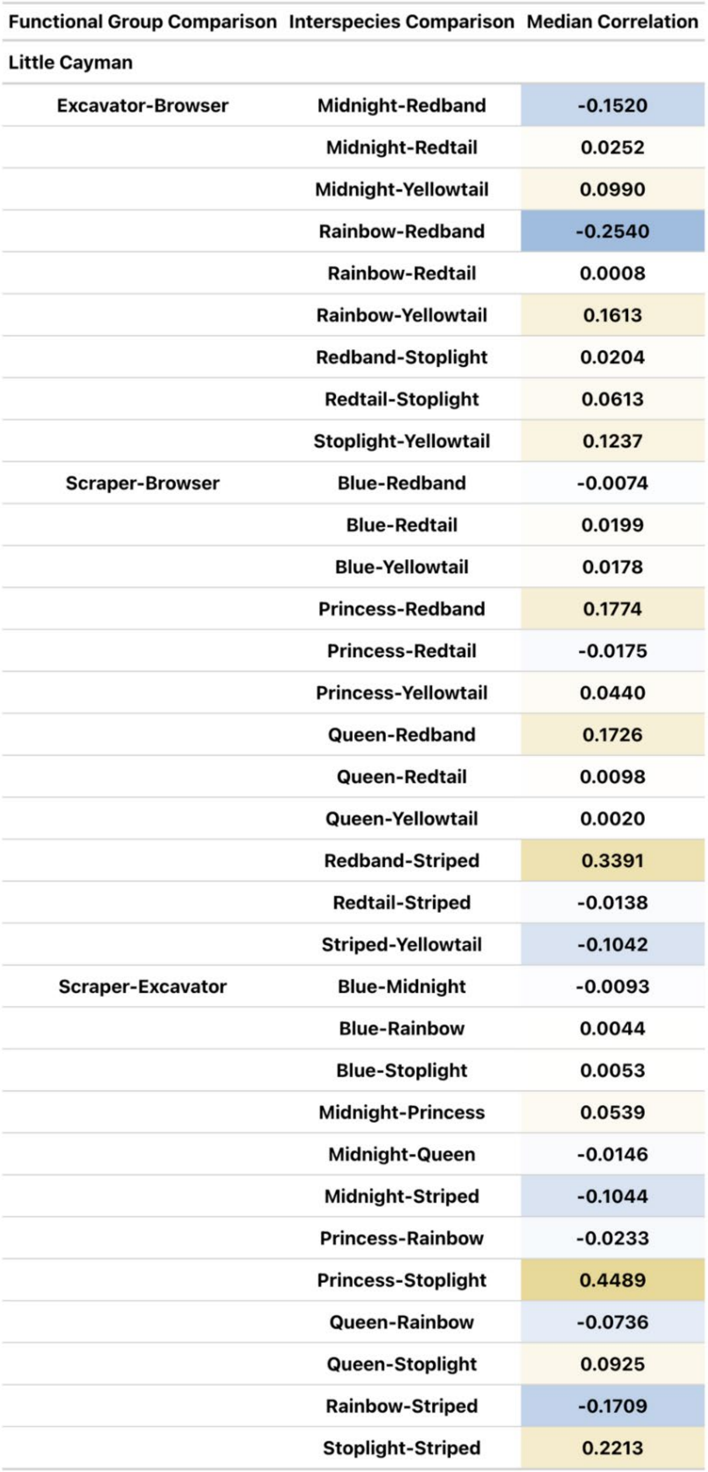
Supplemental Table 2.10: Median correlation for each interspecies comparison across functional groups in Cozumel. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.

Functional Group Comparison	Interspecies Comparison	Median Correlation
Cozumel		
Excavator-Browser	Midnight-Redband	-0.1042
	Midnight-Redtail	-0.2313
	Midnight-Yellowtail	-0.1002
	Rainbow-Redband	-0.1798
	Rainbow-Redtail	-0.1923
	Rainbow-Yellowtail	-0.0937
	Redband-Stoplight	0.0586
	Redtail-Stoplight	0.0661
	Stoplight-Yellowtail	0.1387
Scraper-Browser	Blue-Redband	-0.2536
	Blue-Redtail	-0.1622
	Blue-Yellowtail	0.0010
	Princess-Redband	0.1429
	Princess-Redtail	0.1760
	Princess-Yellowtail	0.0719
	Queen-Redband	0.0290
	Queen-Redtail	-0.0781
	Queen-Yellowtail	-0.0567
	Redband-Striped	0.2141
	Redtail-Striped	0.2422
	Striped-Yellowtail	0.1667
Scraper-Excavator	Blue-Midnight	0.2088
	Blue-Rainbow	0.1264
	Blue-Stoplight	0.0108
	Midnight-Princess	-0.2078
	Midnight-Queen	0.0131
	Midnight-Striped	-0.1503
	Princess-Rainbow	-0.0671
	Princess-Stoplight	0.2803
	Queen-Rainbow	0.0234
	Queen-Stoplight	0.0919
	Rainbow-Striped	-0.0808
	Stoplight-Striped	-0.0426

Supplemental Table 2.11: Median correlation for each interspecies comparison across functional groups in Key Largo. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.

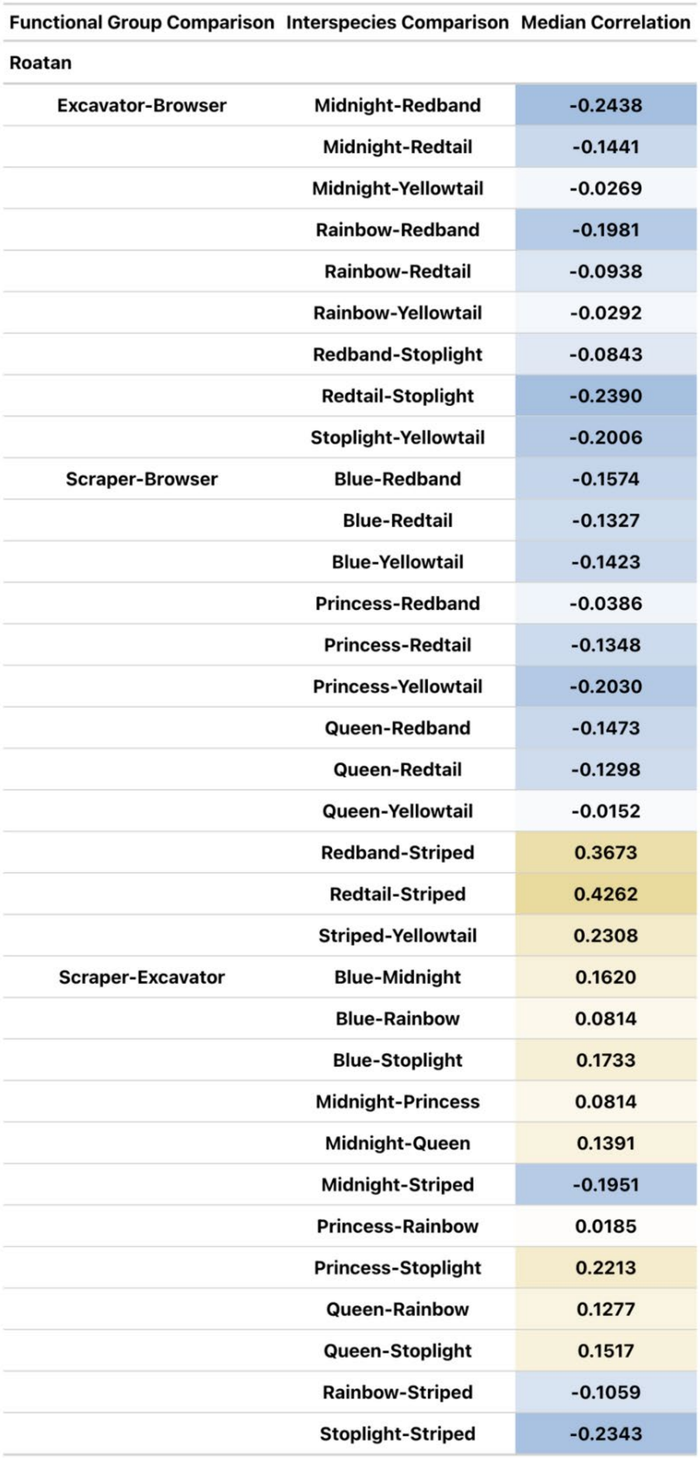
Functional Group Comparison	Interspecies Comparison	Median Correlation
Key Largo		
Excavator-Browser	Midnight-Redband	-0.1410
	Midnight-Redtail	0.1675
	Midnight-Yellowtail	-0.1080
	Rainbow-Redband	-0.3968
	Rainbow-Redtail	-0.1366
	Rainbow-Yellowtail	-0.1024
	Redband-Stoplight	-0.1273
	Redtail-Stoplight	0.0628
	Stoplight-Yellowtail	0.0101
Scraper-Browser	Blue-Redband	-0.1116
	Blue-Redtail	0.1542
	Blue-Yellowtail	0.0560
	Princess-Redband	-0.0185
	Princess-Redtail	-0.1192
	Princess-Yellowtail	-0.1753
	Queen-Redband	0.1895
	Queen-Redtail	0.3940
	Queen-Yellowtail	0.2926
	Redband-Striped	0.4681
	Redtail-Striped	0.5956
	Striped-Yellowtail	0.4239
Scraper-Excavator	Blue-Midnight	0.4243
	Blue-Rainbow	0.4627
	Blue-Stoplight	0.2769
	Midnight-Princess	-0.0373
	Midnight-Queen	0.1345
	Midnight-Striped	-0.0275
	Princess-Rainbow	0.1194
	Princess-Stoplight	-0.2218
	Queen-Rainbow	0.0105
	Queen-Stoplight	0.2403
	Rainbow-Striped	-0.2098
	Stoplight-Striped	-0.1061

Supplemental Table 2.12: Median correlation for each interspecies comparison across functional groups in Little Cayman. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



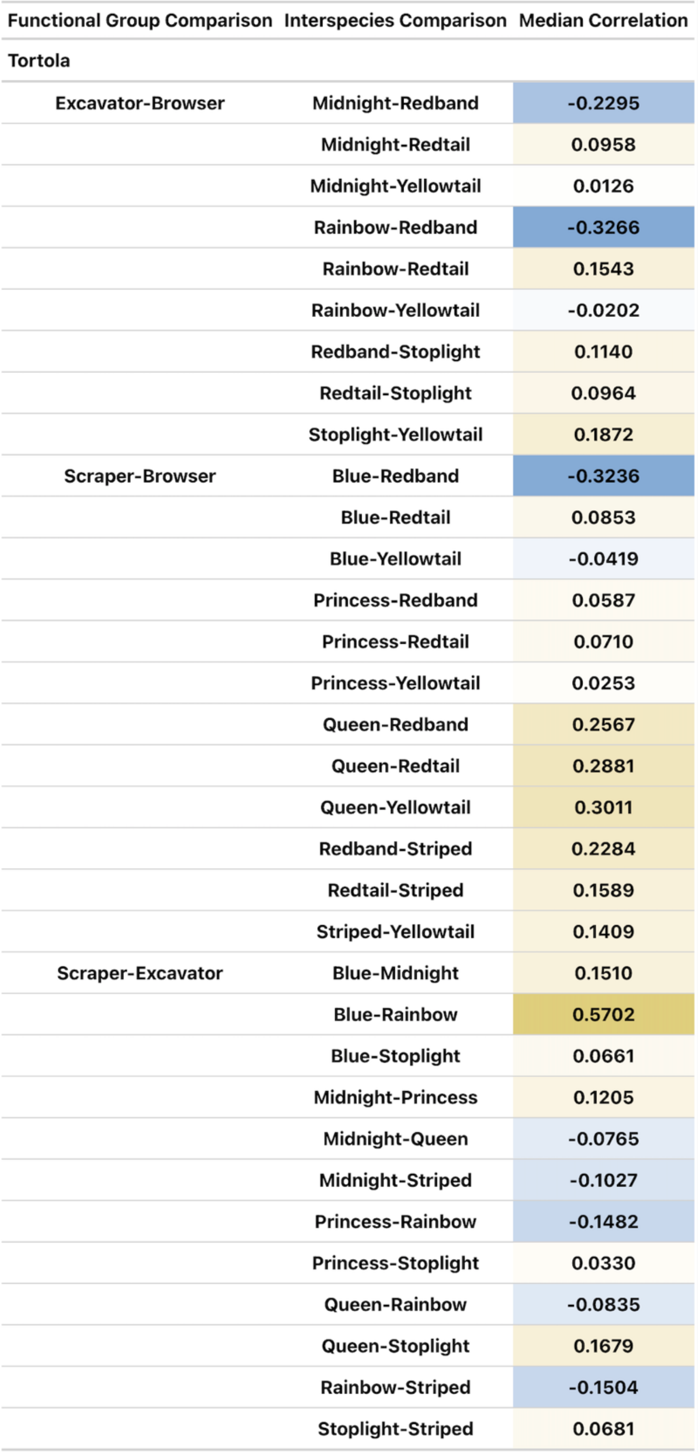
Functional Group Comparison	Interspecies Comparison	Median Correlation
Little Cayman		
Excavator-Browser	Midnight-Redband	-0.1520
	Midnight-Redtail	0.0252
	Midnight-Yellowtail	0.0990
	Rainbow-Redband	-0.2540
	Rainbow-Redtail	0.0008
	Rainbow-Yellowtail	0.1613
	Redband-Stoplight	0.0204
	Redtail-Stoplight	0.0613
	Stoplight-Yellowtail	0.1237
Scraper-Browser	Blue-Redband	-0.0074
	Blue-Redtail	0.0199
	Blue-Yellowtail	0.0178
	Princess-Redband	0.1774
	Princess-Redtail	-0.0175
	Princess-Yellowtail	0.0440
	Queen-Redband	0.1726
	Queen-Redtail	0.0098
	Queen-Yellowtail	0.0020
	Redband-Striped	0.3391
	Redtail-Striped	-0.0138
	Striped-Yellowtail	-0.1042
Scraper-Excavator	Blue-Midnight	-0.0093
	Blue-Rainbow	0.0044
	Blue-Stoplight	0.0053
	Midnight-Princess	0.0539
	Midnight-Queen	-0.0146
	Midnight-Striped	-0.1044
	Princess-Rainbow	-0.0233
	Princess-Stoplight	0.4489
	Queen-Rainbow	-0.0736
	Queen-Stoplight	0.0925
	Rainbow-Striped	-0.1709
	Stoplight-Striped	0.2213

Supplemental Table 2.13: Median correlation for each interspecies comparison across functional groups in Roatan. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



