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Cover photo: Research fishing conducted as part of the bottomfish fishery-independent survey in Hawai'i. Photo credit: Pacific Islands Fisheries Group/Lynker Technologies, under contract to NOAA PIFSC

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Table of Contents

List of Tables

List of Figures

Executive Summary

The Hawai‘i Deep 7 bottomfish complex is a U.S. fishery management unit consisting of six deep-water snapper species and a single grouper species that inhabit waters of the Hawaiian Archipelago. All bottomfish fishing currently takes place in the main Hawaiian Islands zone due to the closure of the Northwestern Hawaiian Islands under Proclamation 8031. Stock assessments conducted since 2011 have assessed the Deep 7 bottomfish complex within the MHI zone.

A benchmark stock assessment of the main Hawaiian Islands Deep 7 bottomfish complex was conducted in 2023 and finalized in 2024. Since this is a benchmark assessment, many improvements were incorporated and all components of the assessment were reevaluated. Most of the improvements incorporated were recommended by the Western Pacific Stock Assessment Review (WPSAR) panelevaluations of the 2018 and 2021 assessments. Some significant changes include the re-examination of previously used reporting ratios for determining non-commercial catch, refining data filtering procedures and improving the generation of abundance indices (i.e., the standardized fishery-dependent CPUE index, fishery-independent index, and single-species ‘ōpakapaka CPUE index), updating the software for the assessment model, exploring new parameterizations of the production function, and reevaluating prior distributions based on the most recent life-history information available for Deep 7 species.

The benchmark Deep 7 assessment used a state-space Bayesian surplus production model within the modeling framework Just Another Bayesian Biomass Assessment (JABBA). Estimates of harvest rate (H), annual biomass (B), the harvest rate associated with overfishing as determined by the harvest control rule (H_{CR}), maximum sustainable yield (MSY), and the biomass at maximum sustainable yield (B_{MSY}) allowed for determination of stock status relative to reference points determining overfishing (H/H_{CR}) and overfished ($B<0.867 \times B_{MSY}$) status. Stock status for the main Hawaiian Islands Deep 7 stock in 2023 is not overfished or undergoing overfishing (Figure S1), similar to the 2018 benchmark and 2021 update assessments. A supplementary single-species JABBA model and a structured Stock Synthesis model for the dominant species in the Deep 7 catch (‘ōpakapaka) both produced corroborating results.

Stock projections were conducted for 2025–2029 for a range of hypothetical five year catches, and the corresponding risk of overfishing was calculated. Projections assumed that the same amount of catch was caught from 2025 to the terminal year of the projections. The overfishing limit (OFL), which is defined as the future amount of reported catch that would yield a 50% probability of overfishing ranged from 507–612 thousand pounds depending on the final year. The smallest Deep 7 future catch that would lead to a roughly 40% chance of overfishing was 464 thousand pounds.

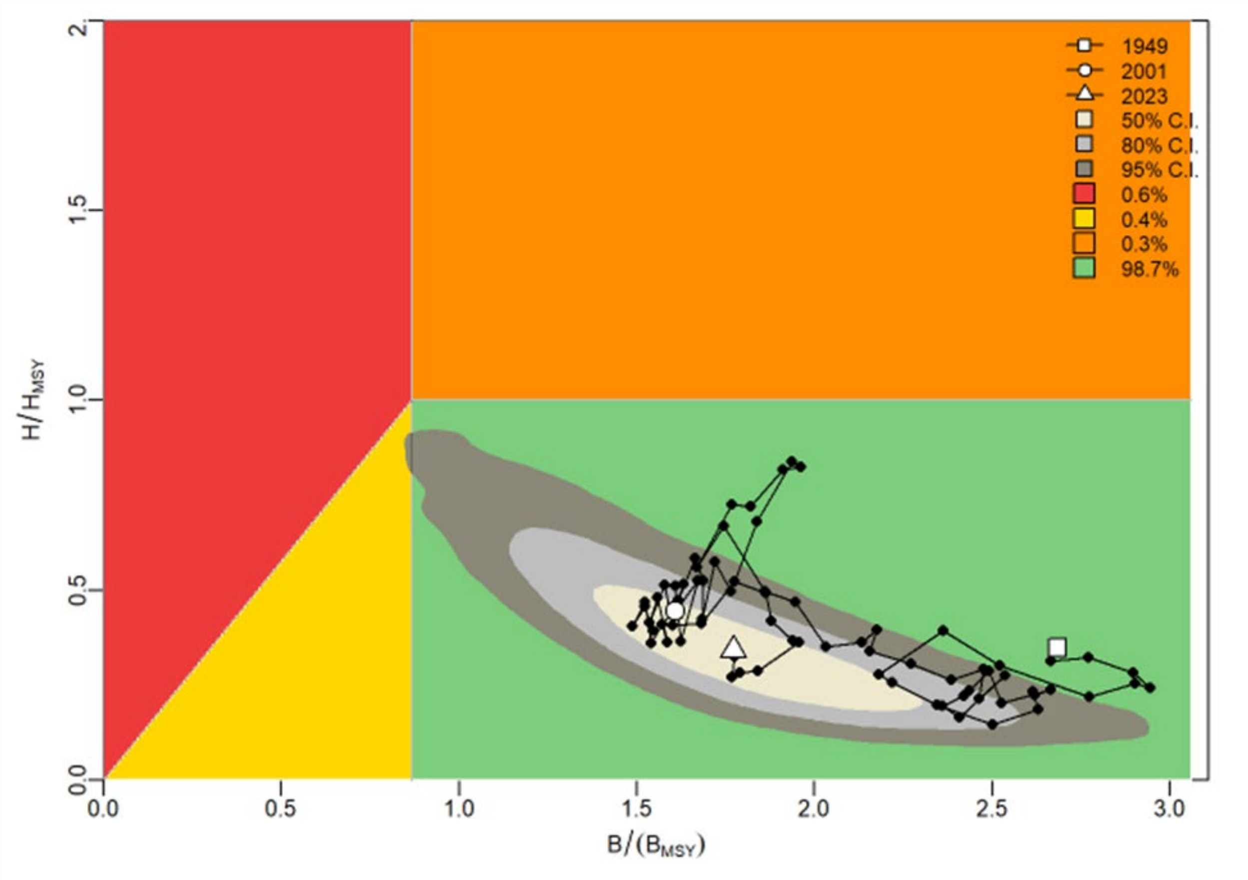

Figure S1. Estimated status for the Deep 7 Bottomfish complex in the main Hawaiian Islands from 1949 through 2023. The white square indicates the first year of the model (1949), the circle indicates the year 70% through the time series (2001), and the triangle indicates the terminal year (2023). The shaded areas represent the 50%, 80%, 95% credible intervals for the terminal year estimate.

1. Introduction

The Hawaiʻi bottomfish complex is a U.S. fishery management unit made up of 13 shallow- and deep-water snapper and jacks species along with a single grouper species that inhabit waters of the Hawaiian Archipelago (Table 1). The ecological niches occupied by the shallow-water and deep-water components of the bottomfish complex differ (WPRFMC 2001). Deep-water bottomfish habitat in the main Hawaiian Islands includes waters of roughly 100–400 m depth (Parke 2007), although some species shoal to mid-water depths to feed.

The bottomfish complex, along with three seamount groundfish species, is managed as a bottomfish management unit species under the *Fishery Ecosystem Plan for the Hawaii Archipelago* (FEP) developed by the Western Pacific Regional Fishery Management Council (WPRFMC 2009). The federal fisheries management regime includes three fishing zones: the main Hawaiian Islands zone, two zones in the Northwestern Hawaiian Islands, the Mau Zone, and the Hoʻomalu Zone. All bottomfish fishing currently takes place in the main Hawaiian Island zone (Figure 1, Figure 2) due to the closure of the Northwestern Hawaiian Islands under Proclamation 8031.

The Deep 7 bottomfish complex, which will henceforth be referred to as the "Deep 7" (Table 1.), comprises a subset of seven species from the bottomfish complex that have been a focus of fishery management measures, including seasonal fishery closures and annual catch limits in the main Hawaiian Islands, since the larger bottomfish complex was determined to be experiencing overfishing on an archipelagic basis in 2005 (Moffitt et al. 2006). This stock assessment report assesses the Deep 7 bottomfish complex within the main Hawaiian Islands zone.

Hawaiʻi bottomfish were targeted by native Hawaiians using deep handlines from canoes for hundreds of years before the advent of the modern fishery after World War II. The modern fishery employs similar handline gear, albeit with braided synthetic line, along with power reels to haul back gear, fish finders to locate schools of fish, and GPS units and other navigational aids to find fishing grounds. Although the efficiency of the modern fishery has likely improved through time (Moffitt et al. 2011), the current Hawaiʻi bottomfish fishery still uses traditional deep handline capture methods for commercial and recreational harvest. Bottomfish restricted fishing areas (BRFAs) were imposed in Hawaiʻi state waters in 1998 and revised in 2006 to conserve fishery resources. Twelve BRFAs were ultimately put into place with the intent to cover consequential areas of bottomfish habitat. Four of these areas were reopened to fishing in 2019 and the remaining eight were reopened in 2022.