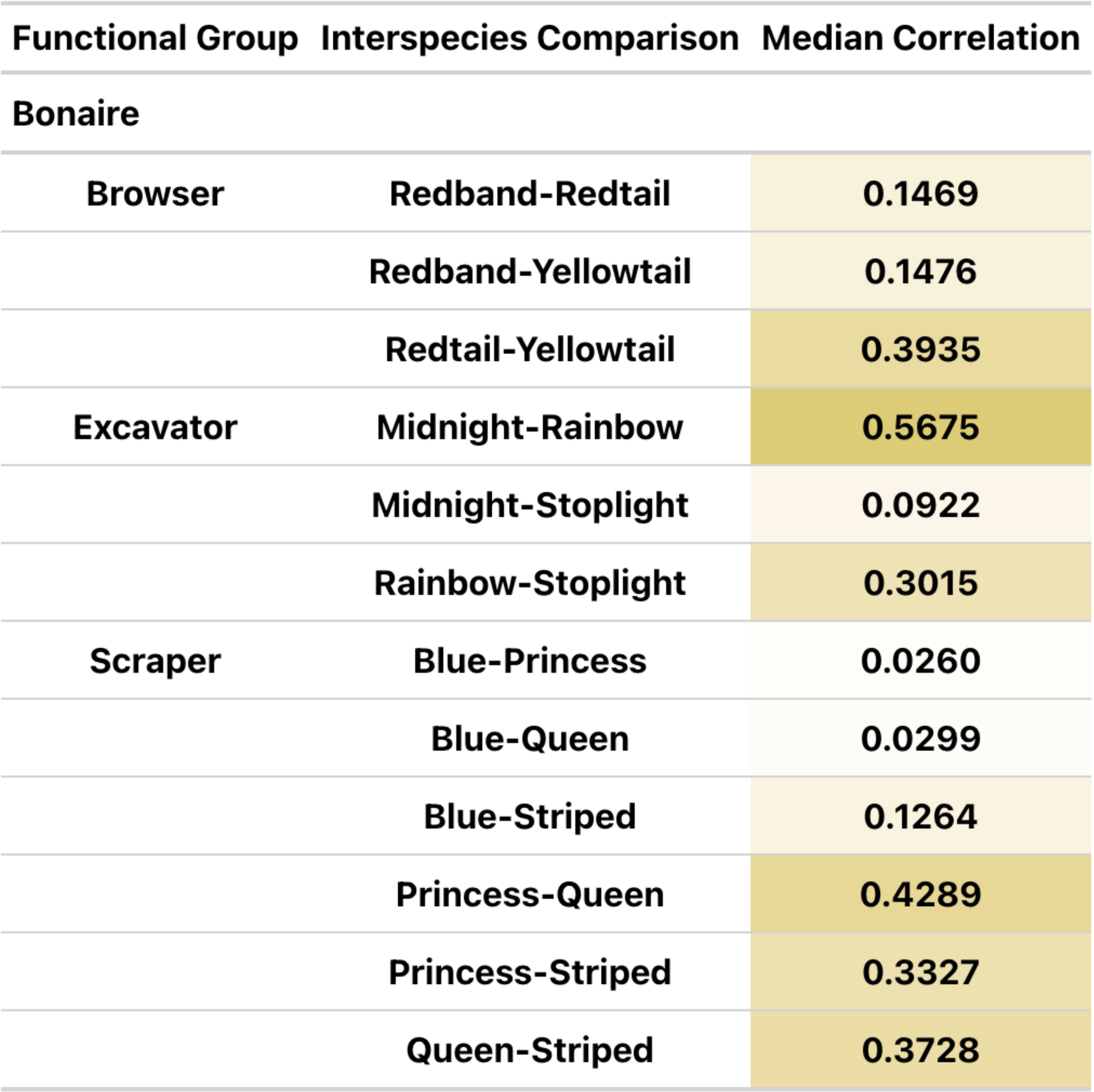
Functional Group Comparison	Interspecies Comparison	Median Correlation
Roatan		
Excavator-Browser	Midnight-Redband	-0.2438
	Midnight-Redtail	-0.1441
	Midnight-Yellowtail	-0.0269
	Rainbow-Redband	-0.1981
	Rainbow-Redtail	-0.0938
	Rainbow-Yellowtail	-0.0292
	Redband-Stoplight	-0.0843
	Redtail-Stoplight	-0.2390
	Stoplight-Yellowtail	-0.2006
Scraper-Browser	Blue-Redband	-0.1574
	Blue-Redtail	-0.1327
	Blue-Yellowtail	-0.1423
	Princess-Redband	-0.0386
	Princess-Redtail	-0.1348
	Princess-Yellowtail	-0.2030
	Queen-Redband	-0.1473
	Queen-Redtail	-0.1298
	Queen-Yellowtail	-0.0152
	Redband-Striped	0.3673
	Redtail-Striped	0.4262
	Striped-Yellowtail	0.2308
Scraper-Excavator	Blue-Midnight	0.1620
	Blue-Rainbow	0.0814
	Blue-Stoplight	0.1733
	Midnight-Princess	0.0814
	Midnight-Queen	0.1391
	Midnight-Striped	-0.1951
	Princess-Rainbow	0.0185
	Princess-Stoplight	0.2213
	Queen-Rainbow	0.1277
	Queen-Stoplight	0.1517
	Rainbow-Striped	-0.1059
	Stoplight-Striped	-0.2343

Supplemental Table 2.14: Median correlation for each interspecies comparison across functional groups in Tortolla. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



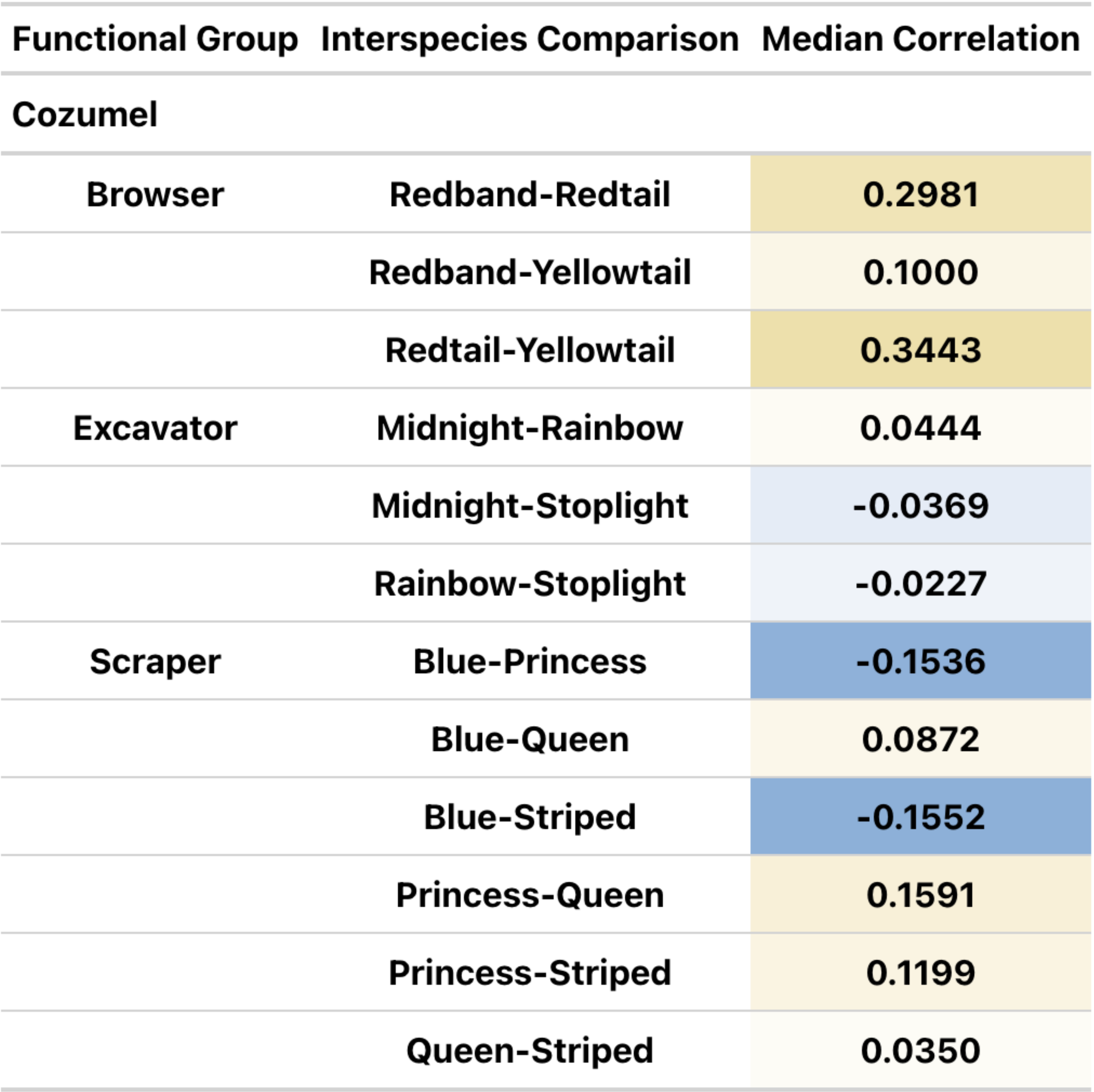
Functional Group Comparison	Interspecies Comparison	Median Correlation
Tortola		
Excavator-Browser	Midnight-Redband	-0.2295
	Midnight-Redtail	0.0958
	Midnight-Yellowtail	0.0126
	Rainbow-Redband	-0.3266
	Rainbow-Redtail	0.1543
	Rainbow-Yellowtail	-0.0202
	Redband-Stoplight	0.1140
	Redtail-Stoplight	0.0964
	Stoplight-Yellowtail	0.1872
Scraper-Browser	Blue-Redband	-0.3236
	Blue-Redtail	0.0853
	Blue-Yellowtail	-0.0419
	Princess-Redband	0.0587
	Princess-Redtail	0.0710
	Princess-Yellowtail	0.0253
	Queen-Redband	0.2567
	Queen-Redtail	0.2881
	Queen-Yellowtail	0.3011
	Redband-Striped	0.2284
	Redtail-Striped	0.1589
	Striped-Yellowtail	0.1409
Scraper-Excavator	Blue-Midnight	0.1510
	Blue-Rainbow	0.5702
	Blue-Stoplight	0.0661
	Midnight-Princess	0.1205
	Midnight-Queen	-0.0765
	Midnight-Striped	-0.1027
	Princess-Rainbow	-0.1482
	Princess-Stoplight	0.0330
	Queen-Rainbow	-0.0835
	Queen-Stoplight	0.1679
	Rainbow-Striped	-0.1504
	Stoplight-Striped	0.0681

Supplemental Table 2.15: Median correlation for each interspecies comparison within each functional group in Bonaire. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



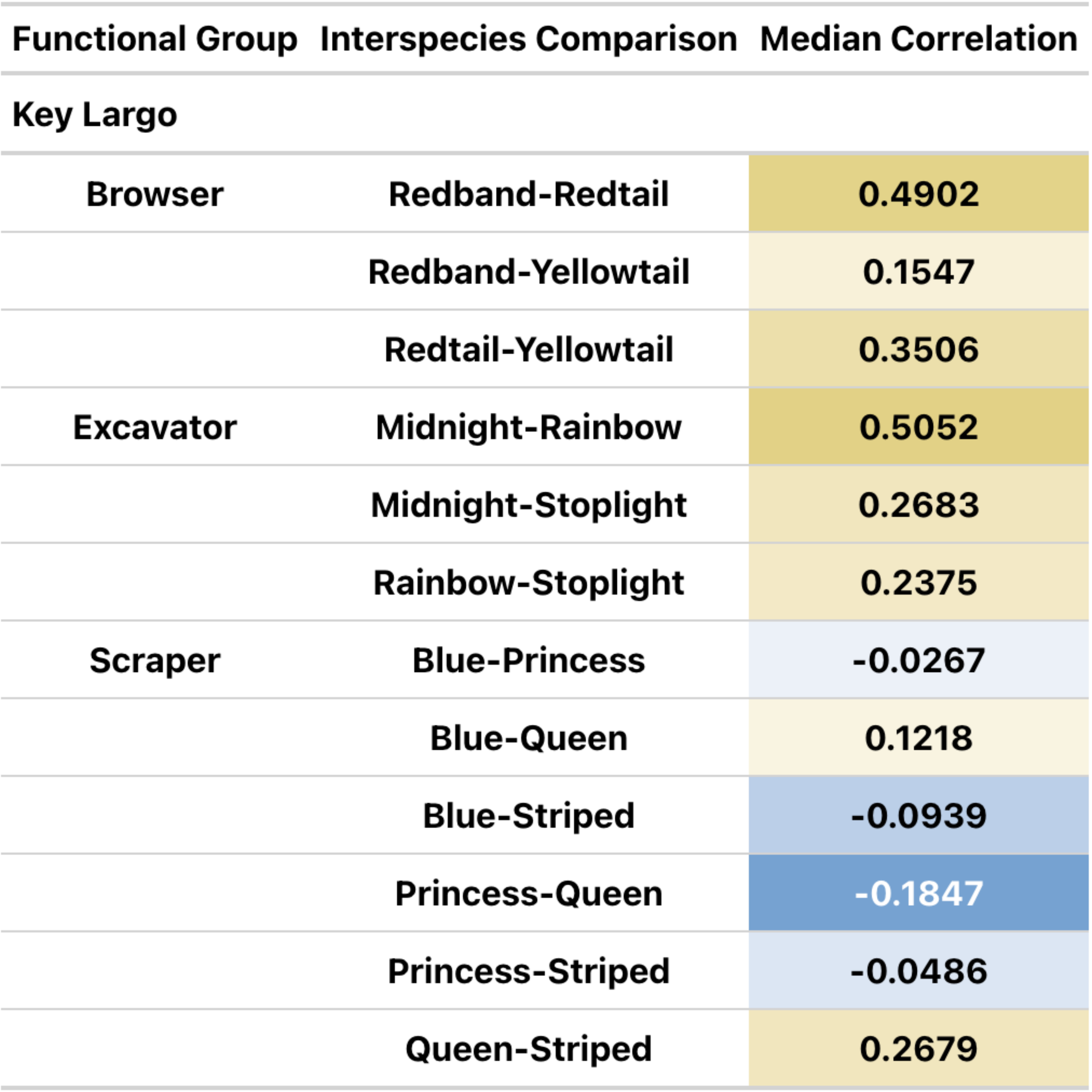
Functional Group	Interspecies Comparison	Median Correlation
Bonaire		
Browser	Redband-Redtail	0.1469
	Redband-Yellowtail	0.1476
	Redtail-Yellowtail	0.3935
Excavator	Midnight-Rainbow	0.5675
	Midnight-Stoplight	0.0922
	Rainbow-Stoplight	0.3015
Scraper	Blue-Princess	0.0260
	Blue-Queen	0.0299
	Blue-Striped	0.1264
	Princess-Queen	0.4289
	Princess-Striped	0.3327
	Queen-Striped	0.3728

Supplemental Table 2.16: Median correlation for each interspecies comparison within each functional group in Cozumel. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



Functional Group	Interspecies Comparison	Median Correlation
Cozumel		
Browser	Redband-Redtail	0.2981
	Redband-Yellowtail	0.1000
	Redtail-Yellowtail	0.3443
Excavator	Midnight-Rainbow	0.0444
	Midnight-Stoplight	-0.0369
	Rainbow-Stoplight	-0.0227
Scraper	Blue-Princess	-0.1536
	Blue-Queen	0.0872
	Blue-Striped	-0.1552
	Princess-Queen	0.1591
	Princess-Striped	0.1199
	Queen-Striped	0.0350

Supplemental Table 2.17: Median correlation for each interspecies comparison within each functional group in Key Largo. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.

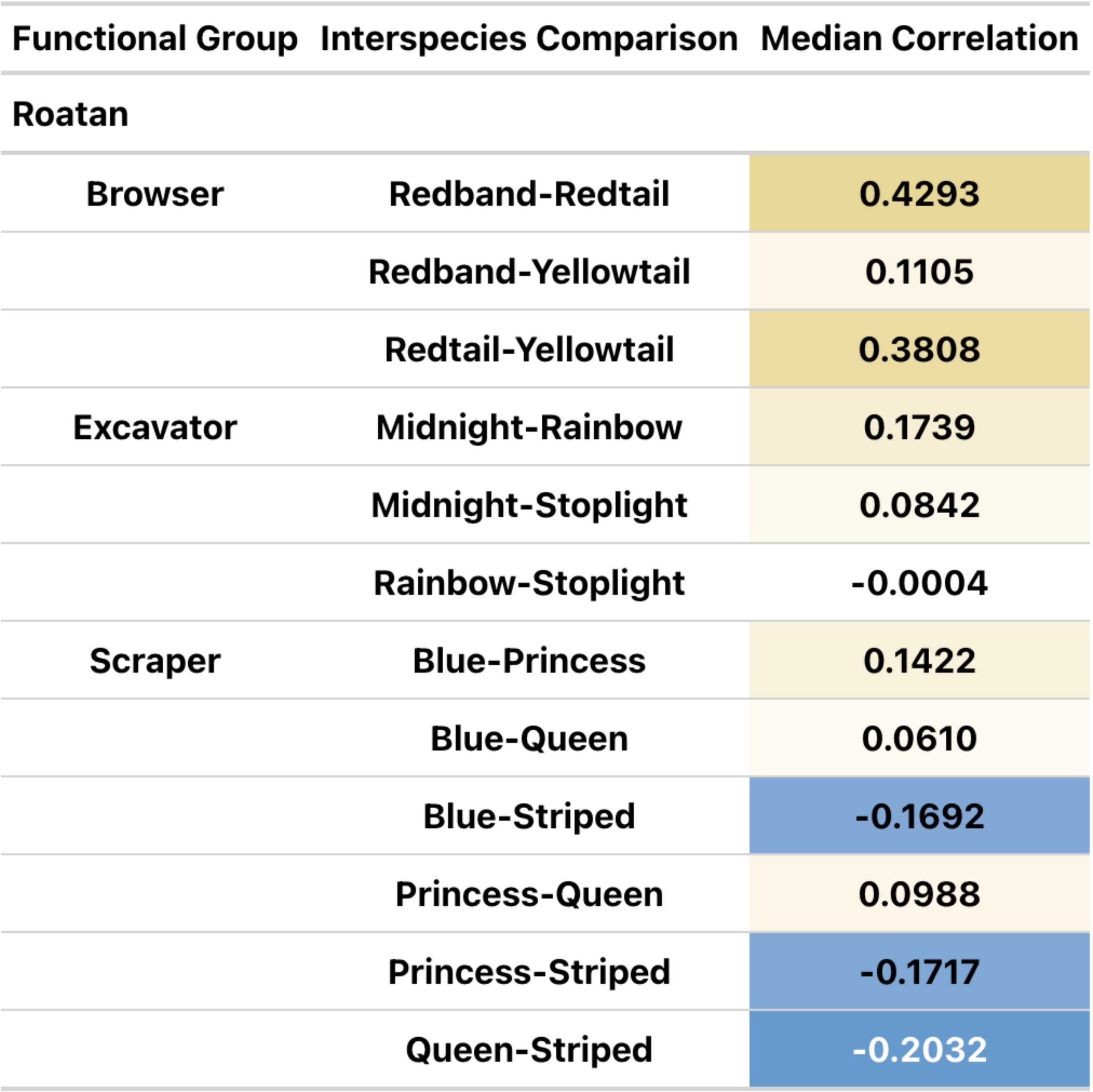


Functional Group	Interspecies Comparison	Median Correlation
Key Largo		
Browser	Redband-Redtail	0.4902
	Redband-Yellowtail	0.1547
	Redtail-Yellowtail	0.3506
Excavator	Midnight-Rainbow	0.5052
	Midnight-Stoplight	0.2683
	Rainbow-Stoplight	0.2375
Scraper	Blue-Princess	-0.0267
	Blue-Queen	0.1218
	Blue-Striped	-0.0939
	Princess-Queen	-0.1847
	Princess-Striped	-0.0486
	Queen-Striped	0.2679

Supplemental Table 2.18: Median correlation for each interspecies comparison within each functional group in Little Cayman. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.

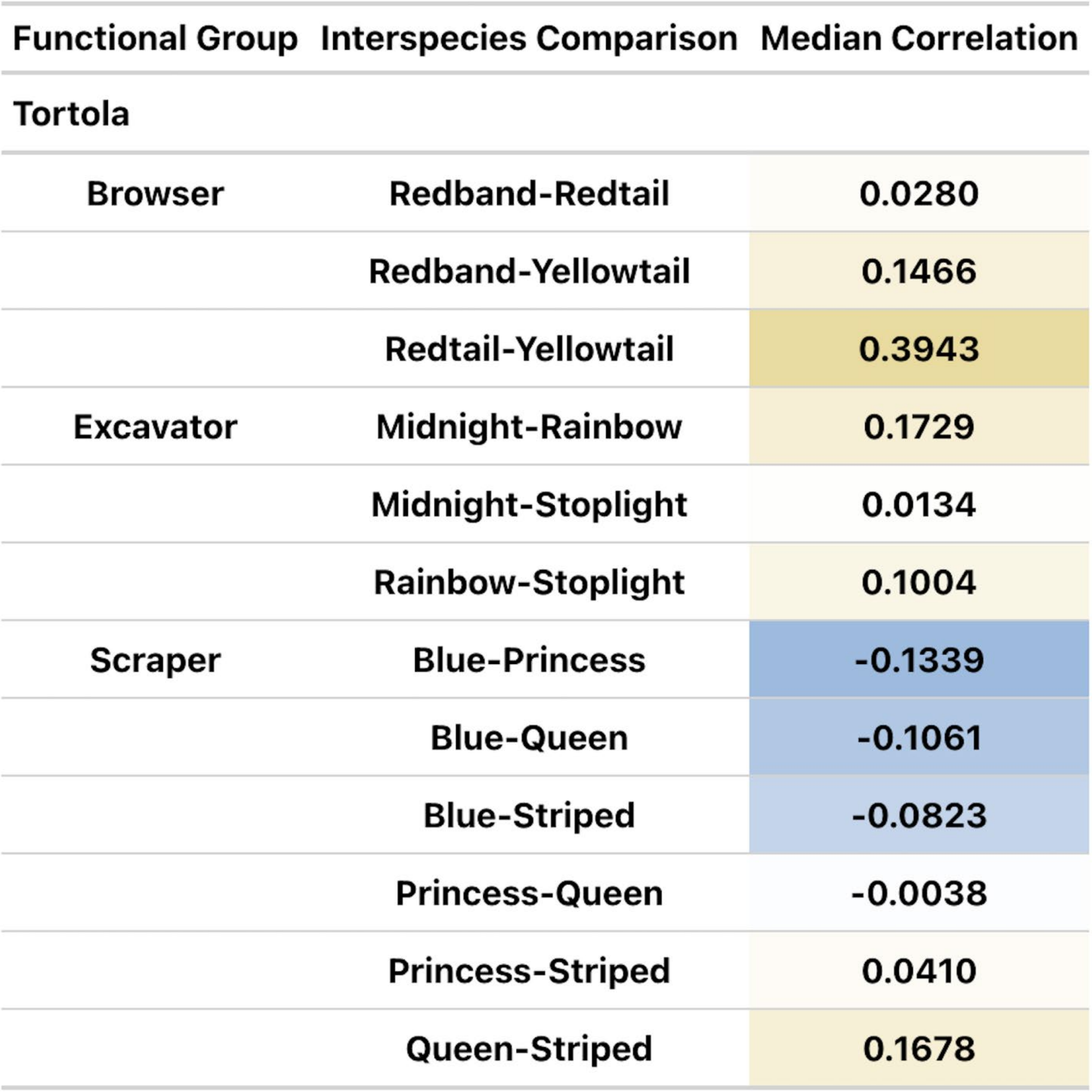
Functional Group	Interspecies Comparison	Median Correlation
Little Cayman		
Browser	Redband-Redtail	0.0363
	Redband-Yellowtail	-0.2031
	Redtail-Yellowtail	0.0053
Excavator	Midnight-Rainbow	0.1629
	Midnight-Stoplight	0.1934
	Rainbow-Stoplight	0.0961
Scraper	Blue-Princess	0.0011
	Blue-Queen	0.0390
	Blue-Striped	0.0053
	Princess-Queen	0.1197
	Princess-Striped	0.3781
	Queen-Striped	0.0555

Supplemental Table 2.19: Median correlation for each interspecies comparison within each functional group in Roatan. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



Functional Group	Interspecies Comparison	Median Correlation
Roatan		
Browser	Redband-Redtail	0.4293
	Redband-Yellowtail	0.1105
	Redtail-Yellowtail	0.3808
Excavator	Midnight-Rainbow	0.1739
	Midnight-Stoplight	0.0842
	Rainbow-Stoplight	-0.0004
Scraper	Blue-Princess	0.1422
	Blue-Queen	0.0610
	Blue-Striped	-0.1692
	Princess-Queen	0.0988
	Princess-Striped	-0.1717
	Queen-Striped	-0.2032

Supplemental Table 2.20: Median correlation for each interspecies comparison within each functional group in Tortolla. The color of the cell denotes sign and degree of correlations, where yellow hues represent positive correlations and blue hues represent negative correlations. The darker the hue the more positive or negative the correlation.



Functional Group	Interspecies Comparison	Median Correlation
Tortola		
Browser	Redband-Redtail	0.0280
	Redband-Yellowtail	0.1466
	Redtail-Yellowtail	0.3943
Excavator	Midnight-Rainbow	0.1729
	Midnight-Stoplight	0.0134
	Rainbow-Stoplight	0.1004
Scraper	Blue-Princess	-0.1339
	Blue-Queen	-0.1061
	Blue-Striped	-0.0823
	Princess-Queen	-0.0038
	Princess-Striped	0.0410
	Queen-Striped	0.1678

Chapter 3

Investigating U.S. consumers’ understanding of seafood ecolabels

Jordan H. DiNardo, Brice X. Semmens

Abstract

Seafood ecolabel emerged in response to growing concern about the state of the world's fish stocks and increased demand for seafood, and are intended to incentivize demand for sustainably caught products. However, the efficacy of this market-based sustainability assessment tool partly hinges on consumer understanding of them. In this study we conducted a nationwide online consumer survey to assess U.S. consumers' familiarity with, and ability to discern, seafood ecolabels as a function of their demographic traits, geography, personal values, and engagement with information providers. Our findings indicated approximately 80% of U.S. consumers had limited or no ability to accurately discern seafood ecolabels. However, consumers who live closer to the coast, had more concern for the future of the environment, and who ate seafood regularly, were more informed of seafood ecolabeling schemes. Additionally, those consumers informed by a diversity of information providers (e.g., retailers, media, fishers) were better able to discern ecolabels. Older (65 and older) U.S. consumers, especially, exhibited the least understanding of seafood ecolabels. This older demographic was also the least willing to pay for ecolabelled seafood products – possibly due to a lack of understanding. To improve overall consumer awareness and understanding of seafood ecolabels, ecolabel schemes would benefit from prioritizing efforts to educate this demographic of consumers. Since retailers were identified as the key information providers for older U.S. consumers, any targeted outreach effort to improve consumer understanding of ecolabels would benefit from retailer training and education programs. Collectively, this study underscores the need for enhanced consumer education and outreach to improve understanding of seafood ecolabels to ultimately advance the efficacy of these market-based sustainability assessment tools.

Introduction

In the late 1900s, seafood ecolabels emerged in response to growing demand for seafood products, and coincident concern about the state of the world's fish stocks. Seafood ecolabels were designed to incentivize consumers to purchase sustainably caught seafood, ultimately increasing the demand for sustainably sourced seafood products (Cooke et al., 2011; Gopal & Boopendranath, 2013; Gutierrez & Thornton, 2014; Parkes et al., 2010). In principle, seafood ecolabels discern seafood that has been sourced in ways that minimize negative impacts on the environment by promoting responsible fishing practices. Consumer preference for ecolabelled seafood products offers several advantages to fishers, including premium pricing of their products (Gutiérrez et al., 2012; Roheim et al., 2011) and access to niche markets that prioritize sustainability (Gulbrandsen, 2009; Gutierrez & Thornton, 2014; Gutiérrez et al., 2012; Parkes et al., 2010; Pérez-Ramírez, Phillips, et al., 2012; Pérez-Ramírez, Ponce-Díaz, et al., 2012). Furthermore, UN Sustainable Development Goal 12 (SDG 12), which focuses on promoting responsible seafood consumption and production patterns (United Nations, 2023), has increased pressure on major food retailers to improve sustainable seafood sourcing. To do so, many retailers have turned to ecolabel schemes for guidance on good seafood sourcing practices.

In addition to seafood ecolabels, seafood rating programs (e.g., Monterey Bay Aquarium Seafood Watch (Monterey Bay Aquarium, 2024), NOAA FishWatch (NOAA Fisheries, 2024), Ocean Wise (Ocean Wise, 2024), etc) have emerged as another tool to assess the sustainability of seafood products. Seafood rating programs enlist the expertise of scientists to assess the sustainability of species or stocks that are considered important to seafood markets (e.g., Pacific Bluefin Tuna) and convey this information to consumers through a stop-light rating system, categorizing species/stocks as best choices (green), good alternatives (yellow), or those to avoid (red). This differs from seafood ecolabels which entail a formal, voluntary certification process sought by individual seafood producers (fishers or farmers) and enlist third-party organization (certification assessment bodies) to assess the sustainability of fisheries/farmers based on specific standards set by the ecolabel scheme. These differences allow for the seafood ecolabels to provide a chain of custody of products (traceability in the seafood supply chain) while seafood rating programs do not (Gutiérrez et al., 2012; Parkes et al., 2010).

The success of seafood ecolabels hinges on their ability to effectively communicate presumed sustainability of seafood to consumers in a manner that influences purchasing decisions. However, confusion resulting from the proliferation of ecolabels may diminish the impact of any one ecolabel scheme. Across seafood ecolabel schemes, there is little consensus on what constitutes "sustainable seafood", which can generate conflicting consumer-facing advice (Parkes et al., 2010). Furthermore, various marketing strategies exist that can mislead consumers into believing seafood products are sustainable when they may not actually meet authentic sustainability standards (i.e., greenwashing) (Client Earth, 2011). This complex seafood retail landscape can leave consumers overwhelmed and dubious of any ecolabel schemes focused on communicating and promoting sustainable seafood (Roheim et al., 2018).

Numerous consumer-oriented studies have been conducted to assess consumer familiarity, trust, and willingness to pay for seafood ecolabels (Carlucci et al., 2017; Feucht & Zander, 2014; Jenny Sun et al., 2017; Lawley et al., 2019; Onozaka et al., 2010; Peiró Signes et al., 2023; Pieniak et al., 2013; Salladarré et al., 2010; Sogn-Grundvåg et al., 2013; Thøgersen et al., 2010; Uchida et al., 2014; Wakamatsu et al., 2017). Most of these studies found that, although consumers' familiarity of seafood ecolabels was fairly low, consumers prefer and are willing to pay for ecolabelled seafood products. Although valuable, these findings fail to capture the consumers' understanding of the differences between seafood ecolabels' focus or intent (ability to discern seafood ecolabels), a necessary ability for consumers to confidently navigate the current seafood market. Accordingly, it is important to examine consumers' understanding of seafood ecolabels and what factors might be influencing such understanding.