Bottomfish resources around Hawaiʻi are managed as one multi-species complex. The FEP establishes methods for determining fishing mortality and stock biomass reference values and—by a comparison of current conditions to the reference values—determining if the stock is being overfished and if overfishing is occurring. Overfished is defined as the stock biomass B falling below the minimum stock size threshold (MSST) of $(1-M)*B_{MSY}$, where M is the natural mortality rate of the complex and B_{MSY} is the biomass that produces the maximum sustainable yield. As was done in the previous assessment, M was set at 0.13, so the overfished definition is defined as $B<0.867*B_{MSY}$. Previous assessments defined overfishing as a fishing / harvest rate H that exceeds the maximum fishing mortality threshold (MFMT) of H_{MSY}, which is the harvest rate that produces maximum sustainable yield.

According to the FEP, the MFMT varies depending on whether biomass is above or below the MSST (Figure 3.; see WPRFMC 2009, pages 143–149 for a description of the harvest control rule). If the stock biomass is above the MSST ($B>0.867*B_{MSY}$), then the MFMT equals H_{MSY}, whereas if the stock biomass falls below the MSST ($B<0.867*B_{MSY}$), then the MFMT declines from H_{MSY} in proportion to the ratio of biomass to the biomass reference point. Throughout this manuscript, we refer to overfishing status in relation to H_{CR} instead of H_{MSY} to reflect the harvest control rule as stated in the FEPs.

Benchmark Stock Assessment in 2018 and 2021 Update

The 2018 benchmark Deep 7 bottomfish stock assessment for the main Hawaiian Islands used a similar assessment methodology as in the 2011 benchmark assessment (Brodziak et al. 2011) and the 2014 assessment update used for management (Brodziak et al. 2014). The 2018 assessment provided stock status determinations for 2015, with projections through 2022.

The baseline assessment model was a Bayesian surplus production model that used updated information on CPUE data along with improved filtering procedures and data analyses. Modifications to data and other improvements to the model structure were incorporated within the 2018 assessment to address both immediate and long-term recommendations raised by the CIE review of the initial 2014 assessment update (Neilson 2015). Improvements included using the results of data workshops to update catch and CPUE analyses, evaluation of assumed prior values that provided greater support for chosen model parameters, and developing a single-species assessment for the dominant species in the catch, ʻōpakapaka. In addition, an absolute biomass estimate based on an estimated scalar and relative biomass estimate from the fishery-independent survey in the MHI was used to scale biomass estimates within the model. Additional details on these improvements can be found in the 2018 stock assessment report (Langseth et al. 2018).

The 2021 assessment update was constrained to using the same data sources and methodology as the 2018 benchmark stock assessment under the framework of the Western Pacific Stock Assessment Review. The 2021 assessment included an additional three years of data for each source, providing stock status determinations for 2018 and projections through 2025.

Current Benchmark Stock Assessment

A Bayseian surplus production model that explicitly accounted for both process and observation errors was used for the 2024 benchmark stock assessment, as was done in previous assessments. The assessment directly estimated MSY-based reference points as well as trajectories and projections of biomass and harvest rate to determine stock status of Deep 7 complex. However, unlike previous assessments, a new modeling software was used to run the production model and estimate the trajectory and status of the Deep 7 complex. In addition to updated modeling software, the 2024 benchmark stock assessment included a number of changes from the 2018 benchmark stock assessment and 2021 update assessment. All of these changes are described in detail throughout this report, and many are the result of the recommendations from the WPSAR review panels for the 2018 and 2021 assessments (see Appendix A for complete lists of recommendations). The WPSAR recommendations can generally be grouped into the following categories:

Catch data

- Re-evaluated the contribution of catch not reported in the State of Hawai'i's Division of Aquatic Resources (DAR) commercial catch records (henceforth, referred to as "unreported catch") to total catch by examining previously used reporting ratios and evaluating data from the Hawai‘i Marine Recreational Fishing Survey (HMRFS).

CPUE data and standardization

- Refined data filtering procedures, evaluated a single time series for fishery-dependent CPUE, and improved the generation of the index during the CPUE standardization process.
- Developed a relative index of abundance for the fishery-independent survey using a spatiotemporal modeling framework to overcome sensitivity of results to the assumed effective radius of the camera gear.
- Explored alternative classification models (i.e., regression trees) for identifying trips targeting ‘ōpakapaka to calculate CPUE for the single-species modeling approach.

Modeling

- Explored alternative software tools for conducting this stock assessment to overcome limitations of the existing OpenBUGS platform.
- Explored alternative parameterization of the surplus production model that allows the B_{MSY}/K ratio to fall below values of 0.378.
- Continued to address the life-history differences in the Deep 7 that could result in changes in the prior distributions for this stock complex.

2. Methods

In this section, basic information on data sources used in the 2024 MHI Deep 7 bottomfish assessment is described including biological information (2.3 Biological Data), fishery catch (2.4 Fishery Catch), fishery CPUE for standardization (2.5 Standardized Fishery Catch per unit effort), and the fishery-independent survey (2.6 Fishery-independent Survey).

2.1. Fishing Year

The 2024 benchmark assessment used the same annual time period for reporting bottomfish catch as in previous assessments. Catch and CPUE data were reported annually from July 1 of the previous year through June 30 of the current year, which is defined as the fishing year. This fishing year coincides with the State of Hawaiʻi's fiscal year and commercial marine license period. However, this differs somewhat from the definition of fishing season in the bottomfish fishery management plan, which extends from September 1 of the previous year through August 31 of the current year. The fishing year beginning on July 1 corresponds to the annual biological cycle of the Deep 7 bottomfish complex, which spawns in late spring to early summer (DeMartini 2016). Estimates of annual production biomass starting in July coincide with the settlement of juvenile bottomfish through midsummer (DeMartini et al. 1994). More importantly, the commercial fishery catch of Deep 7 bottomfish is typically highest during the winter months when there is strong market demand for red-colored fish during New Year holidays, and, therefore, is not operationally separated on a calendar year definition.

2.2. Data Sources

Biological data—along with catch, CPUE, and survey biomass data—were used to assess the main Hawaiian Islands Deep 7 bottomfish complex stock. Catch and CPUE data were derived from Fisher Reporting System (FRS; PIFSC 2023) data collected by DAR. Survey biomass data were derived from data collected during the 2016–2022 Bottomfish Fishery-Independent Survey in Hawai'i (BFISH) conducted by PIFSC in partnership with cooperative research fishers (PIFSC 2022a; PIFSC 2022b). Catch and effort data were available from January 1, 1948, to June 30, 2023, but—because data for 1948 covered only half the fishing year—data starting in fishing year 1949 were used in the assessment model. Details on each individual data component are described in the sections below.

2.3. Biological Data

The quantitative information on the life history parameters of the Deep 7 bottomfish has improved in recent years (Table 2). Surplus production models have relatively few parameters, and there are studies for Deep 7 bottomfish that can be used to infer values for surplus production model parameters, particularly the intrinsic growth rate (*r*) and the shape of the production function. Previous assessments relied on ranges of values for the intrinsic growth rate (Musick 1999) related to estimates of maximum age, age at maturity, and growth.

The current assessment improved on this approach by using an age-structured equilibrium model (ASEM) parameterized with published life history values (Table 2) to generate informative Bayesian priors for the intrinsic growth rate (*r*) and production shape parameter (*m*), following Winker et al. (2020a and b). The ASEM uses life history parameters in an age-structured model to obtain estimates of *r* and the proportion of unexploited biomass present at maximum sustainable yield (B_{MSY}/B_0, referred to as the "biomass ratio" in this text), which can then be converted into *m*. The details of the ASEM approach can be found in Winker et al. (2020a and b).

The ASEM requires the following parameters: growth (von Bertalanffy L_{inf}, K, A_0; length-weight A and B), maturity (lengths at 50% and 95% maturity), longevity (A_{max}) and its associated natural mortality (M), stock-recruitment steepness (h), and fishery selectivity (lengths at 50% and 95% selectivity). Growth, maturity, and longevity parameters were obtained from published life history studies (Table 2), and M was estimated from A_{max} following Hamel and Cope (2022). Stock-recruitment steepness was obtained from the FishLife meta-analytical tool (Thorson et al. 2017).

For all species except ʻōpakapaka, fishery selectivity was estimated by fitting a length-based spawning potential ratio model (LBSPR; Hordyk et al. 2016) to the size composition data for each species from the research fishing component of the fishery-independent survey (Richards 2023). For ʻōpakapaka, the estimate of fishery selectivity for the commercial fishery from a single-species Stock Synthesis model (Supplemental Material A) was used. We included uncertainty in natural mortality (Coefficient of Variation (CV) of 0.2) and the steepness parameters (Standard Deviation (SD) in *h* among the seven species = 0.06) using a Monte Carlo simulation that included 100 natural mortality random draws and 100 steepness values for each *M* draw, for a total of 10,000 iterations per species.

The distributions of *r* and the biomass ratio were combined into a single dataset for all seven species resulting in a dataset with 70,000 iterations (7 species x 10,000 iterations). This dataset was bootstrapped 100,000 times with replacement and sampling proportional to the prevalence of each species in the total catch. This generated a final dataset with 100,000 observations of each parameter (*r* and biomass ratio) for the Deep 7 complex. The final step was to fit a log-normal distribution to this outputted dataset for both *r* and the biomass ratio, using the R package MASS

(Venables and Ripley 2002). The resulting distributions for both r and B_{MSY}/B_0 can be found in Figure 4 and Figure 5, respectively.

For the 2018 benchmark stock assessment, a value of natural mortality consistent with expected longevity (Then et al. 2015) was used, as suggested by reviewers from the previous assessment update (Neilson 2015). The 2024 benchmark assessment used the Then et al. (2015) longevity estimator corrected for lognormal bias (Hamel and Cope 2022), which describes the relationship between natural mortality (M) and maximum age (A_{max}), M=5.4/A_{max}. Species-specific estimates of A_{max} were used to estimate M, and the estimates of M were combined among Deep 7 species as a weighted average with weight determined by the species-specific relative abundance in the total catch (Table 3). The resulting estimate of natural mortality was 0.13, and this value was used for the purposes of determining a minimum stock size threshold (see Introduction).

2.4. Fishery Catch

Catch data for the 2024 assessment included a combination of reported catch data as well as estimates of unreported catch. Unreported catch based on estimates of unreported to reported catch ratios were calculated prior to use within the assessment model. Reported catch and unreported catch were added together to determine total catch for Deep 7 bottomfish in the main Hawaiian Islands. Details on each component of catch are provided in sections 2.4.1 Reported Catch of Deep 7 Bottomfish to 2.4.3 Estimates of Total Bottomfish Catch.

2.4.1. Reported Catch of Deep 7 Bottomfish

Reported fishery catch data used in the model were based on Deep 7 bottomfish catch data extracted from approximately 5.2 million DAR catch records submitted by fishers during fishing years 1949–2023 (PIFSC 2023). A subset of the records was used to calculate reported Deep 7 bottomfish catch in weight, based on methods agreed upon at the data workshops (Yau 2018).

First, catch data for Deep 7 bottomfish species (Table 1) were separated from all other species based on species codes reported within the catch data set. There were two species codes for ehu (*Etelis carbunculus*) in the data set (Moffitt et al. 2011), so both were used and combined into a single code. Second, catch data of Deep 7 bottomfish species were assigned to the main Hawaiian Islands and the Northwestern Hawaiian Islands fishing zones based on the reported DAR fishing areas in the data set (Figure 2). Some (170) records of Deep 7 bottomfish catches were reported in unknown or invalid fishing areas, and the minor catch amount (23,086 lb) from these records was prorated to the main Hawaiian Islands fishing zone based on the percentage of Deep 7 bottomfish caught annually by species in known areas of the main Hawaiian Islands compared to known areas of both the main Hawaiian Islands and Northwestern Hawaiian Islands fishing zones. The final reported catch of Deep 7 bottomfish in the main Hawaiian Islands was tabulated by fishing year and species during fishing years 1949–2023 (Table 3).

2.4.2. Estimates of Unreported Bottomfish Catch

Currently, there is no directed long-term monitoring program in place for quantifying the amount of catch not reported in DAR catch records of bottomfish in the main Hawaiian Islands. However, the Hawaiʻi Marine Recreational Fishing Survey (HMRFS) was initiated in 2001 and expanded to five of the main Hawaiian Islands in 2003. The HMRFS provides estimates of species-specific catch statewide through the combination of intercept surveys used to estimate catch rates and a mail survey (previously a telephone survey) used to estimate statewide fishing effort (Ma et al. 2023).

The HMRFS catch estimates have been improved for use in this assessment (Ma et al. 2023). Briefly, the HMRFS estimation algorithm was updated to estimate only unsold catch, a correction factor was applied to the statewide effort estimates to account for the shift from landlines to cell phones prior to the implementation of the mail survey in 2018, and a smoother was used to reduce variation in catch rate among sampling waves (i.e., 2-month periods) in the intercept survey (Ma et al. 2023). Despite the application of the smoother, annual estimates of unsold catch from the HMRFS were still considered sufficiently variable to present concerns for estimating harvest rate in a given year against the MFMT. Therefore, the species-specific ratio of unsold catch estimated by the HMRFS to sold catch reported by CML holders was calculated for each year from 2003–2022, and the average species-specific ratio among years was used to calculate unreported catch by species and year. It should be noted that ratios provided in Ma et al. (2023) were calculated by calendar year and differ slightly from those presented in Table 4, which were calculated by fishing year.

The approach to including unreported bottomfish catch for 1949–2003 was based on estimated ratios of unreported to reported catch as summarized by Courtney and Brodziak (2011), which were used in previous assessments (Brodziak et al. 2011 and 2014; Langseth et al. 2018; Syslo et al. 2021). However, we determined that the ratios previously used were developed using reported catch from the entire Hawaiian archipelago to estimate (scale up) unreported catch for the main Hawaiian Islands and were therefore biased high. The approach to correct the reporting ratios used for 1948 to 2003 is described in Appendix B.

Estimates of the unreported catch ratio U (Table 4) indicated that unreported catch (Table 5.) was similar in magnitude to the reported commercial catch from 1949–2003 (U=0.99). Overall, the average ratio of noncommercial catch from the HMRFS to reported catch ratio during 2004–2022 was U=1.09 and the average unreported catch from 2019–2023 was approximately 202 thousand pounds. The survey of bottomfish fishers conducted by Hospital and Beavers (2014) reported on the disposition of the catch, and their data indicated that the ratio of not sold to sold was 1.33 for commercial fishers.

Uncertainty in estimates of unreported catch was included as sensitivity analyses. The first scenario excluded variation in total catch from the JABBA model altogether and a second scenario used annual HMRFS estimates and CVs (Table 6) in the calculation of total catch and the associated variability for 2004 through 2023. Details on each of these alternative scenarios are provided in section 2.9 Sensitivity Analyses.

2.4.3. Estimates of Total Bottomfish Catch

The total catch of Deep 7 bottomfish in the main Hawaiian Islands was the sum of reported and unreported catch (Table 8). Uncertainty in the amount of unreported bottomfish catch was taken from the average CV from the HMRFS estimates (Table 6). To account for uncertainty in estimates of unreported bottomfish catch, it was assumed that there was an independent error distribution for each annual estimate of unreported catch for fitting parameters of the production model used in the stock assessment. As for previous Deep 7 stock assessments, we assumed no underreporting of commercial catch. Therefore, the CV of total catch was reduced by 50% because the available estimates of the ratios of estimated noncommercial to reported commercial catch were near 1.0. The error distribution for catch is described in further detail in section 2.7.3 Prior Distributions.

2.5. Standardized Fishery Catch per unit effort

Estimation of standardized commercial fishery CPUE for Deep 7 bottomfish continued to be improved over methods used for previous stock assessments. The 2018 benchmark stock assessment used information from an extensive series of stakeholder-focused workshops to aid in interpreting the commercial fishery CPUE data (Yau 2018). The 2018 workshop report continued to provide the framework for data filtering and CPUE standardization used in this assessment. However, continued data exploration and a two-day workshop with collaboration from the state of Hawaiʻi and the fishing industry resulted in additional improvements to CPUE standardization in the 2024 benchmark assessment.

2.5.1. Fishery Data for use in CPUE Analyses

As in previous assessments, fisher reported data were used for standardizing CPUE indices of abundance (PIFSC 2023). Fisher reported catch and effort data from fishing years 1948–2023 were used in this stock assessment to calculate standardized indices of abundance. Previous benchmark assessments separated CPUE into two periods to address the availability of fisher-specific information on license number (Brodziak et al. 2014) and to account for the hours fished being available as a measure of effort after 2002 (Langseth et al. 2018). Previous limitations were overcome in the 2018 assessment and individual fishers were able to be tracked throughout the entire time series (Langseth et al. 2018). However, information on fishers was not available for all records in years prior to 1976 and fisher name information for records in 1976 could not be located (Table 8.).

Additionally, participants of a stakeholder workshop held in 2023 indicated that the number of hours fished was not consistently reported over the past decade as fishers transitioned from reporting the total duration of a trip to reporting only the soak time of fishing gear. This explanation matched with a consistent pattern of average hours per trip declining from about 7.3 hours in 2014 to about 6 hours in 2022. In contrast with the 2018 benchmark stock assessment, a single CPUE series was estimated using reporting day as the unit of effort. Although the number of days fished was not always known with certainty (see below), it was considered a better alternative to using hours fished, which contained a known bias.