In this study we examined the efficacy of seafood ecolabels designed to promote sustainable seafood through the lens of consumer awareness. By implementing a nationwide online consumer survey, we first assessed U.S. seafood consumer's familiarity with and ability to discern seafood ecolabels and rating programs, collectively termed market-based sustainability assessment (MBSA) tools. Since seafood ecolabels are more readily available for consumers to use when making their seafood purchasing decisions, we assessed U.S. consumers' understanding of seafood ecolabels through their ability to discern three seafood ecolabel schemes. Through the collection of factors related to consumer demographics, geography, personal values, and engagement with information providers, we investigated how such factors influence consumers' understanding of seafood ecolabels. Finally, we assessed U.S. consumers' willingness to pay for ecolabelled seafood products. Our findings recognize the need to prioritize education and outreach strategies amongst consumer demographics and how best to engage with them to ultimately improve U.S. consumers' understanding of seafood ecolabels.

Material and Methods

Data Collection

We collected data through a nationwide online consumer survey in July of 2022 administered by Qualtrics International Inc., a third-party experience management company that provides services for survey research and customer experience. We used Qualtrics' consumer panel database to solicit 500 seafood consumers across the U.S. Consumers who did not consume seafood products were excluded from the survey. In an effort to obtain equal sampling across sociodemographic groups, we set quotas for sociodemographic factors (gender, age, region, race, household income, and education). For factors where we were unable to meet the quota (region and income), we eased our quota constraints (see final response rate by factor in Table 3.1) to obtain 500 responses. Consumers were asked 34 questions (although total survey length varied based on responses) relating to their individual sociodemographics, values, knowledge and attitude towards environmental issues regarding seafood production, perceived consumer responsibility, familiarity, understanding, reliance (use of seafood ecolabels when shopping for seafood), and trust in seafood ecolabels and rating programs, engagement with information sources to acquire knowledge on sustainable seafood, and willingness to pay for seafood products with ecolabels (

Supplemental). We included multiple question types (yes/no, multiple-response, rank-ordered Likert scale, and free-response) and designed the survey so that consumers could complete their responses in approximately 10 minutes. The University of California, San Diego Institutional Review Board (IRB) certified this study of U.S. seafood consumers after IRB review.

Seafood consumer sociodemographics, geography, values, and knowledge and attitude towards seafood production

The sociodemographic categories in the consumer survey included age, gender, race, highest level of education, annual household income, political views, frequency (per week) of seafood consumption, and geography (distance from nearest coastline derived from consumers' zip code). Consumer personal values included concern for the future of the environment and perceived consumer responsibility (perception that they have an obligation as a consumer to engage in responsible and ethical purchases). We assessed consumers' knowledge of the seafood production systems (ability to discern wild-capture from aquaculture practices) by asking them to choose any statements that appropriately describe wild-capture fisheries from a list of four statements, three of which described wild-capture fisheries while one statement described aquaculture practices. Consumers that incorrectly selected the statement describing aquaculture practices were categorized to have no knowledge. The remaining consumers were categorized into knowledge categories based on the number of correct statements chosen (e.g., consumers that chose all three statements that described wild-capture fisheries correctly was categorized to have high knowledge of the seafood production systems). Only those consumers who had moderate to high knowledge of the seafood production systems were also asked to provide their concern for the impacts of wild-capture fisheries on the environment.

Familiarity, understanding, reliance, and trust in seafood ecolabels and rating programs

To capture consumer familiarity of seafood ecolabels we asked consumers if they were familiar with seafood ecolabels and rating programs (yes/no). To assess consumers' ability to discern seafood ecolabels and rating programs, we asked consumers who self-reported as familiar with both MBSA tools (n=74) to match four statements to the appropriate MBSA tool. We derived consumers' ability to discern seafood ecolabels and rating programs from the number of correct matches between MBSA tools and the description, making it an ordered categorical variable ranging from 0 correct matches (no ability) to 4 correct matches (high ability).

We assessed consumers' understanding of seafood ecolabels by their ability to discern seafood ecolabels. We asked consumers who self-reported as familiar with seafood ecolabels (n=147) to match up three seafood ecolabel logos (Dolphin Safe, Friend of the Sea, Marine Stewardship Council (MSC)) with the appropriate description. We derived consumers' understanding of seafood ecolabels from the number of correct matches between ecolabel logos and descriptions, making it an ordered categorical variable ranging from 0 correct matches (no understanding) to 3 correct matches (high understanding).

Among the consumers who self-reported as familiar with seafood ecolabels and rating programs (n=74), we also asked consumers to share their reliance and trust in both these MBSA tools (according to a five-point rank-ordered Likert scale). Rather than asking consumers to share their explicit trust in seafood rating programs, we asked consumers to share their trust in seafood rating programs relative to seafood ecolabels.

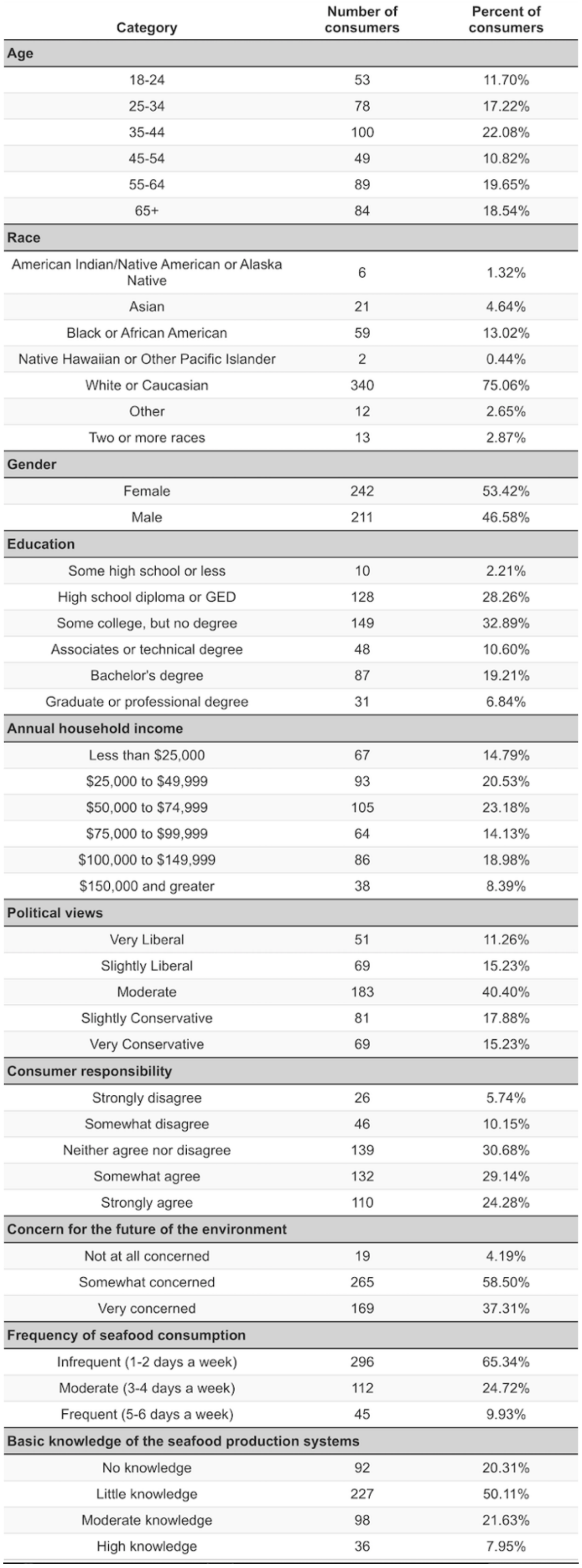
Engagement with information providers

To understand where consumers are acquiring their knowledge on sustainable seafood, we asked them to select from a list of potential information providers they engage with (e.g., visiting their website, attend educational events or conferences, read published reports, etc.), including international (e.g., Inter American Tropical Tuna Commission) and domestic (NOAA Fisheries) management organizations, non-governmental organizations (NGOs; e.g., The Nature Conservancy), ecolabel schemes (e.g., MSC), seafood rating programs (e.g., Monterey Bay Aquarium Seafood Watch), fishers, retailers, and media (e.g., social media platforms (Instagram, TikTok), television news, radio, documentaries, etc.). Consumers were asked to select all information providers they engage with. For each consumer response we derived a level of engagement based on the total number of information providers they selected. We also investigated general consumer engagement with information providers, across all consumers and by general age categories. For each information provider we derived the total number of consumers engaged with, also termed 'degree centrality' in network analysis literature (Magaia et al., 2015)).

Willingness to pay for seafood products with ecolabels

Although capturing willingness to pay typically entails an applied, experimental approach using contingent valuation, we asked consumers questions about their willingness to pay for ecolabelled seafood products. Consumers (n=500) were asked if ecolabel seafood products should cost more than seafood products without ecolabels (yes/no) and how willing they are to pay more for ecolabelled seafood products (3-point rank-ordered Likert scale). Like consumer engagement, we examined how consumer willingness to pay for ecolabelled seafood products varied across age categories.

Table 3.1: Sociodemographic and basic characteristics of U.S. seafood consumers survey consumers.



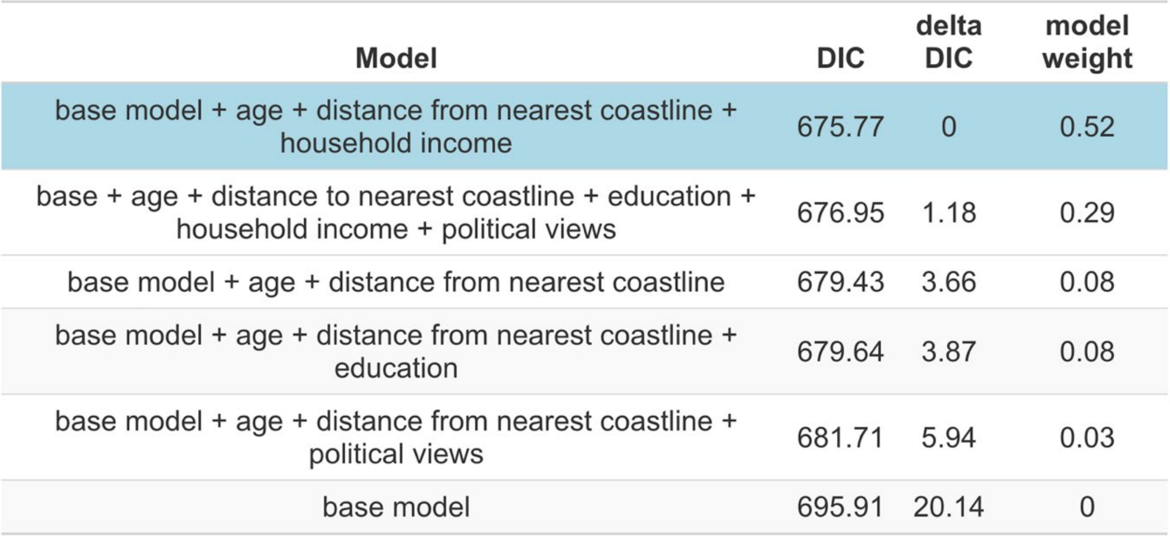
Category	Number of consumers	Percent of consumers
Age		
18-24	53	11.70%
25-34	78	17.22%
35-44	100	22.08%
45-54	49	10.82%
55-64	89	19.65%
65+	84	18.54%
Race		
American Indian/Native American or Alaska Native	6	1.32%
Asian	21	4.64%
Black or African American	59	13.02%
Native Hawaiian or Other Pacific Islander	2	0.44%
White or Caucasian	340	75.06%
Other	12	2.65%
Two or more races	13	2.87%
Gender		
Female	242	53.42%
Male	211	46.58%
Education		
Some high school or less	10	2.21%
High school diploma or GED	128	28.26%
Some college, but no degree	149	32.89%
Associates or technical degree	48	10.60%
Bachelor's degree	87	19.21%
Graduate or professional degree	31	6.84%
Annual household income		
Less than $25,000	67	14.79%
$25,000 to $49,999	93	20.53%
$50,000 to $74,999	105	23.18%
$75,000 to $99,999	64	14.13%
$100,000 to $149,999	86	18.98%
$150,000 and greater	38	8.39%
Political views		
Very Liberal	51	11.26%
Slightly Liberal	69	15.23%
Moderate	183	40.40%
Slightly Conservative	81	17.88%
Very Conservative	69	15.23%
Consumer responsibility		
Strongly disagree	26	5.74%
Somewhat disagree	46	10.15%
Neither agree nor disagree	139	30.68%
Somewhat agree	132	29.14%
Strongly agree	110	24.28%
Concern for the future of the environment		
Not at all concerned	19	4.19%
Somewhat concerned	265	58.50%
Very concerned	169	37.31%
Frequency of seafood consumption		
Infrequent (1-2 days a week)	296	65.34%
Moderate (3-4 days a week)	112	24.72%
Frequent (5-6 days a week)	45	9.93%
Basic knowledge of the seafood production systems		
No knowledge	92	20.31%
Little knowledge	227	50.11%
Moderate knowledge	98	21.63%
High knowledge	36	7.95%

Statistical Analysis

Since consumer understanding was collected as an ordered categorical variable, we developed an ordinal logistic model with cut points representative of the 4 levels of understanding ranging from no understanding (a combination of consumers with no familiarity of seafood ecolabels or those that incorrectly identified all three seafood ecolabel logos) to high knowledge (those consumers who self-reported as familiar with seafood ecolabel, and that correctly matched all three seafood ecolabels to their corresponding definitions).

To determine the key factors that are the most influential on consumer understanding of seafood ecolabels, we developed six models with differing sets of factors (Table 3.2). First, we defined a base model that comprised three essential factors thought to influence consumer understanding of seafood ecolabels most: concern for the future of the environment, frequency of seafood consumption, and level of engagement with information providers. Subsequently, we expanded on this base model by including additional factors such as distance from nearest coastline, age, education, household income, and political views, all of which could potentially influence consumer understanding of seafood ecolabels. We performed model selection, using Deviance Information Criterion (DIC) (Spiegelhalter et al., 2002), to evaluate relative performance of candidate models with differing factors used to explain consumer understanding of seafood ecolabels. Factors related to consumers' concern for the future of the environment, frequency of seafood consumption, level of engagement, and distance from nearest coastline were treated as continuous variables. The sociodemographic factors included in the candidate models (age, education, household income, and political views) were treated as random effects. For a full description of the best performing model, refer to the supplemental information (Supplemental Table 3.1).

Table 3.2: Model description and selection to determine the best set of factors influencing U.S. consumers' understanding of seafood ecolabels. The base model includes age, household income, perceived consumer responsibility, concern for the future of the environment, seafood consumption, proximity to closest coastline, and level of engagement. The best performing model (base model) is highlighted in blue.



Model	DIC	delta DIC	model weight
base model + age + distance from nearest coastline + household income	675.77	0	0.52
base + age + distance to nearest coastline + education + household income + political views	676.95	1.18	0.29
base model + age + distance from nearest coastline	679.43	3.66	0.08
base model + age + distance from nearest coastline + education	679.64	3.87	0.08
base model + age + distance from nearest coastline + political views	681.71	5.94	0.03
base model	695.91	20.14	0

We used JAGS software (Plummer, 2003) and the 'R2jags' package (Su & Yajima, 2021) to fit the ordinal logistic model in R (v4.04) using Bayesian Monte Carlo Markov Chain (MCMC) sampling. We ran each model using 4 parallel MCMC chains, each for a total of 20,000 iterations. Model parameters were updated every 1,000 iterations. Within each chain, thinning was applied to retain every 5th iteration, resulting in 4,000 samples per chain, and 16,000 posterior samples across all chains. We confirmed model convergence by conducting visual inspections of trace plots and assessing the potential scale reduction factor ($\hat{R}$; striving for $\hat{R}$ ratio <1.05 for all variables) for all variables included in the model (Gelman & Rubin, 1992) using the 'coda' package (v0.19-4) (Plummer, 2003) in R. We used posterior estimates to investigate how each factor influenced consumer understanding of seafood ecolabels.

Results

Among the 500 seafood consumers solicited, we obtained 496 useable responses. The four surveys omitted from the study inputted invalid zip codes of residence. For the analysis we omitted 43 additional responses that preferred not to provide information on their sociodemographic characteristics, resulting in 453 consumer survey responses included in the statistical analysis.

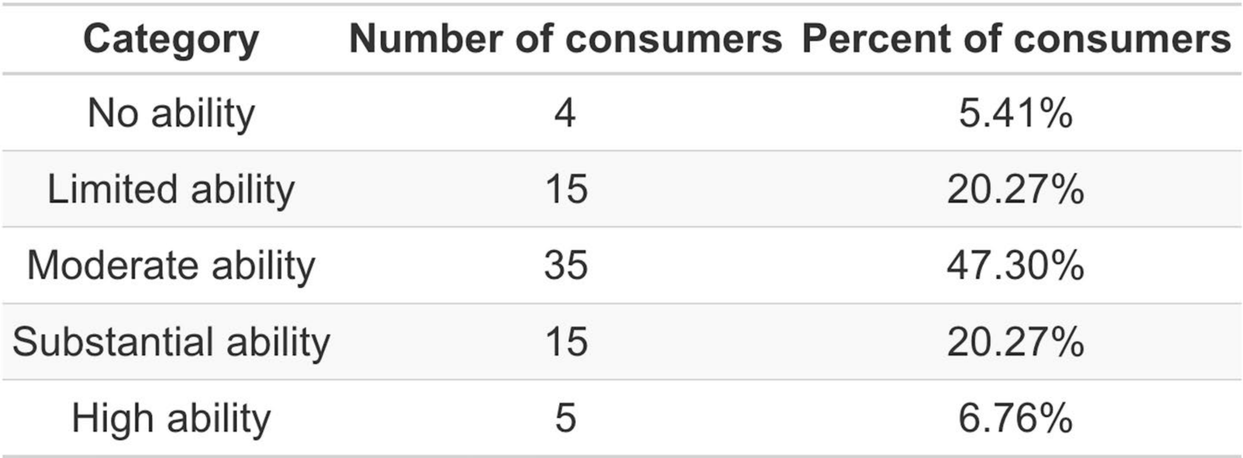
Consumer demographic and basic characteristics

Consumers were evenly distributed across age categories (Table 3.1), with the largest and smallest age categories being 35-44 (22.08%) and 45-54 (10.82%), respectively. Most consumers identified as white (75.06%), followed by black (13.02%). Each of the remaining race categories represented less than 5% of all consumers. Consumers were approximately evenly distributed across genders (Female: 53.42%; Male: 46.58%). Most consumers had some college, but no degree (32.89%), followed by a high school diploma or GED (28.26%) and bachelor's degree (19.21%). Each of the remaining education categories represented approximately 10% or less of consumers. Nearly 60% of consumers had a household income less than $75,000 (Less than $25,000: 14.79%; $25,000-$49,999: 20.53%; $50,000-$74,999: 23.18%). Among the remaining consumers, most had a household income of $100,000-$149,000 (18.98%), followed by $75,000 to $99,999 (14.13%), and $150,000 and greater (8.39%). The distribution of consumers' political views was centrally peaked, where most consumers identified as moderate (40.40%). The two extreme categories represented nearly 30% of consumers (Very Liberal: 11.26%; Very Conservative: 15.23%). While most consumers (30.68%) felt indifferent about consumer responsibility, the general feeling tended towards the perception that consumers have an obligation to engage in responsible and ethical purchases (Somewhat agree: 29.14%; Strongly agree: 24.28%). Many consumers (58.50%) were somewhat concerned for the future of the environment. Among the remaining consumers, 37.1% were very concerned and 4.19% were not concerned at all about the future of the environment. Most respondents (65.34%) consumed seafood infrequently (1-2 days per week), while 24.72% and 9.93% consumed seafood moderately (3-4 days per week) or frequently (5-6 days per week), respectively. The distribution of consumers' basic knowledge of the seafood production systems skewed towards uninformed, with most consumers having no (20.31%) or little (50.11%) knowledge. Relatively few consumers had moderate (21.63%) or high (7.95%) knowledge of seafood production systems. Among consumers that had moderate to high knowledge of the seafood production systems (n=134 consumers), most consumers were somewhat concerned of the impact of wild-capture fisheries on the environment (70.15%), while the remaining consumers were evenly distributed across not at all concerned (14.93%) and extremely concerned (14.93%).

Familiarity with and ability to distinguish seafood ecolabels and rating programs

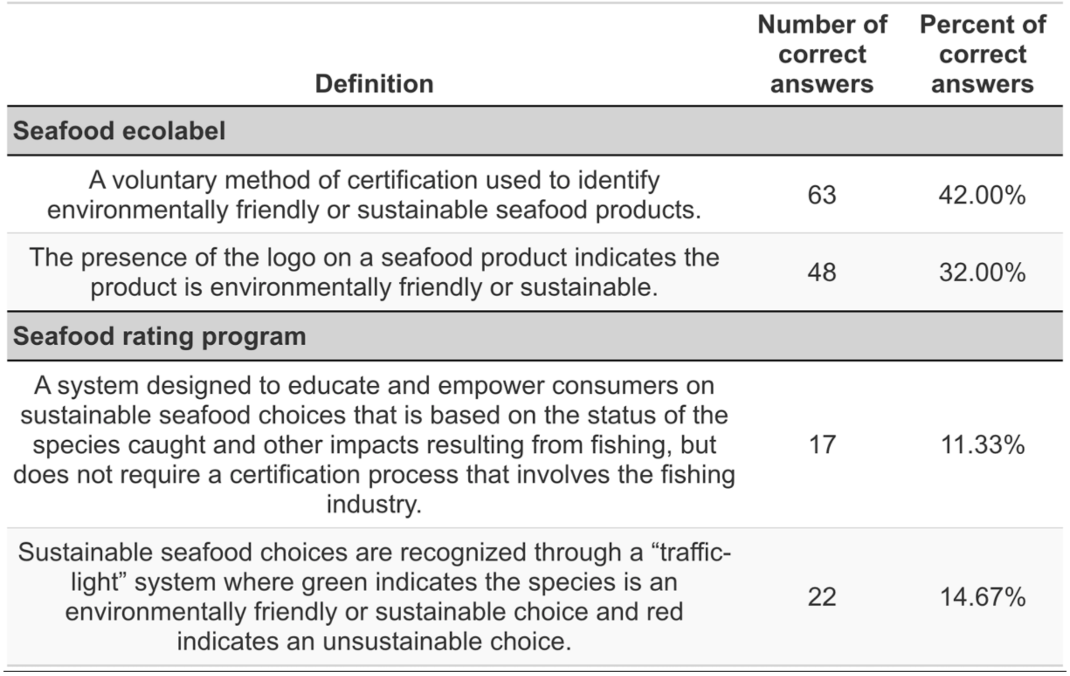
Most consumers self-reported as familiar with seafood ecolabels (67.55%) and rating programs (71.52%). Slightly more consumers self-reported as familiar with seafood ecolabels (32.45%) compared to seafood rating programs (28.48%). The distribution of consumers' ability to discern seafood ecolabels and rating programs was centrally peaked (Table 3.3). Among consumers who self-reported as familiar with seafood ecolabels and rating programs (n=74), most consumers (47.30%) had moderate ability to discern seafood ecolabels and rating programs (correctly matched two descriptions to their corresponding tool). Approximately 20% of consumers had limited ability to discern seafood ecolabels and rating program (20.27%; correctly matched one description to its corresponding tool). Similarly, 20.27% of consumers had substantial ability to discern seafood ecolabels and rating program (correctly matched three descriptions to their corresponding tool). Finally, few consumers had no ability (5.41%; failed to match the tools with any of their appropriate descriptions) and high ability (6.76%; correctly matched four descriptions to their corresponding tool) to discern seafood ecolabels and rating programs. Seafood ecolabels had higher rates of correct answers (42.00% and 32.00% among the two descriptions) while seafood rating programs had much lower rates of correct answers (11.33% and 14.67%) (Table 3.4).

Table 3.3: Ability to discern seafood ecolabels from rating programs. Only consumers who self-reported as familiar with seafood ecolabels and rating programs (n=74) were assessed.



Category	Number of consumers	Percent of consumers
No ability	4	5.41%
Limited ability	15	20.27%
Moderate ability	35	47.30%
Substantial ability	15	20.27%
High ability	5	6.76%

Table 3.4: Survey question aiming to assess consumers' ability to discern seafood ecolabels from rating program. The statements presented to consumers to match to the appropriate MBSA tool are categorized into seafood ecolabels and rating programs with the associated number of correct answers and percent of correct answers among consumers. Only consumers who self-reported as familiar with both seafood ecolabels and rating programs (n=74) were assessed.



Definition	Number of correct answers	Percent of correct answers
Seafood ecolabel		
A voluntary method of certification used to identify environmentally friendly or sustainable seafood products.	63	42.00%
The presence of the logo on a seafood product indicates the product is environmentally friendly or sustainable.	48	32.00%
Seafood rating program		
A system designed to educate and empower consumers on sustainable seafood choices that is based on the status of the species caught and other impacts resulting from fishing, but does not require a certification process that involves the fishing industry.	17	11.33%
Sustainable seafood choices are recognized through a "traffic-light" system where green indicates the species is an environmentally friendly or sustainable choice and red indicates an unsustainable choice.	22	14.67%

Understanding of seafood ecolabels

Among the consumers who self-reported as familiar with ecolabels (n=147), most had limited understanding of ecolabels (75.51%; correctly matched only one ecolabel logo to the appropriate description) (Table 3.5). Consumers with moderate and high understanding of ecolabels (correctly matched two or three ecolabel logos to their appropriate descriptions, respectively) represented approximately 19% of consumers who self-reported as familiar with seafood ecolabels, cumulatively (Moderate understanding: 8.84%; High understanding: 9.52%). Few consumers who self-reported as familiar with seafood ecolabels had no understanding (6.12% failed to match any ecolabel logo with the appropriate description). The seafood ecolabel with the highest rate of correct answers was the Dolphin Safe ecolabel (73.74%), followed by MSC (17.88%), and Friend of the Sea (8.38%) ecolabels (Table 3.6).

Table 3.5: Understanding of seafood ecolabels. Only consumers who self-reported as familiar with seafood ecolabels (n=147) were assessed on their understanding of seafood ecolabels.

Category	Number of consumers	Percent of consumers
No understanding	9	6.12%
Limited understanding	111	75.51%
Moderate understanding	13	8.84%
High understanding	14	9.52%

Table 3.6: Survey question aiming to assess consumers' understanding of seafood ecolabels (n=147). The seafood ecolabel logos presented to consumers to match to the appropriate statement describing their focus is listed next to the appropriate statements with the associated number of correct answers and percent of correct answers among consumers.

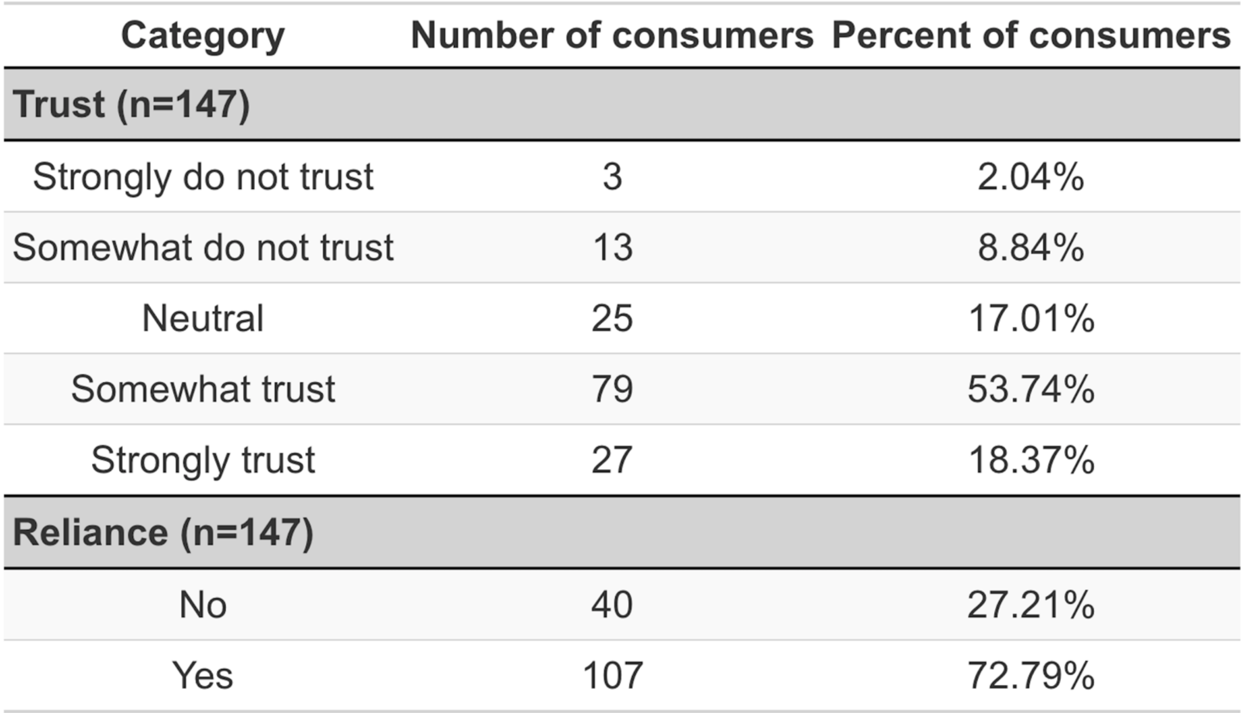
Ecolabel Logo	Correct answer	Number of correct answers	Percent of correct answers
SAFE®	An ecolabel used to show the seafood product was caught in a way that does not harm dolphins.	132	73.74%
FRIEND OF THE SEA	An ecolabel used to show the seafood product is not overfished, uses fishing methods with minimal environmental impacts (on other species and the environment), does not catch any vulnerable, endangered, or threatened species, reduction in carbon footprint, and has effective management in place on the target species caught.	15	8.38%
CERTIFIED SUSTAINABLE SEAFOOD MSC www.msc.org™	An ecolabel used to show the seafood product is not overfished, caught using methods with minimal environmental impacts (on other species and the environment), and has effective management in place on the target species caught.	32	17.88%

Trust and reliance on seafood ecolabels and rating programs

Among the consumers who self-reported as familiar with seafood ecolabels, most consumers indicated they somewhat trust seafood ecolabels (53.74%) (Table 3.7). Some consumers indicated they strongly trust (18.37%), or they were neutral (17.01%) about seafood ecolabels. The remaining categories represent approximately 10% of consumers, cumulatively (Somewhat do not trust: 8.84%; Strongly do not trust: 2.04%). Most consumers who self-reported familiar with seafood ecolabels indicated they rely on them when making decisions on their seafood purchases (72.79%; conversely 27.21% do not rely on them).