2.5.2. Fishery Data Filtering Steps for CPUE

Improved data filtering procedures were discussed at the 2018 data workshops (Yau 2018) and an agreed-upon approach was used for the 2018 and 2021 assessments to select data for standardizing indices of CPUE. In brief, Deep 7 bottomfish catch per effort data from 1948–2023 were summarized for directed deep-sea handline fishing in the main Hawaiian Islands while accounting for potential multi-day trips. Multi-day trips were a large concern among reviewers in stock assessments before 2018.

Procedures for preparing fisher reported data for CPUE are described below in four steps, as outlined in the data workshop report (Yau 2018). The four steps included (1) selecting records targeting Deep 7 bottomfish, (2) analyzing records to account for multi-day trips, (3) selecting records representative of the fishery, and (4) preparing the data to incorporate factors affecting CPUE. Each step is described below, and each was applied sequentially to the data.

2.5.2.1. Selecting records targeting Deep 7 bottomfish

The first step in preparing fisher reported data for CPUE analysis was to remove any records not targeting Deep 7 bottomfish. The FRS database did not indicate what records targeted Deep 7 bottomfish, so filtering procedures were used to select records considered to target Deep 7 bottomfish within the spatial and temporal range of the assessment.

Gear was identified as a critical determination for bottomfish fishing. Among records reporting Deep 7 bottomfish catch, 95% of the records and 98% of the fish weight summed over the records occurred with deep-sea handline gear. Fishers catching bottomfish primarily use and report this gear. Of the 940,400 records from 1948 to 2023 that reported deep-sea handline gear, 884,057 records were retained after filtering for records within the main Hawaiian Islands. The definition of main Hawaiian Islands areas for this stock assessment was consistent with the 2018 and 2021 stock assessments (Figure 2).

After initial filtering procedures for gear and location, fisher reported data were next filtered to remove records not targeting Deep 7 bottomfish. Henceforth, the term “fishing event” is used to describe a set of records for a unique commercial marine license (CML) number associated with a given unit of effort. The effort metric is referred to as “single-reporting days” to avoid the use of the term “trip,” which is commonly defined by a fisher coming in and out of port. This definition of fishing event may result in overnight fishing being split into two fishing events for the purpose of CPUE calculation.

The definition of what constituted Deep 7 bottomfish fishing was discussed at length at the data workshops (Yau 2018). Prior to the 2018 assessment, a cutoff point (17%) based on the weight of Deep 7 bottomfish caught in a single-reporting day was used to determine targeting of Deep 7 bottomfish (Brodziak et al. 2011). This removed all single-reporting days with less than 17% pounds of Deep 7 bottomfish. An alternative definition of targeted Deep 7 bottomfish fishing was used for the 2018, 2021, and 2024 stock assessments to avoid using a weight-based criterion that could remove fishing events targeting bottomfish that caught low percentages based on total weight.

Filtering of non-bottomfish, single-reporting days (and, therefore, defining bottomfish single-reporting days) was mostly done for records that occurred prior to October 2002, when the fisher reporting form was less detailed. After October 2002, fishers could report on the hours fished, start and end times, and were given the option to record catches of 0 pounds, whereas prior to October 2002, this was not possible. Our definition of a non-bottomfish single-reporting day for records prior to October 2002 was twofold.

First, single-reporting days not targeting Deep 7 bottomfish were defined as single-reporting days that caught zero pounds of Deep 7 bottomfish and caught any Pelagic Management Unit Species (PMUS; WPRFMC 2009) listed in the DAR species code list, caught uku, or caught unknown species (species code = 0). Workshop participants stated that it was possible to target Deep 7 bottomfish without catching any Deep 7, but that it would be unlikely that pelagic species would be caught if Deep 7 were truly targeted. Similarly, single-reporting days with catches of uku but without any catches of Deep 7 bottomfish were believed to be reflective of fishing specifically targeting uku.

Second, in waters around the southwestern shore of the Big Island (management grids 100–102, 108, 120–122, and 128 in Figure 2) during 1985 and before, single-reporting days with the weight of Deep 7 bottomfish of less than 50 pounds and with catches of PMUS were considered to not be targeting bottomfish. This definition was restricted to the southwestern shore of the Big Island due to the uniqueness of the fishery. The ocean bathymetry of this region drops off steeply very quickly, and fishers who catch pelagic species (in particular tuna) can easily also catch Deep 7 bottomfish and vice versa. Consequently, Deep 7 bottomfish can be caught when targeting pelagic species. In addition, the gears commonly used to target tuna were not given their own unique gear codes until 1981, before which these gears were recorded as deep-sea handline.

Hence, it was difficult to determine whether single-reporting days using deep-sea handline off this region of the Big Island were actually targeting Deep 7 bottomfish or pelagic species. 1985 was chosen instead of 1981—when pelagic gear codes were implemented—because an analysis showing that for waters around the southwestern shore of the Big Island, the percentage of bottomfish by weight within single-reporting days was more consistent and stable after 1985 (Figure 19 in Yau 2018). Additionally, 1985 was chosen based on the notion that it would take time for fishers to begin reporting the new gear codes consistently. Based on these two definitions of non-bottomfish single-reporting days, 85,402 single-reporting days were not considered to be targeting Deep 7 bottomfish, and all 162,168 records from these single-reporting days were removed from further analysis, leaving records of only single-reporting days that were assumed to have targeted bottomfish.

The Deep 7 bottomfish fishery closed four times during 1948–2022. These seasonal closures began on April 16, 2008, July 6, 2009, April 20, 2010, and March 12, 2011, and extended to the end of the fishing season (August 31) for each year. Directed bottomfish fishing was not allowed during this time; therefore, an additional 4,153 records were removed from the 1,885 fishing events that occurred when the Deep 7 bottomfish fishery was closed, leaving a total of 717,736 records remaining from fishing events considered to have targeted the main Hawaiian Islands Deep 7 bottomfish.

2.5.2.2. Accounting for multi-day trips

Following the procedure first implemented in the 2018 benchmark stock assessment, distance traveled was used as the primary determinant of whether single-reporting days occurred over multiple days, and a measure of how often an individual fisher reported was used as a secondary determinant. Although the data contained fields indicating the start and end dates of fishing events after October 2002, these fields were blank for about half of the records. Therefore, distance traveled was used as the primary determinant of number of days. The number of days for fishing events was generally similar between an approach using the start and end dates and the distance approach for this subset of data. Details of each of these choices are described below.

Distance traveled

Port landed and area fished should be reported for each record in the fisher reported data. Consequently, the distance traveled between the port and the center of the fishing area was used to determine whether a single-reporting day likely occurred over one or multiple days. The distance traveled between each port and area was determined based on an independent key table constructed for a separate and ongoing analysis of the fisher reported data (J. Ault and S. Smith, University of Miami, personal communication). To reduce the number of distances required to be calculated, all of which were done by hand, the key table provided distances from a common port rather than distances from all possible ports.

The common port was centrally located among a group of ports in a similar geographic area on each island. In addition to saving time, a common port allowed for the potential of landing a vessel and driving to neighboring ports to sell the catch. Distances were calculated based on expected travel paths from the common port to the center of the fishing area while accounting for land barriers. Some records did not have a valid port recorded, so a common port could not be assigned, while other records' port-area combinations were not calculated in the key table. Overall, only 6,390 of the 717,736 records could not be assigned a valid distance.

During initial analyses, it was noticed that a few single-reporting days reported multiple areas fished, and multiple ports landed. Fishing in multiple areas and landing in multiple ports on a single-reporting day is possible, but for some combinations of areas and ports that are distant from one another, this is highly improbable, and most likely represents records from multiple single-reporting days recorded together or represents a database error. To initially account for multi-day trips, information on the number and location of ports and areas recorded on a single-reporting day were first used to refine the records into separate single-reporting days where applicable.

The criteria for splitting records within a single-reporting day differed based on the number of areas and common ports visited. There were multiple areas and a maximum of two common ports visited within any one single-reporting day. For single-reporting days with one area and one common port reported (1–1 single-reporting days), no further refinement was used. For single-reporting days with one area and two common ports reported (1–2 single-reporting days), records were separated into multiple single-reporting days only when the common ports were on two separate and nearby islands. If the common ports were on the same island, the single-reporting days were considered to accurately be a single-reporting day and were left unchanged. Similarly, if the common ports were at least two islands away (either Big Island-O‘ahu, or Kauai-Maui Nui), it was assumed to be a database error, and, therefore, the single-reporting day was left unchanged.

Single-reporting days with multiple areas and a single common port (2^{+}–1 single-reporting days) were also considered accurate, and the distance for the single-reporting day was assigned as the greatest distance among each port-area distance. For single-reporting days with multiple areas and two common ports (2^{+}–2 single-reporting days), nearly all single-reporting days had areas uniquely associated with one common port or the other common port. Therefore, the common port was used as a unique identifier to further separate among single-reporting days, and the distance assigned followed the same approach as for 2^{+}–1 single-reporting days. Accounting for multiple areas and common ports recorded for a single-reporting day added 101 new single-reporting days to the data set.