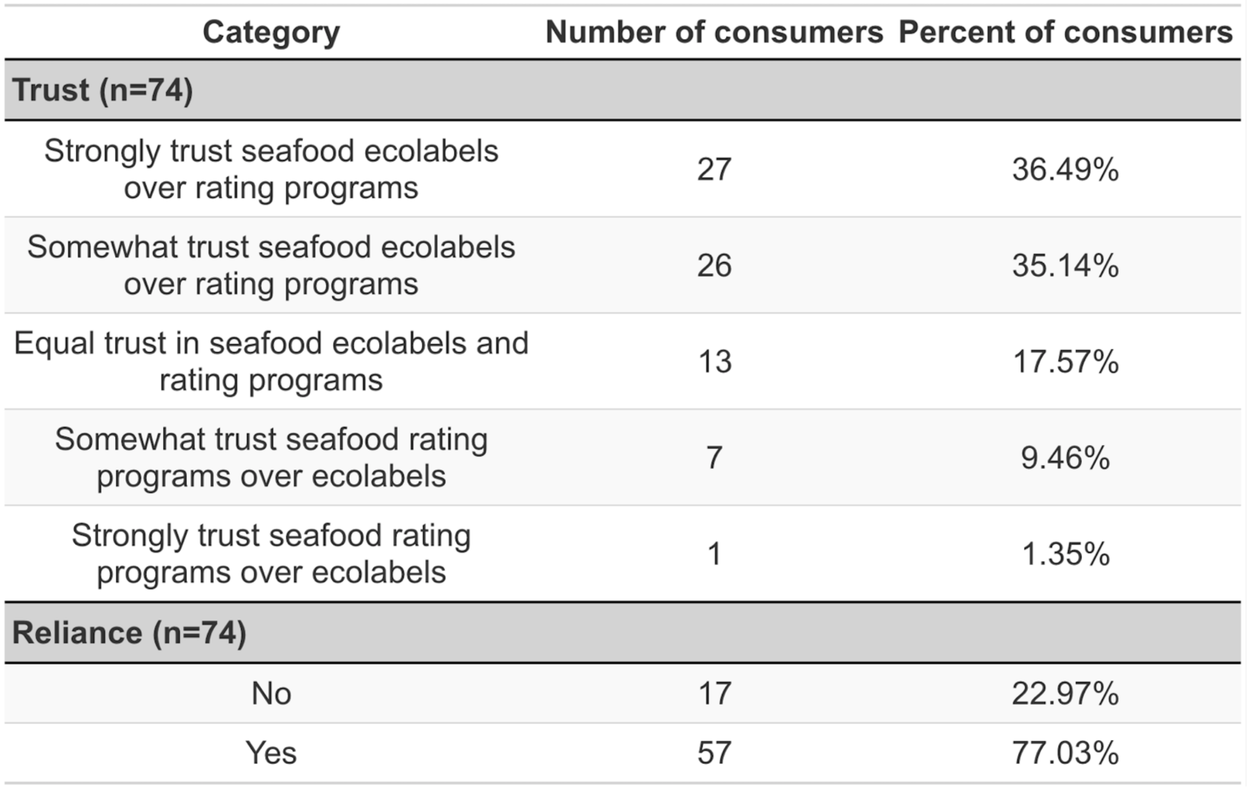
Among consumers who self-reported as familiar with seafood ecolabels and rating programs (n=74), nearly 72% of consumers indicated they strongly (36.49%) or somewhat (35.14%) trust ecolabels more than rating programs, cumulatively (Table 3.8). Conversely, approximately 10% of consumers indicated they strongly (1.35%) or somewhat (9.46%) trust rating programs over ecolabels, cumulatively. The remaining 17.57% of consumers indicated they trust seafood ecolabels and rating programs equally.

Table 3.7: Trust and reliance on seafood ecolabels among U.S. seafood consumers. Reliance refers to whether consumers rely on seafood ecolabels when making decisions on their seafood purchases. Only consumers who self-reported as familiar with seafood ecolabels were included (n=147).



Category	Number of consumers	Percent of consumers
Trust (n=147)		
Strongly do not trust	3	2.04%
Somewhat do not trust	13	8.84%
Neutral	25	17.01%
Somewhat trust	79	53.74%
Strongly trust	27	18.37%
Reliance (n=147)		
No	40	27.21%
Yes	107	72.79%

Table 3.8: Trust and reliance on seafood rating programs among U.S. seafood consumers. Trust in seafood rating programs is reported relative to seafood ecolabels. Reliance refers to whether consumers rely on seafood ecolabels when making decisions on their seafood purchases. Only consumers who self-reported as familiar with seafood ecolabels and rating programs were included (n=74).



Category	Number of consumers	Percent of consumers
Trust (n=74)		
Strongly trust seafood ecolabels over rating programs	27	36.49%
Somewhat trust seafood ecolabels over rating programs	26	35.14%
Equal trust in seafood ecolabels and rating programs	13	17.57%
Somewhat trust seafood rating programs over ecolabels	7	9.46%
Strongly trust seafood rating programs over ecolabels	1	1.35%
Reliance (n=74)		
No	17	22.97%
Yes	57	77.03%

Engagement with stakeholders

The top three information providers that consumers relied on to gain knowledge on sustainable seafood included retailers (nc=161), and media (e.g., Instagram, TikTok, televisions news, radio, etc.; nc=115), and fishers (nc=88), respectively (Figure 3.1). Ecolabel schemes (nc=72) and seafood rating programs (nc=70) had the lowest number of connections. When consumers were aggregated into general age categories (18-34, 35-54, and 55+), the top three information providers for each age category varied slightly in rank. The largest proportion of consumers of ages 18-34 engaged with media (20.14%), followed by retailers (15.47%), and international management organizations (14.03%). Consumers of ages 35-54 engaged with retailers the most (20.00%), followed by international management organizations (13.46%), and non-government organizations (12.31%). The largest proportion of consumers who were 55 and older engaged with retailers (31.28%), followed by media (14.69%), and fishers (13.27%).

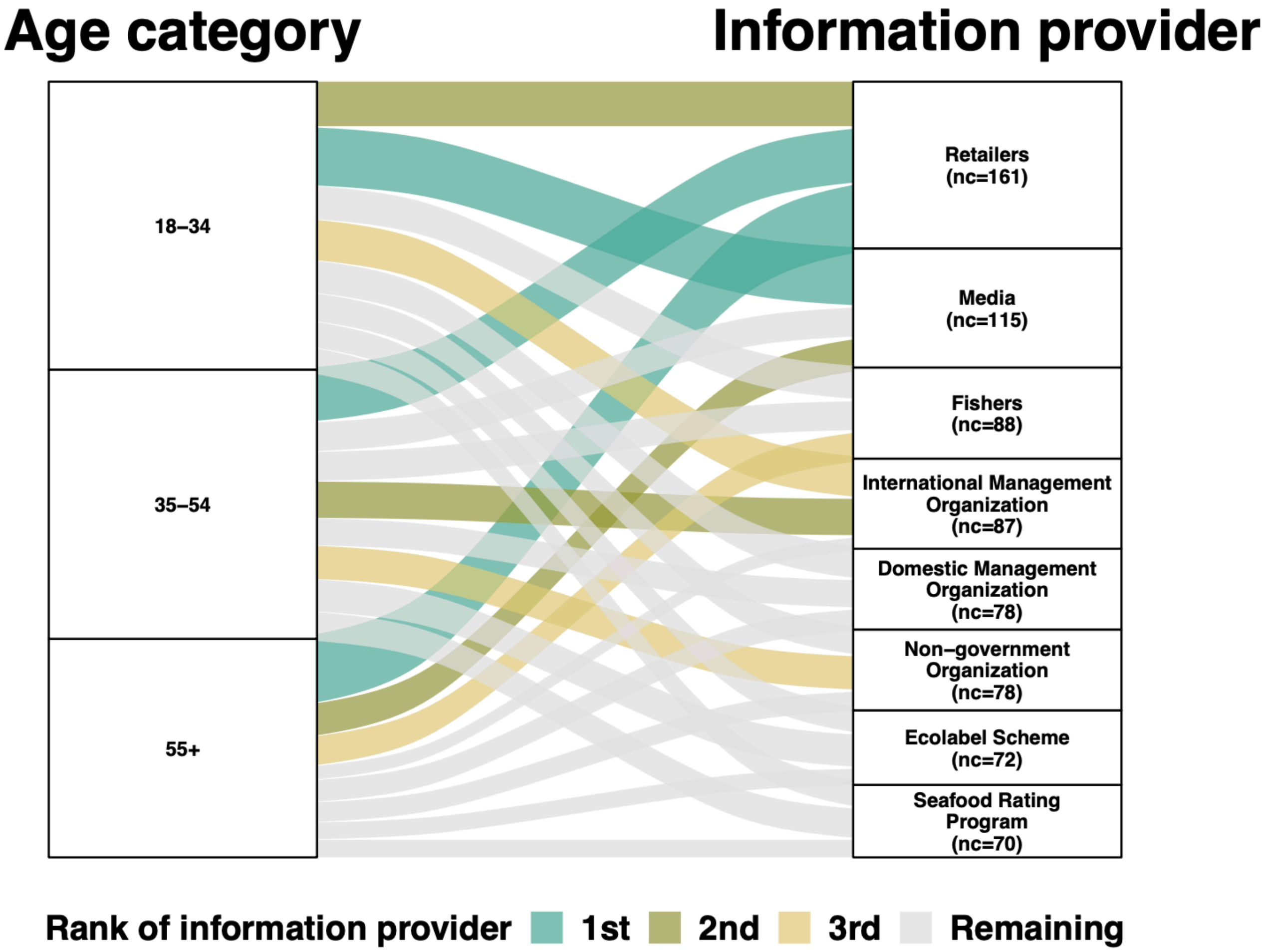


Figure 3.1: Engagement with information providers among U.S. consumers 18-34, 35-54, and 55 and older. Information providers are listed in descending order of total number of consumers (nc) engaged with. Colors denote the first (blue), second (green), third (yellow), and remaining (grey) information providers in descending order of nc by age category.

Factors influencing knowledge of seafood ecolabels

Model selection indicated that the most parsimonious model was comprised of the consumers' concern for the future of the environment, frequency of seafood consumption, level of engagement with information providers, distance from nearest coastline, age, and household income (Table 3.2, Figure 3.2), suggesting these factors were most influential on consumers' understanding of seafood ecolabels. Concern for the future of the environment, frequency of seafood consumption, and level of engagement with information providers were all positively related to understanding of seafood ecolabels. The distance from the nearest coastline was negatively related to consumer understanding of seafood ecolabels. Age was generally inversely related to understanding of seafood ecolabels, especially with consumers 35 and older (Figure 3.2). Consumers 65 and older had the lowest understanding of seafood ecolabels of any age category. Household income had a bimodal-like relationship with seafood ecolabel understanding, where consumers with household incomes less than $25,000 and $75,000 to $99,000 had the least understanding of seafood ecolabels and consumers with household incomes of $50,000 to $74,999 and $150,000 and greater had the highest understanding of seafood ecolabels.

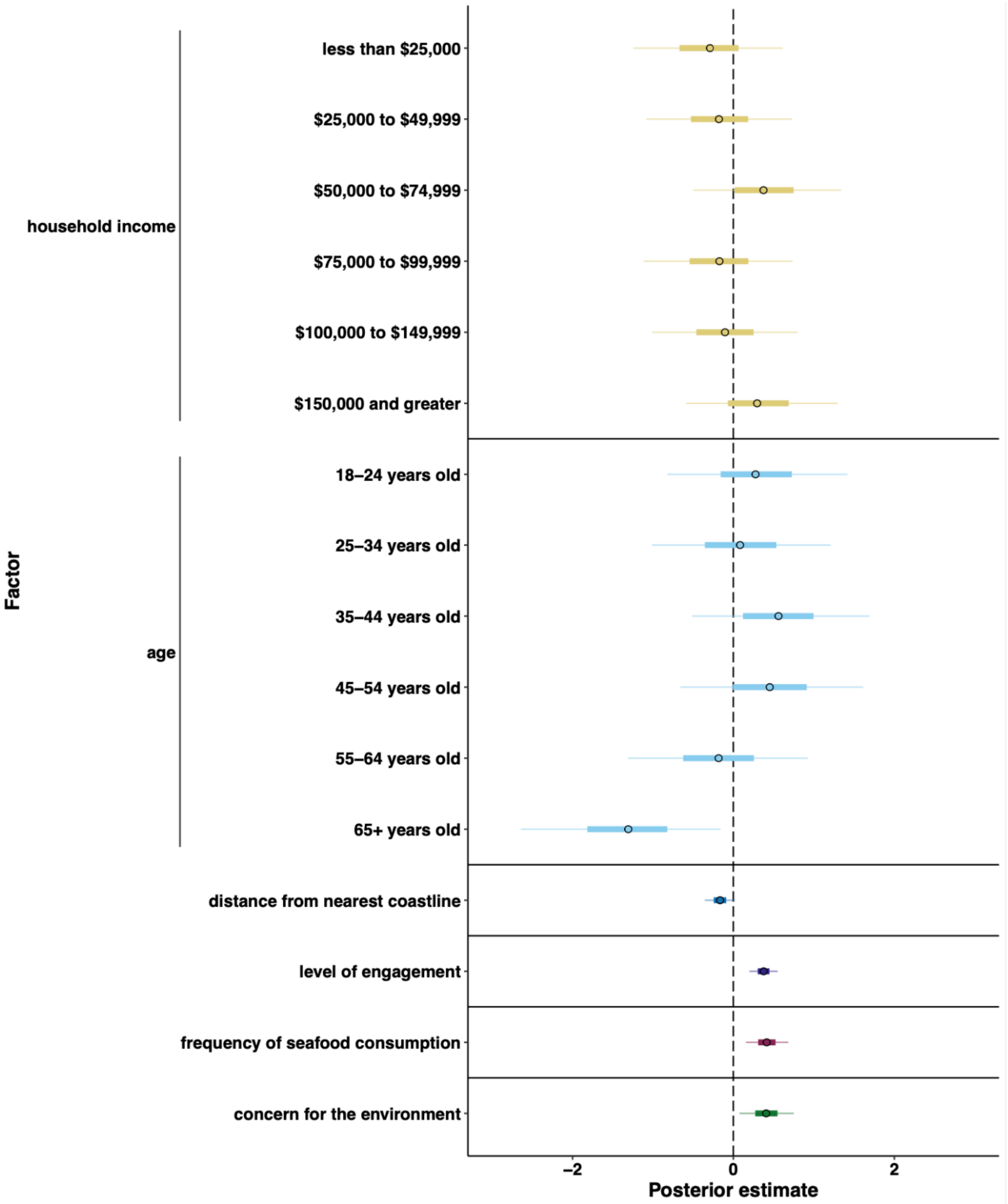


Figure 3.2: Posterior estimates of factors influencing U.S. consumer understanding of seafood ecolabels. Posterior estimates are represented the by median estimate (point) as well as the 50% confidence interval (CI) (thick segment) and 90% CI (thin line segment). The colors correspond to the various factors including in the model.