Once distance was assigned to each single-reporting day based on the common ports and fishing areas visited, an expected number of days was assigned to each single-reporting day based on a selected distance cut-off value. The cut-off value was selected based on the frequency of distances in 10-year time blocks (Figure 21 in Yau 2018) and from conversations with participants at the data workshops. Based on these discussions, it was expected that the cutoff in the earlier part of the time series would be smaller than that in the later part of the time series due to the vessels participating in the fishery early on being larger and slower.

However, a cutoff of 30 nautical miles (nm) was applied to all years as it indicated a clear break in the number of single-reporting days occurring in years after 1960. The 30 nm cutoff was also inclusive of possible single-reporting days in the 15–30 nm range for years prior to 1960 where a clear breakpoint was slightly less obvious. Each single-reporting day was assumed to last a day for every multiple of its cutoff. Thus, a distance between 0 and 30 nm would reflect a single-reporting with one day of effort, 30.1–60 nm a single-reporting day with two days of effort, and so on. Based on this criterion, a total of 31,740 single-reporting days were adjusted to have more than one day of effort, with the longest timeframe being 11 days. Single-reporting days without distances were assumed to have one day of effort.

Timing of reporting

The timing of reporting was also explored as a way to account for single-reporting days occurring over multiple days. Prior to October 2002, the number of records that were reported on the first and last day of a month was higher than the number of records reported on other days (Figure 20 in Yau 2018), which suggested that some of the single-reporting days were likely reported together as monthly reports rather than as daily reports.

Since there was no way to determine the number of days fished that made up a monthly report, all 4,498 records from the 834 license-year combinations that only reported on the first or last day of a month in a year were removed from analyses. This analysis was done using all years from 1949–2023, noting that the prevalence of license-year combinations meeting this criterion was reduced after 2002. Even though removing records from fishers that only ever reported on the first or last day of a month in a year may remove a valid single-reporting day that occurred during this time, such instances were expected to be inconsequential for CPUE calculation.

2.5.2.3. Selecting records that accurately represent trends in the fishery

The third step in preparing fisher reported data for CPUE analysis was to filter out records considered to be unrepresentative of the Deep 7 bottomfish fishery. Many options on how to do this were described in detail in the workshop report (Yau 2018). Ultimately, workshop participants agreed on two simple criteria: to filter out records from fishers who never reported catching a Deep 7 bottomfish, and to filter out records from fishers on days where they were participating in the fishery-independent bottomfish survey activities. The former criteria removed 8,440 records from 1,490 fishers who never reported a Deep 7 bottomfish, and the latter removed 256 records from fishers on days that they participated in the fishery-independent bottomfish survey.

The logic for removing records from individuals who never reported Deep 7 bottomfish was that such individuals were unlikely targeting bottomfish even though they were fishing deep-sea handline gear. The logic for removing records from the fishery-independent survey was that the fishing method would be functionally different for the survey than for the fishery and, therefore, unrepresentative. In what fishers described as rare, some fishers participated in the survey but also fished on their own that same day. Records that were part of the survey were indistinguishable from other records fished on the same day; therefore, all records from days where the fisher participated in the survey were removed.

2.5.2.4. Preparing data to incorporate variables affecting CPUE

The final step in preparing the fisher reported data for CPUE analysis was to incorporate additional variables needed for the standardization and to prepare the data for CPUE analysis. Details on each step are described below.

Additional variables affecting CPUE

Participants at the data workshops identified several variables they considered to influence bottomfish catch rates. Given logistical constraints, the top three were added to the data set for use in the CPUE standardization process for the 2018 and 2021 assessments. They were (1) a measure of fisher experience, (2) the pounds of uku caught, and (3) wind speed and direction. However, wind was not included for this assessment because wind data were not available before 1987 and had a minor influence on the resulting CPUE trend. A single series of CPUE could be estimated by omitting wind as a factor and using days fished as the unit of effort. The entire list of variables used in the standardization is discussed in section 2.5.3.1 Model selection. Other variables that workshop participants viewed as important (Table 8 in Yau, A. 2018) were not incorporated into CPUE standardization due to practical constraints but may be explored in future assessments.

Similar to the 2018 and 2021 assessments, fisher experience was calculated for each fishing event as the cumulative number of fishing events taken previously. This benchmark assessment was able to include fisher experience because individual fishers could be tracked from 1948 to 2023. Participants discussed and acknowledged that such a measure could not account for fishing conducted as crew members or experience gained through generational knowledge passed down by elders. Nonetheless, the cumulative number of fishing events remained a way to account for differences between experienced and inexperienced fishers.

Cumulative experience had a substantial influence on the standardized CPUE during initial analyses. Accounting for cumulative experience resulted in large changes in predicted CPUE during the first three decades of the analysis, corresponding to the period with incomplete fisher license information (Table 8). Additionally, fishers in the first several years would be determined to have less experience simply due to the time needed for data accrual. This raised concerns that the variable was a flawed indicator of actual experience. An alternative approach that selected the core fleet was used to account for experience. On average, the top 30% of fishers each year caught about 90% of the Deep 7 catch for years with complete fisher license information (Figure 6). Therefore, the core fleet was defined as fishers in the top 30% of annual Deep 7 catch in any year (see below).

Pounds of uku caught in each fishing event was included as a variable due to gear competition with Deep 7 bottomfish and the potential for fishers targeting Deep 7 bottomfish to switch to uku (and, therefore, away from Deep 7 bottomfish) when uku were present. Uku can be found in large numbers and although are not always targeted, it can be valuable when encountered, and, therefore, would alter the catch rates of Deep 7 bottomfish.

Preparing data for standardization

The filtered record-based data set for use in the CPUE standardization included 704,469 records from the 234,781 fishing events considered most representative of the Deep 7 bottomfish fishery. This data set was a record-based data set, which included fishing events containing records reporting different areas or separate hours. For CPUE standardization, only a single value of each dependent and independent variable can be included in the analysis. Consequently, the record-based data set was summarized into an event-based data set so that each data point used in the analysis contained information on a single unique fishing event.

The number of hours fished was not used as the unit of effort. However, reported hours fished were used to further define the event-based data set following procedures used in the 2018 benchmark assessment. Starting in October 2002, fishing events with multiple values of hours reported were divided into multiple fishing events so that each had only a single corresponding value of effort in hours. This was only possible for fishing events that occurred since October 2002 since effort for each record could be reported. A total of 938 fishing events that occurred since October 2002 had multiple hour values reported. Within each reported area for fishing events with multiple values of effort, records that had the same reported hours were treated as a single fishing event with catch equal to the sum of the reported weight of Deep 7 bottomfish.

Records within each area that had different hours reported were treated as separate fishing events. This approach assumed that when the same effort was reported across many records, the value represented total hours fished on a fishing event, whereas when multiple hour values were reported, the values represented individual fishing events that occurred in either multiple parts of the same fishing area or in separate areas altogether. Although this assumption combines separate fishing events with the same reported effort, such cases could not be known for certain. This approach at least accounted for unique fishing events with different effort values. Using this approach added 1,011 fishing events to the data set.

Multiple areas within a single fishing event were also reported for 4,941 fishing events; however, area-specific effort information for separating these into unique fishing events was not available. Consequently, the area with the greatest amount of Deep 7 bottomfish by weight was assigned to the fishing event for use in the standardization. In cases where the weight of Deep 7 bottomfish was the same across multiple areas, the smaller numbered management grid was selected, reflecting a general preference towards management grids nearer to land (Figure 2). This choice was somewhat arbitrary, but given the few occurrences (167 fishing events), the effect on the standardization was expected to be negligible.

Once the data set was summarized into individual fishing events, CPUE for each fishing event was calculated as the total weight in pounds of Deep 7 bottomfish caught across all records within a fishing event, divided by the unit of effort. After accounting for multiple values of independent variables so that each fishing event had only one value for any independent variable, the final filtered event-based data set for use in the CPUE standardization consisted of 235,701 data points.

The CPUE standardization included HDAR reporting area and year *x* area interactions (see below). A cutoff of at least 15 years of data per area was applied to avoid areas with sparse data—which may have also been misreported—and to increase the ability to estimate area effects. Applying the 15 year cutoff removed areas that were unlikely to represent suitable Deep 7 habitat, and retained areas within the domain of the BFISH survey and over Middle Bank (Figure 2, Figure 7). This criterion reduced the event-based data set to 234,999 records.

Selecting records for the core fleet required removing records without information on fisher. All records from 1976 were removed from the data set because fisher information was unavailable (Figure 6). The data set contained 142,073 records after selecting the core fleet, defined as fishers with annual catch in the top 30% of fishers in any year.

2.5.3. CPUE standardization

2.5.3.1. Model selection

Deep 7 bottomfish CPUE was standardized using generalized linear and generalized linear mixed models (McCulloch et al. 2008). It was acknowledged during the data workshops that catching zero pounds of Deep 7 bottomfish was possible when targeting Deep 7 bottomfish. Consequently, zero catches of Deep 7 bottomfish were included in CPUE standardization for the base case scenario for this assessment. For the core fleet data set, seven percent of the total data points had zero catches of Deep 7 bottomfish.