Willingness to pay for seafood products with ecolabels

Although most consumers (63.13%) believed ecolabelled seafood products should not cost more than their non-ecolabelled counterparts, most consumers indicated they were somewhat (52.32%) or very willing (15.45%) to spend more on ecolabelled seafood products (Figure 3.3A). When we examined willingness to pay across general age categories (Figure 3.3B), most consumers who were not willing to spend more on ecolabelled seafood products were 55 and older (51.37%). Consumers of the 18-34 and 35-54 age categories comprise nearly 85% of consumers very willing to spend more for ecolabelled seafood products. Relatively few consumers who were 55 and older were very likely to spend more on ecolabelled seafood products (15.71%).

A

Category	Number of consumers	Percent of consumers
Should eco-labelled seafood products cost more?		
No	286	63.13%
Yes	167	36.87%
How willing are you to spend more on eco-labelled seafood products?		
Not willing	146	32.23%
Somewhat willing	237	52.32%
Very willing	70	15.45%

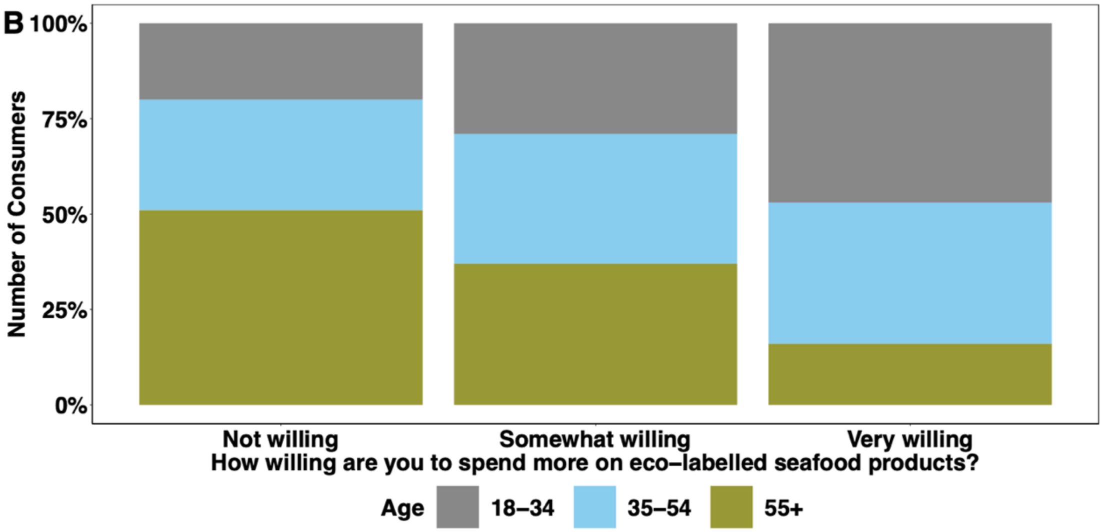


Figure 3.3: Willingness to pay for ecolabelled seafood products among U.S. consumers. A. Consumers responses to pricing and willingness to pay for ecolabelled seafood products. B. Willingness to pay among U.S. consumers of ages 18-34 (grey), 35-54 (blue), and 55 and older (green).

Discussion

Seafood ecolabel schemes are variable in their intent, and typically focus on different aspects of fisheries management and marine conservation. The extent to which any given ecolabel motivates consumers to make purchasing decisions that reflect these aspects of management and conservation is a direct consequence of consumer familiarity and understanding of ecolabeling schemes. That is, the consumers' responsiveness to ecolabelling largely hinges on their understanding of seafood ecolabels as MBSA tools for discerning sustainable seafood. In this study we conducted a nationwide online consumer survey to investigate U.S. consumers' understanding of seafood ecolabels. In general, we found that most consumers (~80%) had limited ability to discern between seafood ecolabels. However, consumers who live closer to the coast, who are more concerned for the future of the environment, and who consume seafood regularly, were more informed of seafood ecolabeling schemes. Additionally, those consumers informed by a diversity of information providers (e.g., media, retailers, fishers) were better able to discern ecolabels. Older (65 and older) U.S. consumers exhibited the least understanding of seafood ecolabels, which possibly explains why they are the least willing to pay for ecolabelled seafood products. These findings demonstrate education efforts to enhance consumer awareness and comprehension of seafood ecolabels will be most effective through targeted demography and geography.

U.S. consumers' understanding of seafood ecolabels

The majority of the U.S. consumers surveyed in our study self-reported as unfamiliar with seafood ecolabels and rating programs, which aligns with past peer-reviewed studies (Feucht & Zander, 2014; Lawley et al., 2019; Thøgersen et al., 2010; Wessells et al., 1999). Of those that self-reported as familiar with such MBSA tools, most consumers (~80%) had limited or no understanding of seafood ecolabels, meaning they did a poor job of discerning between seafood ecolabels. Among the seafood ecolabels included in this study, consumers found it easiest to discern the Dolphin Safe ecolabel as opposed to the MSC or Friend of the Sea ecolabels. These findings are likely attributable to the Dolphin Safe label's clear visual depiction (a picture of a dolphin) and its more focused scope compared to the other labels.

Individuals who regularly consume seafood, who are more concerned for the future of the environment, live closer to the coast, and engage with a diversity of information providers had a higher understanding of seafood ecolabels. Individuals residing in coastal areas often have a higher consumption of seafood due to the convenience and availability of local marine resources (Love et al., 2020), which may increase exposure, interest, and thus understanding in seafood ecolabels. While our results don't show a strong correlation (Supplemental Figure 3.1), proximity to the coast has been shown to be linked to an overall concern for the environment in the context of climate change, potentially attributed to experiencing or anticipating the impacts of climate change firsthand (Milfont et al., 2014). Similarly, proximity to the coast may increase exposure, reliance, and interest in the seafood production systems through firsthand experience with recreational fishing, dining at restaurants that serve local (sustainable) seafood and engaging in direct sales of seafood (purchasing seafood directly from fishers). This direct engagement with local seafood production systems may explain the increased ability to discern seafood ecolabels amongst consumers who reside closer to the coast. Finally, consumers who engaged with multiple information providers exhibited higher levels of understanding of seafood ecolabels. This may be attributed to the diverse perspectives gained, leading to a more comprehensive understanding of seafood ecolabel schemes.

Our findings indicated understanding of seafood ecolabels generally decreased with age. Other studies found a similar pattern with age in relation to the understanding the meaning of seafood sustainability (Winson et al., 2022) and demand for ecolabelled seafood products (Galati et al., 2021; Mulazzani et al., 2019). Young people (ages 18-34) have an increased concern for the future of the environment (Supplemental Figure 3.2) and perceived consumer responsibility (belief that as consumers they have an obligation to engage in responsible and ethical purchases)

(Supplemental Figure 3.3), which could explain their ability to discern seafood ecolabels. Interestingly, in addition to younger age groups having relatively better understanding of seafood ecolabels, they were also more willing to pay for ecolabelled seafood products. Coincidentally, consumers 65 and older who exhibited the least understanding of seafood ecolabels were also the least willing to pay for ecolaebelled seafood products. This correlation between understanding of ecolabels, and willingness to pay for ecolabelled seafood products may derive from the tendency for individuals to be hesitant to spend discretionary income on products they don't fully understand. On the other hand, the reluctance to pay more for ecolabelled seafood products amongst older consumers could also be attributed to these individuals being in or approaching retirement and thus potentially operating on a fixed budget that limits discretionary funds and thus willingness to pay. Further research aimed at understanding the apparent interaction between consumers' understanding of seafood ecolabels and willingness to pay for ecolabelled products would help efforts to increase the effectiveness of ecolabels as a MBSA tool.

Household income exhibited an interesting bimodal-like pattern, where consumers with household incomes of $50,000-$74,999 and $150,000 or greater had the highest understanding of seafood ecolabels, while consumers with household incomes less than $25,000 and $75,000-$99,999 had the lowest understanding of seafood ecolabels. This pattern could be attributed to shift in personal values influenced by age and household income, particularly evident in milestones like purchasing a home, starting a family, or retirement (Lobaugh et al., 2021), which can limit discretionary funds and thus exposure to markets that prioritize sustainability (Foster, 2014). In fact, the median income of first-time home buyers in the U.S. in 2022 was $88,000, one of the household income categories with the lowest understanding of seafood ecolabels. With limited discretionary funds, consumers are less likely to actively search for and pay a premium for ecolabelled seafood, especially when more affordable, non-certified seafood products are readily available in stores. Consequently, these consumers may have limited exposure to and subsequently understanding of seafood ecolabels.

Consumer education and outreach strategies

Engagement with information providers positively influenced U.S. consumers' understanding of seafood ecolabels. We found that U.S. consumers predominantly relied on retailers and media to acquire information on sustainable seafood. These findings underscore the most effective channels ecolabel schemes can leverage to raise consumer awareness and understanding their seafood ecolabel's focus and intent.

Retailers represent a direct line of communication with consumers, particularly through fish mongers and seafood deli personnel who provide essential information about the seafood sold in-store (*The Seafood Shoppers Have Spoken*, 2023). Retailers ranked among the top three information providers across all age categories and were the primary information provider for consumers aged 55 and older, who exhibited the least understanding of seafood ecolabels. Based on these findings, efforts to improve ecolabel scheme understanding (and by association trust and willingness to pay) should prioritize implementing training initiatives for retail staff of businesses that carry their ecolabelled seafood products. Seafood ecolabel schemes offer resources for seafood buyers to make informed decisions about their seafood sourcing (FOS, 2024; MSC, 2024). However, our findings that most U.S. consumers have a limited understanding of seafood ecolabel suggests this knowledge does not seem to extend to those employees who engage with consumers and provide guidance for consumers about their in-store seafood purchases.

Media also emerged as an important information provider consumers rely on to gain knowledge about sustainable seafood, suggesting ecolabel schemes should leverage this communication channel to convey information about their intent and focus. While our findings suggest consumers of most ages rely heavily on media for information on sustainable seafood, the preferred form of media typically varies across ages. Young people greatly depend on social media (e.g., Instagram, TikTok, etc.), for their daily news consumption while older individuals rely more heavily on network and cable news and online-only news sites (Statista, 2024). Accordingly, ecolabel schemes should consider leveraging these media platforms, especially those preferred by older consumers since they exhibited the least understanding of seafood ecolabels.

Another information provider not covered in this study is the hospitality sector (e.g., restaurants, hotels, catering, etc.), which offers another channel for direct engagement with consumers. In response to the growing trend of seafood consumption, restaurants are expanding their seafood selections on menus and improving their seafood sourcing standards (REF). Similar to retailers, many establishments in the hospitably sector have adopted the approach of relying on MBSA tools to inform their seafood buying practices (Roheim et al., 2018). This in turn has led to a comparable landscape seen in the aisles of retailers now on restaurant menus. Consequently, ecolabel schemes should also prioritize implementing training initiatives for workers in the hospitality sector, such as chefs and restaurant servers, who also engage with consumers when they are making seafood purchasing decisions.

Conclusion

Seafood ecolabels are intended to incentivize sustainable fishing practices by increasing the market value of sustainably produced seafood. However, capitalizing on this value requires that consumers recognize the meaning of ecolabels, and are able to distinguish among the differing labelling schemes. This is particularly true given that each ecolabeling scheme differs in the underlying sustainability values they encapsulate. Based on our survey results, most U.S. consumers are currently uninformed of the differences between major seafood ecolabel schemes, even though most have trust in such schemes and are willing to pay more for ecolabelled products. The challenge, then, is to improve general understanding and discernment of such schemes, so that consumers can properly affix their values to the seafood ecolabeling schemes that best reflect them. Doing so will both increase the sustainability impact of ecolabeling schemes, and the market incentives they aim to capture. Our findings clearly demonstrate that education efforts will be most effective through targeted demography and geography (older individuals in non-coastal states), and via retail suppliers. That said, individuals that made use of a diversity of information providers were more informed; thus, outreach in a variety of forms will ultimately enhance consumer awareness and understanding of seafood ecolabels.

Acknowledgements

Chapter 3, in part is currently being prepared for submission for publication of the material. Semmens, Brice X. The dissertation author was the primary researcher and author of this material.

Supplemental Online Questionnaire

Introduction:

Thank you for taking the time to complete this survey. Your responses will help to inform a national study on consumer awareness and willingness to pay for environmentally friendly seafood. Please note that all your responses will be kept confidential. If you are interested, the results of this study will be described in a published, peer-reviewed journal article in the future.

Part 1: Screening question

1. In general, how frequently do you consume seafood (canned, fresh, or frozen)?
    - ○ Not at all--if selected opt respondent out of survey
    - ○ Infrequent (1-2 days a week)
    - ○ Moderate (3-4 days a week)
    - ○ Frequent (5-6 days a week)

Part 2. Current seafood purchasing behavior

2. What factors influence your decision to purchase a particular seafood product over another? Select up to 5 factors.
    - ○ Form of seafood (canned, fresh, frozen)
    - ○ Wild-caught versus farmed
    - ○ Brand
    - ○ Price
    - ○ Nutrition (Non-GMO, organic)
    - ○ Type of seafood (species)
    - ○ Presence of ecolabel (environmentally friendly)
    - ○ Fishing method (if wild-caught, e.g., pole and line)
    - ○ Fishing location (if wild-caught e.g., U.S. waters)
    - ○ Fairly traded

3. Rank the selected factors in order of importance, where 1 is the most important and 5 is the least important. –only provide those factors selected in previous question

4. When shopping for seafood, what details do you notice on the product's label? Click the details in the label below in the order of which you notice them.

5. Which response best describes your feelings about the following statement?

"As a consumer my seafood purchases affects the health of the environment"

1 (Strongly disagree) -- 2 (Somewhat Disagree) -- 3 (Neither agree nor disagree) -- 4 (Somewhat agree) --5 (Strongly agree)

Part 3. Awareness and position towards the environmental impacts of seafood production

6. Which of the following best describes your feelings about the future of our environment?

1 (Not at all concerned) -- 2 (Slightly concerned) -- 3 (Somewhat concerned) -- 4 (Moderately concerned) --5 (Very concerned)

7. Select all the statements that you believe describe wild-caught seafood appropriately.
    - ○ Seafood that comes from the natural habitat (ocean)
    - ○ Seafood that comes from regularly monitored fish populations.
    - ○ Seafood that comes from large tanks on land, or pens (cages or nets) that are floating or anchored to the sea floor.
    - ○ Seafood caught using many different fishing methods that involve pole and line, hooks, large nets, traps, etc.

8. Which of the following best describes your feelings about the effects of wild-caught fisheries on the environment? -- only provided to those respondents who are knowledgeable about wild-caught seafood production (select ⅔ correct answers in the previous question).

1 (Not at all concerned) -- 2 (Slightly concerned) -- 3 (Somewhat concerned) -- 4 (Moderately concerned) --5 (Extremely concerned)

Part 4. Awareness and trust in seafood ecolabels

9. Are you familiar with seafood ecolabels?
    - ○ Yes -- continue to next question
    - ○ No -- skip to Part 4.1

10. Do you rely on seafood ecolabels to help make decisions around your seafood purchases?
    - ○ Yes
    - ○ No-ask why they do not use them as a resource to help make decisions about their seafood purchases and consumption.

11. Are you familiar with seafood rating programs (e.g., Monterey Bay Aquarium: Seafood Watch)?
    - ○ Yes -- continue to next question
    - ○ No -- skip to question 13

12. Do you rely on seafood rating programs to help make decisions around your seafood purchases?
    - ○ Yes

    - ○ No-ask why they do not use them as a resource to help make decisions about their seafood purchases and consumption.

13. Match the logo to the appropriate description.

An ecolabel used to show the seafood product was caught in a way that does not harm dolphins.