There are numerous ways to deal with zero catches when standardizing CPUE (Maunder and Punt 2004). A delta-lognormal approach was used in this assessment wherein CPUE was modeled as the product of two processes: a Bernoulli process modeling the probability of positive catches, and a positive process modeling the distribution of CPUE given a positive catch, which was assumed lognormal. The response variable for the Bernoulli process was a binomial variable that was added to the data set, indicating whether a Deep 7 bottomfish was captured (1 = captured, 0 = not captured). The relationship between the response variable and the predictor variables was modeled as a binomial distribution using a logit link function, and fishing effort was included as an offset. The response variable for the positive process, henceforth referred to as the lognormal process, was the natural logarithm of CPUE from positive catches of Deep 7 bottomfish. A Tweedie model and hurdle gamma model were also considered in place of the delta-lognormal as alternative ways to include zero catches, but they were not used due to difficulties matching the dispersion and zero inflation of the data.

Model selection techniques were used for each of the Bernoulli and lognormal processes to select from the suite of possible predictors those predictors that most improved model fit. Predictor variables for model selection included a mix of categorical and continuous variables, as well as fixed and random effects. Each variable was considered to have some effect on bottomfish CPUE that varied on an annual basis because of changes in the distribution of fish or the spatial pattern and effectiveness of fishing effort.

Categorical variables included fishing year, management area, island region, quarter, and individual fisher as first-order variables, and area-fishing year and area-quarter as second-order interactions. Island region was defined as Big Island for management areas 100–299, Maui-Nui for management areas 300–399, Oahu for management areas 400–499, and Kauai-Niihau for management areas 500 and above. Quarters were separated along the definition of fishing year with July to September as quarter one, October to December as quarter two, January to March as quarter three, and April to June as quarter four. The pounds of uku caught was the only continuous variable. All variables were modeled as fixed effects except for fisher, and an area x year interaction, which were modeled as random effects.

Selection among CPUE standardization models was performed using Akaike’s information criterion (AIC = 2 × number of parameters – 2 × ln(likelihood evaluated at its maximum)) to judge the relative goodness of fit (Burnham and Anderson 2002). Model selection was done using a forward-selection process with a threshold of 0.1% of the previous model’s AIC. Thus, if the improvement in AIC of a model after adding a new predictor was greater than 0.1% of the previous model’s AIC, the added predictor was considered significant, and kept for the best-fitting model.

A percentage-based threshold was used as opposed to a constant value due to large likelihood values caused by the high number of data points, following the suggestion by Maunder and Punt (2004). The significance of the random effect of fisher was tested first, and model selection using fixed effect terms was done thereafter. The random area *x* year interaction was added to the model following the selection of main effects.

Fishing year was required for the index, so year was retained first among fixed effect terms in model selection regardless of AIC score. Model selection was done using maximum likelihood for all models. Estimation was done for generalized linear mixed models using restricted maximum likelihood once the best-fit model was determined.

Restricted maximum likelihood accounts for degrees of freedom used in estimating fixed effects and estimates variance components of the random effects without influence from fixed-effect terms (Harville 1977; McCulloch et al. 2008). Statistical modeling was done within the R software package version 4.2.2 (R Core Team 2022) using the glmmTMB package (Brooks et al. 2017) for the Bernoulli process and the lme4 package version 3.2 (Bates et al. 2015) for the lognormal process.

The best fit model for the Bernoulli process included fisher, fishing year, area, an interaction term for area and year, pounds of uku, and quarter (Table 9) and reduced deviance by 29% relative to the null model (i.e., intercept only) and 8.2% from the model with only year and fisher. The best fit model for the lognormal process included fisher, fishing year, area, an interaction for area and year, quarter, and pounds of uku and reduced deviance by 14.5% relative to the null model and 2.7% from the model with only year and fisher. Including fisher in the model reduced total model deviance the most among predictors.

2.5.3.2. Model diagnostics

Regression diagnostics were used to qualitatively check model assumptions. Model fit was assessed through visual comparison of residuals plotted against predicted values of the response variable and against values of the predictor variables. Pearson residuals were used for all models for the lognormal process. Quantile residuals were used for all models for the Bernoulli process as recommended by Dunn and Smyth (1996). Plots of the quantiles of the standardized residuals to the quantiles of a standard normal distribution were also used to assess assumptions of normality for models for the lognormal process.

Diagnostic residual plots and summary output of best-fit models show some deviation from assumptions about heteroscedasticity in models for the Bernoulli process, but in general, models were appropriate. For the early time period, the histogram of quantile residuals indicated that distributional assumptions were not violated, and the plot of quantile residuals to the response variable showed some presence of heteroscedasticity (Figure 8). The smaller range in residuals at lower values of the response variable was attributed to fewer data points at these low probabilities. Plots of residuals against predictor variables indicated no patterning with individual variables. For the later time period, the histogram of quantile residuals did not indicate a violation of normality. With the exception of some patterning in pounds of uku, plots of quantile residuals against predictor variables showed no patterning (Figure 8). Altogether, the diagnostic plots were not considered indicative of serious violations in model assumptions for the Bernoulli process.

Initial diagnostics of models for the lognormal process indicated skewed residuals for the predictor pounds of uku, which was the reason the square-root transformation on this parameter was used for both processes. The square root transformation was used for uku because there were instances with zero uku pounds. Residual plots with the transformed variables improved the patterning of the Pearson residuals for both overall predictions and parameter-specific residuals.

There remained some skewness towards smaller response values, as evident by the quantile-quantile plot (Figure 9). A gamma distribution with log link was explored to determine if it would improve the residual patterns and add greater probability to the lower tails; however, the Gamma model was unable to converge with random effects. Comparison between a fixed-effect only model under the gamma distribution with an identical model under the lognormal distribution showed no improvement in residual patterns. Therefore, the lognormal distribution was kept for the best-fit model.

2.5.3.3. Index calculation

The best-fit models were used to develop the index following the “predict-then-aggregate” approach (Hoyle et al. 2024). For both the Bernoulli and lognormal models, predictions were generated for a factorial grid that included the selected categorical variables year, quarter, and area. Mean values were used for the continuous variables square root of pounds of uku caught and days fished (for the Bernoulli effort offset). Predicted values of the response variable for the factorial grid from each model were calculated using the predict function in R.

For the lognormal process, predicted values were generated using the glmmTMB package with a Guassian family because the package properly generates predicted standard errors for mixed effects models, whereas we found the lme4 package to generate predicted standard errors (SEs) that decreased as the number of random effects in the model increased. The population mean was used to interpolate missing area *x* year combinations given that this interaction was modeled as a random effect (Campbell 2015). The predicted values from the positive process were multiplied by the exponential of one-half the residual variance to correct for bias when back-transforming from ln(CPUE) to CPUE. The index, *IT*, was then calculated as the product of the mean probability of catching a Deep 7 bottomfish in year *T* and the mean CPUE in year *T.* The index was weighted by area *k*, using the proportion of the total Deep 7 essential fish habitat, A_k (defined as area with depths from 75 to 500 m). Thus, the index of abundance in year *T*, and quarter *j* was calculated as:

Equation 1: $I(T,j) = \sum_{k=1}^{Na} A_k \cdot I_{T,j,k},$

where N_a gives the number of areas. The arithmetic average was then used to combine among quarters within a year.

The variance for each process was calculated as the square of the predicted standard error using the predict function in R. The variance of the index in year *T,* quarter *j,* and area *k* was calculated as the variance of the product of two independent random variables, the Bernoulli (Δ) and lognormal process (φ; Goodman 1960):

Equation 2:

$$Var(I_{T,j,k}) = Var(\Delta_{T,j,k})Var(\varphi_{T,j,k}) + Var(\Delta_{T,j,k})E(\varphi_{T,j,k})^2 + Var(\varphi_{T,j,k})E(\Delta_{T,j,k})^2.$$

The area-weighted variance in year *T* and quarter *j* was then calculated (Campbell 2015):

Equation 3: $Var(I_{T,j}) = \sum_{k=1}^{N_a} A_k^2 Var(I_{T,j,k})$.

The annual variance, Var(I_T) was then calculated as the sum of the quarter-specific variances and the standard error was calculated as the square root of Var(I_T). The standard error was used to calculate the annual CV for the survey.

The assessment model requires the user to input variability around the CPUE index as the standard error of the mean index on the scale of the logarithm. Consequently, we calculated standard errors on the scale of the logarithm in each year from the CV in each year using the relationship:

Equation 4: $CV_t = \sqrt{e^{\sigma_t^2} - 1}$.

The yearly indices and standard errors on the scale of the logarithm were used as input into the assessment models and are provided in Table 10.

2.6. Fishery-independent survey

PIFSC has developed a Bottomfish Fishery-Independent Survey in Hawai‘i to provide independent estimates of Deep 7 biomass and worked with cooperative research fishers to conduct the survey (Richards et al. 2016). The survey consisted of two gears, research fishing (PIFSC 2022a) and underwater stereo video cameras (PIFSC 2022b). Research fishing utilized fishing gears and techniques similar to those used in the Deep 7 fishery. Fishing effort was identical among all fishing events, and the locations for fishing were pre-determined based on a stratified random sampling design.