An ecolabel used to show the seafood product is not overfished, caught using methods with minimal environmental impacts, and has effective management is in place on the target species caught.

An ecolabel used to show the seafood product is not overfished, uses fishing methods with minimal environmental impacts (on other species and the environment), does not catch any vulnerable, endangered, or threatened species, reduction in carbon footprint, and has effective management in place on the target species.

14. How much do you trust in ecolabels to identify environmentally friendly seafood products?

1 (Strongly do not trust) -- 2 (Somewhat do not trust) -- 3 (Neutral) -- 4 (Somewhat trust) --5 (Strongly trust)

15. In order of importance, with 1 being the most important, list the factors that influence your trust in ecolabels to identify environmentally friendly seafood products.
    - ○ Open-ended

Part 4.1

What are seafood ecolabels?

Ecolabels are labels found on (seafood) products that help consumers quickly identify products that meet environmental standards (e.g., have minimal impact on the environment) and are therefore considered “environmentally preferable”. There are many different ecolabels, all of which vary in their standards. Because ecolabelled products are considered “environmentally preferable” they are sold at a price premium, sometimes double the price of a comparable seafood product without an ecolabel. Here are some examples of ecolabels you may see at your local grocery store:

The “Dolphin Safe” ecolabel ensures that fishing gear was not intentionally deployed on or used to encircle dolphins during the fishing trip and that no dolphins were killed or seriously injured during fishing.

The “Friend of the Sea” ecolabel ensures products come from a population that is not overfished, caught using environmentally friendly methods, and have a reduced carbon footprint.

Part 5. Engagement with stakeholders in the sustainable seafood network

16. From which stakeholder(s) listed below do you get your information from on environmentally friendly seafood choices? Select all that apply.
    - ○ Governmental Management organizations (international and/or domestic)
    - ○ Ecolabel companies (e.g., Marine Stewardship Council)
    - ○ Environmental Non-Governmental organizations (NGOs)
    - ○ Seafood Rating Programs (e.g., Monterey Bay Aquarium: Seafood Watch)
    - ○ Seafood Producers/Fishers
    - ○ Retailers/Grocers
    - ○ Media
    - ○ Friends
    - ○ None of the above -- skip to Part 6

17. To get the information about environmentally friendly seafood, how do you interact with each stakeholder you identified? (Provide the same stakeholders the consumer identified in question 16 so they can characterize engagement with each selected stakeholder).
    - ○ Visit their social media (e.g., website) for educational information
    - ○ Attend educational events, conferences, etc.
    - ○ Interact with employees/individuals
    - ○ Read regular reports, newsletters, etc.
    - ○ Participate in informal outreach (e.g., cooking demonstrations)
    - ○ Other (open answer)

18. How often do you interact with each stakeholder you identified to get the information about environmentally friendly seafood? (Provide the same stakeholders the consumer identified in question 16 so they can characterize engagement with each selected stakeholder).
    - ○ Infrequent (1-5 times a month)
    - ○ Moderate (6-10 times a month)
    - ○ Frequent (11+ times a month)

Part 6. Willingness to pay for sustainable seafood

19. Do you think seafood products with ecolabels should cost more?
    - ○ Yes -- continue to next question
    - ○ No -- skip to Part 7

20. How much more are you willing to spend on a seafood product with an ecolabel?
    - ○ Provide a percentage bar (0-100%)

Part 7. Demographics

21. Age: What year were you born in?

  - ○ Drop down menu

22. Which state do you live in?
    - ○ Drop down menu

23. Gender: To which gender do you most identify?
    - ○ Male
    - ○ Female
    - ○ Other
24. Ethnicity: To which ethnicity/ethnicities do you most identify?
    - ○ American Indian or Alaska Native
    - ○ Asian
    - ○ Black or African American
    - ○ Hispanic or Latin
    - ○ Native Hawaiian or Pacific Islander
    - ○ White
25. Education: What is the highest degree or level of school you have completed?
    - ○ High school diploma
    - ○ College degree (2-4 years)
    - ○ Advanced degree (PhD, MS, MD, etc.)
26. Political Affiliation: How would you describe your political view?
    - ○ Very Liberal
    - ○ Slightly Liberal
    - ○ Moderate
    - ○ Slightly Conservative
    - ○ Very Conservative
    - ○ Prefer not to answer
27. Annual income: Which of these describes your household income in 2019?
    - ○ <$10,000
    - ○ $10,000 to $24,999
    - ○ $25,000 to $49,999
    - ○ $50,000 to $74,999
    - ○ $75,000 to $99,999
    - ○ $100,000 to $149,999
    - ○ $150,000 and greater
    - ○ Prefer not to answer

Supplemental Table 3.1: Full model specification of the best performing model (base model)

$$\boldsymbol{Ordered\ Logistic(y_i|\eta_{d,e,a,hi,cr,ce,sc}, c_{1-K-1}}$$

$$\begin{cases} \boldsymbol{logit^{-1}(c_1 - \eta_i)} & \boldsymbol{Y = No\ understanding} \\ \boldsymbol{logit^{-1}(c_2 - \eta_i) - logit^{-1}(c_1 - \eta_i)} & \boldsymbol{Y = Limited\ understanding} \\ \boldsymbol{logit^{-1}(c_3 - \eta_i) - logit^{-1}(c_2 - \eta_i)} & \boldsymbol{Y = Moderate\ understanding} \\ \boldsymbol{1 - logit^{-1}(c_3 - \eta_i)} & \boldsymbol{Y = High\ understanding} \end{cases}$$

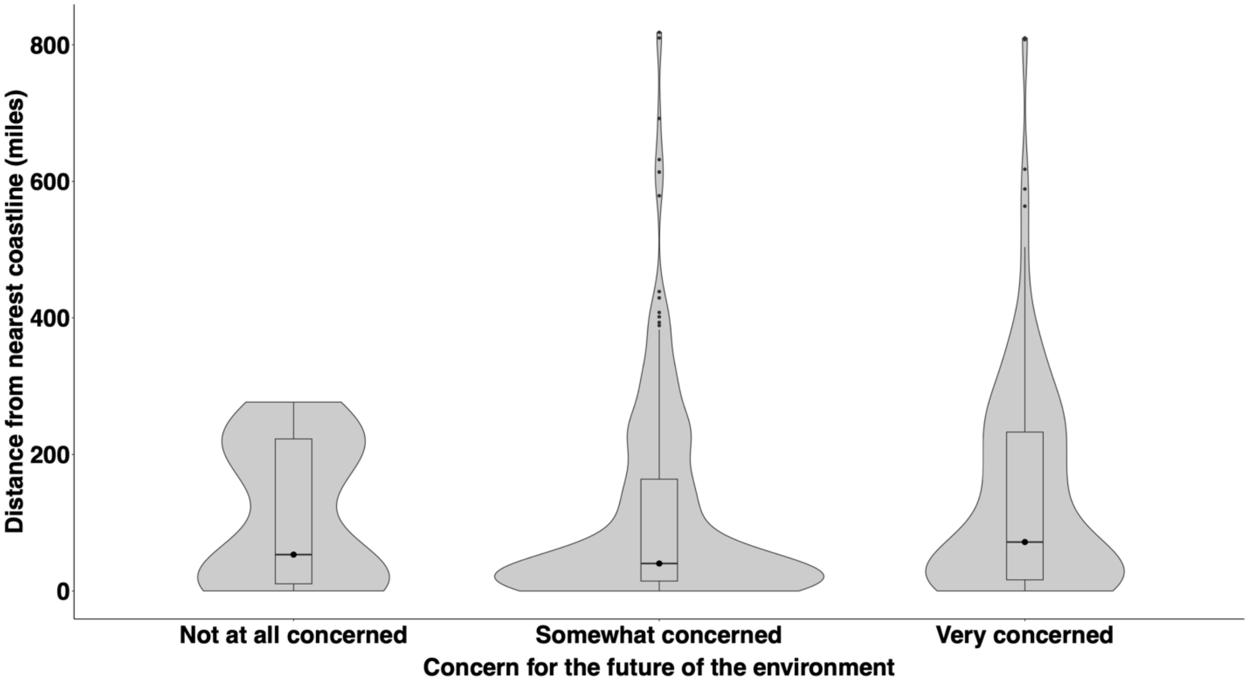
$$\boldsymbol{\eta_{d,e,a,hi,cr,ce,sc,i} = z_d * \sigma_d + z_e * \sigma_e + z_{hi} * \sigma_{hi} + z_{cr} * \sigma_{cr} + z_{ce} * \sigma_{ce} + z_{sc} * \sigma_{sc}}$$

$z_d \sim N(0,0.1)$ [distance from nearest coastline effect deviate]
$z_e \sim N(0,0.1)$ [level of engagement effect deviate]
$z_{hi,1:j} \sim N(\mu_{hi}, \tau_{hi})$ [household income effect deviate]
$z_{cr,1:j} \sim N(\mu_{cr},, \tau_{cr})$ [perceived consumer responsibility effect deviate]
$z_{ce,1:j} \sim N(\mu_{ce}, \tau_{ce})$ [concern for the future of the environment income effect deviate]
$z_{sc,1:j} \sim N(\mu_{sc}, \tau_{sc})$ [seafood consumption effect deviate]

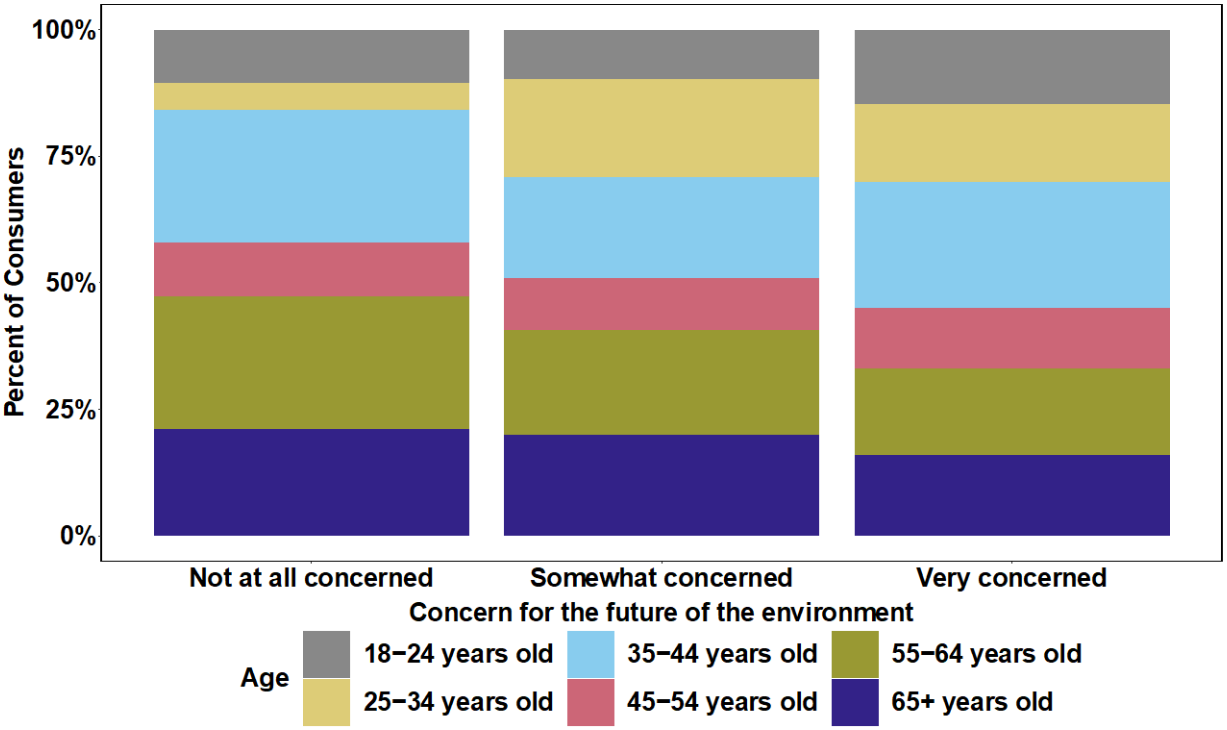
$j = number\ of\ respective\ categories\ for\ each\ factor$

$\tau_{hi} \sim \frac{1}{{\sigma_{hi}}^2}$ [precision parameter for household factor]
$\tau_{cr} \sim \frac{1}{{\sigma_{cr}}^2}$ [precision parameter for consumer responsibility factor]
$\tau_{ce} \sim \frac{1}{{\sigma_{ce}}^2}$ [precision parameter for concern for the future of the environment factor]
$\tau_{sc} \sim \frac{1}{{\sigma_{sc}}^2}$ [precision parameter for seafood consumption factor]

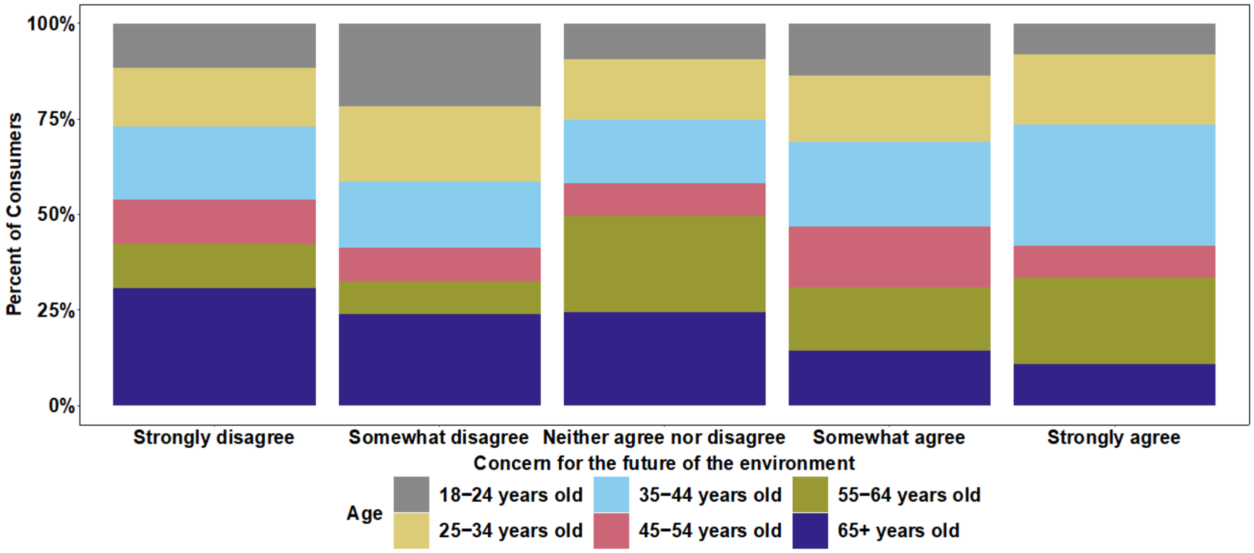
$\boldsymbol{\mu_{hi}, \mu_{cr}, \mu_{ce}, \mu_{sc} \sim N(0, 5)}$
$\boldsymbol{\sigma_{hi}, \sigma_{cr}, \sigma_{ce}, \sigma_{sc} \sim Uniform(0, 5)}$



Supplemental Figure 3.1: Relationship between concern for the future of the environment and distance from nearest coastline (in miles).



Supplemental Figure 3.2: Relationship between concern for the future of the environment and age of U.S. consumers. The colors denote the various age categories.



Supplemental Figure 3.3: Relationship between perceived consumer responsibility and age of U.S. consumers. The colors denote the various age categories.

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