Underwater stereo video cameras were used to complement research fishing, and on occasion were used to focus sampling in sensitive areas and provide estimates on fish biomass that may be present in the water but not caught during research fishing.

Surveys covered the entirety of the main Hawaiian Islands. The first operational survey was conducted in calendar year 2016, and this assessment used data from surveys conducted from 2016 through 2022. The fishery-independent survey was conducted in early fall of calendar years 2016–2022, corresponding approximately to the beginning of fishing years 2017–2023.

The 2018 benchmark and 2021 update stock assessments were fit to estimates of absolute biomass estimates. See Richards et al. (2016) for complete details on the fishery-independent survey and Ault et al. (2018a) for the methods used to calculate the overall absolute biomass estimate. Estimating biomass requires a gear calibration factor for converting research fishing observations to camera observations and an assumed effective radius of a single sample from the camera gear.

The 2018 and 2021 assessments used a prior distribution for the radius, the estimate of which ultimately determined the catchability of the survey. The prior mean for the radius was 27.6 m, based on the best estimate from Ault et al. (2018a) and a CV of 50%. The minimum and maximum values, respectively, for the effective radius of a single sample were assumed to be 7.5 m and 60.6 m (Ault et al. 2018a and b). The results from the biomass dynamics model are sensitive to the CV of the camera radius prior, which determines how closely to fit the absolute survey estimate (Syslo et al. 2021).

The 2024 benchmark stock assessment incorporated the BFISH survey as a relative index of abundance to avoid the requirement for a swept-area assumption for the camera gear. A spatiotemporal delta-general linear mixed model (GLMM) was developed using the Vector Autoregressive Spatio-Temporal (VAST) package in R (Thorson 2019), which provides a robust spatiotemporal modeling framework that can accommodate multiple gears (e.g., camera and research fishing) and evaluate covariates that influence abundance or catchability (Ducharme-Barth 2023). A multivariate delta-lognormal model was used to develop an aggregate index for the entire Deep 7 complex.

The overall trend in Deep 7 relative abundance from the model-based approach was relatively similar to the design-based (i.e., absolute) estimates, however, the 2024 benchmark fit the index in a substantially different manner than the 2018 and 2021 assessments (see below). The relative abundance and CV for the BFISH index are given in Table 11. See Ducharme-Barth (2023) for a detailed description of the model-based index development, including sensitivity scenarios and comparisons to the design-based estimates from Richards, B. L. (2023).

2.7. Assessment Model

In this section, the production model assumptions and structure that were used to estimate biomass and fishing mortality for the Deep 7 stock assessment are described. The current assessment uses new modeling software compared to the previous benchmark and update assessments, yet still implements a Bayesian state-space surplus production modeling framework.

2.7.1. Biomass Dynamics Model

This stock assessment used Just Another Bayesian Biomass Assessment (JABBA), which is an open source modeling framework for conducting state-space Bayesian surplus production models (Winker et al. 2018). JABBA uses R (R Core Team 2022) to set up the model and call the software program Just Another Gibbs Sampler (JAGS, Plummer 2003) using the R package ‘rjags’ (Plummer et al. 2016). JABBA explicitly estimates both process error variance and observation error variance that have been commonly used for fitting production models with biomass indices (Meyer and Millar 1999; McAllister et al. 2000; Punt 2003; Brodziak and Ishimura 2011), and estimates Bayesian posterior distributions of model outputs using Markov chain Monte Carlo (MCMC) simulation (Gilks et al. 1995).

Surplus production models are frequently implemented to estimate sustainable levels of harvest (biomass removals) at corresponding levels of stock biomass. The exploitable biomass time series comprised the unobserved state variables, and was estimated by fitting model predictions to the observed biomass indices (i.e., CPUE) and catches using observation error likelihood functions and prior distributions for the model parameters. The observation error likelihood measured the discrepancy between observed and predicted CPUE, as well as between observed and predicted biomass indices, while the prior distributions represented the relative degree of belief about the probable values of model parameters. Assumptions of this model were that production followed a specified functional form, the assessments applied to exploitable individuals, all exploitable individuals were mature and equally vulnerable to fishing, and that biomass was proportional to CPUE.

The process dynamics represented the temporal fluctuations in exploitable bottomfish biomass due to density-dependent population processes (e.g., growth) and fishery catches. JABBA formulates the surplus production function as a generalized three-parameter equation following the formulation of Pella and Tomlinson (1969) and Fletcher (1978) (Gilbert 1992; Thorson et al. 2012). Under this three-parameter production function, exploitable biomass at the start of year t(B_t) depended only on the previous time period’s exploitable biomass (B_{t-1}) and total catch (C_{t-1}), and on the intrinsic growth rate (r), carrying capacity (K), and production shape (m) parameters:

Equation 5: $B_t = B_{t-1} + \frac{r}{m-1} B_{t-1}\left(1 - \left(\frac{B_{t-1}}{K}\right)^{m-1}\right) - C_{t-1}$.

The production shape parameter *m* determined where surplus production peaked as biomass varied as a fraction of carrying capacity. If the shape parameter m=2, the model reduces to the Schaefer form with the surplus production function attaining a maximum at a biomass value equal to K/2. If 0 <m< 2, MSY occurs when biomass values are smaller than K/2, and when m>2, MSY occurs when biomass values are greater than K/2. The Pella-Tomlinson formulation reduces to a Fox form if *m* approaches 1, resulting in MSY at approximately 1/e≈0.368K, but there is no exact solution for MSY when m=1.

For computational purposes, the production model in Equation 5 was expressed in terms of the proportion of carrying capacity (*P*) in year *t* (i.e., setting P_t=B_t/K) to improve the efficiency of the MCMC algorithm for estimating parameters (e.g., Meyer and Millar 1999). As such, the process dynamics for the temporal changes in the proportion of carrying capacity were

Equation 6: $P_t = P_{t-1} + \frac{r}{m-1} P_{t-1}(1 - (P_{t-1})^{m-1}) - \frac{C_{t-1}}{K}$.

The values of exploitable biomass and harvest rate that maximized biomass production were relevant as biological reference points for fishery management and for estimating the MSY of bottomfish stocks. Based on Equation 5, the exploitable biomass that was required to produce maximum sustainable yield (B_{MSY}) was

Equation 7: $B_{MSY} = Km^{\frac{-1}{m-1}}$.

It follows that the shape parameter *m* can be arithmetically translated into a ratio of B_{MSY} to *K* (Prager, 1994), such that

Equation 8: $\frac{B_{MSY}}{K} = m^{\left(-\frac{1}{m-1}\right)}$.

Within JABBA, the user specifies a value of B_{MSY}/K to assign the prior mean of *m*. The harvest rate that was required to produce maximum sustainable yield (H_{MSY}) was

Equation 9: $H_{MSY} = \frac{r}{m-1}\left(1 - \frac{1}{m}\right) = \frac{r}{m}$,

where the harvest rate *H* was defined as the ratio of catch over biomass. The estimate of *MSY* was

Equation 10: $MSY = H_{MSY} B_{MSY} = \frac{r}{m-1}\left(1 - \frac{1}{m}\right) Km^{\frac{-1}{m-1}}$.

Process error was added to the deterministic process dynamics (Equation 11). The process error model related the dynamics of exploitable biomass to natural variability in demographic and environmental processes affecting the Deep 7 stock. The deterministic process dynamics were subject to natural variation due to fluctuations in life history parameters, trophic interactions, environmental conditions, and other factors. In this case, the process error represented the joint effects of many random multiplicative events which combined to form a multiplicative lognormal process under the central limit theorem. As a result, the process error terms were set to be independent and lognormally distributed random variables.

The process error model defined the stochastic process dynamics by relating the unobserved biomass states to the observed catches and the estimated population dynamics parameters. Given the multiplicative lognormal process errors, the state equations for the initial year (*t*=1) and subsequent years (*t*>1) were

Equation 11:

$$P_t = \begin{cases} \varphi e^{\eta_t} & for\ t = 1 \\ \left(P_{t-1} + \frac{r}{(m-1)} P_{t-1}(1 - P_{t-1}^{m-1}) - \frac{C_{t-1}}{K}\right) e^{\eta_t} & for\ t > 1 \end{cases},$$

where η_t were identically distributed normal random variables with mean 0 and constant process error variance σ_η^2The coupled process dynamic equation set the prior distribution for the proportion of carrying capacity $p(P_t)$ in each year t>*1*, conditioned on the proportion of carrying capacity in the previous period. The proportion of carrying capacity in the initial year was assigned its own prior *p(ψ),* which is described in section 2.7.3 Prior Distributions.

2.7.2. Observation Error Model

Two observation error models were applied for this current stock assessment: one for the commercial CPUE and the other for the fishery-independent survey CPUE. The observation error models related the observed fishery CPUE to the exploitable biomass of the Deep 7 stock. We assumed that the standardized fishery CPUE index (I_t) in year *t* was proportional to biomass with catchability coefficient *q* as

Equation 12: $I_T = qB_t = qP_tK.$

Observation error was added to the deterministic index equation (Equation 12). The observed CPUE dynamics were subject to natural sampling variation which was assumed to be lognormally distributed. Given the lognormal observation errors, the observation equations for the CPUE index for each year *t* were

Equation 13: $I_T = qP_tKe^{\tau_t},$

where τ_t were identically distributed normal random variables with mean 0 and total observation error variance $\sigma^2_{\tau_t}$. This specifies the CPUE observation error likelihood function $p(I_t|\theta)$ given model parameters and unobservable states θ.

JABBA partitions the annual total observation error variance into three separate components following Francis et al. (2003). The three components were: 1) $\sigma^2_{\tau_{SE_t}}$, the external estimable observation error inputted as the standard error on the scale of the logarithm of standardized CPUE in year t; 2) $\sigma^2_{\tau_{fixed}}$, an optional user-provided component of observation error that is constant across all years, and 3) $\sigma^2_{\tau_{estimated}}$, an estimated observation error that is constant across all years. Consequently, total observation error variance for year t is:

Equation 14: $\sigma^2_{\tau_t} = \sigma^2_{\tau_{SE_t}} + \sigma^2_{\tau_{fixed}} + \sigma^2_{\tau_{estimated}}$.

Total observation error typically ranges from 0.1 to 0.4 (Francis et al. 2003). For this assessment, we did not use the optional fixed component of observation error ($\sigma^2_{\tau_{fixed}} = 0$) for either index, but allowed the model to estimate the total observation error variance for the commercial CPUE index. For the fishery-independent survey CPUE index, observation error only consisted of the standard error from the standardized CPUE index (i.e., $\sigma^2_{\tau_{SE_t}}$). Additional observation error for the commercial CPUE index was estimated because the CV from the standardized commercial CPUE was considerably smaller than from the fishery-independent survey; the annual CV from the standardized CPUE averaged 0.048 compared to 0.15 for the fishery-independent survey.

The amount of total observation error is inversely related to the amount of influence (i.e., weight) an index will have on model results, and the fishery-independent index was hypothesized to be at least as representative of actual trends in biomass as the commercial data. Therefore, it was decided to avoid fitting a model where the commercial CPUE series had three times the weight of the fishery-independent survey.

2.7.3. Prior Distributions

A Bayesian estimation approach was used to estimate production model parameters. Prior distributions were used to represent existing knowledge and beliefs about the likely values of model parameters. The intrinsic growth rate parameter r, the carrying capacity parameter K, the ratio of initial biomass to carrying capacity parameter ψ, the catchability parameter q, the production function shape parameter m, and the process error σ^2_{η} and the estimable component of observation error $\sigma^2_{\tau_{estimated}}$ variance parameters had prior distributions.

Unobserved biomass states expressed as the proportion of carrying capacity were included in the joint prior distribution and were conditioned on the parameter estimates and the previous biomass as a proportion of carrying capacity and catch. A summary of prior distributions is provided below and in Table 12. The effect of the choice of prior assumptions on model results was assessed through sensitivity analyses as described in section 2.9 Sensitivity Analyses.

Prior for Intrinsic Growth Rate

The prior distribution for intrinsic growth rate *p*(*r*) was a moderately informative lognormal distribution with mean (μ_r) and variance (σ_r^2) parameters:

Equation 15: $p(r) = \frac{1}{r\sigma_r\sqrt{2\pi}} exp\left(-\frac{(ln\, r-\mu_r)^2}{2\sigma_r^2}\right)$.

As input for a lognormal prior on *r*, JABBA requires the user to enter the mean on the original scale, and the standard deviation on the scale of the natural logarithm. The prior mean and variance values were calculated from the ASEM approach detailed in section ‘2.3 Biological Data.’ The value of the prior mean of the intrinsic growth rate parameter was set to μ_r = 0.096 and the prior variance σ_r was set such that the CV was 0.66. The CV estimated from the ASEM approach was 0.33, but this was doubled to account for sources of variability that were not included in the approach (i.e., growth, maturity, maximum age, and selectivity).

Prior for Carrying Capacity

The prior distribution for carrying capacity *p(K)* was an informative lognormal distribution with mean (μ_K) and variance (σ_K^2) parameters:

Equation 16: $p(K) = \frac{1}{K\sigma_K\sqrt{2\pi}} exp\left(-\frac{(ln\, K-\mu_K)^2}{2\sigma_K^2}\right)$.

As input for a lognormal prior on *K*, JABBA requires the user to enter the mean and coefficient of variation on the original scale. The value of the prior for carrying capacity for the base case model used the default setting in JABBA, where μ_K was calculated as 8 times the maximum catch (9.14 million lb) with a CV of 1.0.

Prior for Ratio of Initial Biomass to Carrying Capacity

The prior distribution for the ratio of initial biomass to carrying capacity *p(ψ)* was a moderately informative lognormal distribution with mean (μ_ψ) and variance (σ_ψ^2) parameters:

Equation 17: $p(\psi) = \frac{1}{\psi\sigma_\psi\sqrt{2\pi}} exp\left(-\frac{\left(ln\, \psi-\mu_\psi\right)^2}{2\sigma_\psi^2}\right)$.

As input for a lognormal prior on ψ, JABBA requires the user to enter the mean and coefficient of variation on the original scale. The value of the prior mean of the ratio of initial biomass to carrying capacity was μ_ψ 0.75, with a CV of 0.5. This value reflected the understanding that biomass was unlikely to be substantially depleted in 1949 because fishing was minimal during World War II and the years immediately following.

Prior for Production Shape Parameter

The prior distribution for the production function shape parameter *p(m)* for the Pella-Tomlinson formulation was a moderately informative lognormal distribution with mean (μ_m) and variance (σ_m^2) parameters:

Equation 18: $p(m) = \frac{1}{m\sigma_m\sqrt{2\pi}} exp\left(-\frac{(ln\, m-\mu_m)^2}{2\sigma_m^2}\right).$

JABBA parameterizes the *m* prior based on user input for B_{MSY}/K, where *m* is determined from B_{MSY}/K according to Equation 8. As input for the lognormal prior on *m*, JABBA requires the user to enter the mean of B_{MSY}/K on the original scale, and the standard deviation of *m* on the scale of the natural logarithm. The mean value of B_{MSY}/K (B_{MSY}/B_0 from the ASEM approach described in Section 2.3 Biological Data) was 0.33 with a CV=0.1. The CV was doubled to 0.2 to account for sources of variability that were not included in the ASEM approach.

Prior for Catchability

The prior distribution for fishery catchability *p(q)* was chosen to be an uninformative uniform distribution on the interval [10^{-10}, 10]. This diffuse prior was chosen to allow the data and model structure to completely determine the distribution of fishery catchability estimates.

Prior for Variability Around Catch

An informative prior was used to incorporate annual variability in total catch in year *t* into model estimates. The prior distribution for catch uncertainty $p(C_t)$ was an informative lognormal distribution with mean (μ_{C_t}) and variance ($\sigma_{C_t}^2$) parameters:

Equation 19: $\boldsymbol{p(C_t) = \frac{1}{C_t\sigma_{C_t}\sqrt{2\pi}} exp\left(-\frac{(ln\, C_t-\mu_{C_t})^2}{2\sigma_{C_t}^2}\right).}$

The mean parameter μ_{C_t} was set to the value of catch in each year and a CV of 0.13 for each year, which was based on the CV calculated for the HMRFS catch estimates (2.4.2 Estimates of Unreported Bottomfish Catch). The average CV from HMRFS was about 0.26, which was reduced by 50% given that the ratio of catch from HMRFS to reported commercial catch was about 1:1. The catch variation prior was chosen to propagate uncertainty inherent in the noncommercial catch into the estimation of sustainable harvest rates and biomasses and assumes complete reporting of commercial catches.

Priors for Error Variances

The prior distribution for the process error $p(\sigma_\eta^2)$ and the estimated component of the observation error $p(\sigma^2_{t_{estimated}})$ followed inverse-gamma distributions with rate parameter λ and shape parameter *k:*

Equation 20: $$\boldsymbol{p(\sigma_x^2) = \frac{\lambda^k (\sigma^2)^{-k-1} exp\left(\frac{-\lambda}{\sigma^2}\right)}{\Gamma(k)}},$$

where *x* represents either η or $t_{esimated}$. The inverse-gamma distribution is a useful choice for priors that describe model variances (Congdon 2001). A less informed distribution was used for the process error variance, where the rate (λ) and shape (*k*) parameters were both set to 0.001 (Winker et al. 2018). For the estimated observation error variance, a moderately informative inverse gamma distribution was used with the rate parameter set to λ=0.1 and the shape parameter *k*=0.2 (Langseth et al. 2018). The choice of the observation error prior matched the expected scaling of observation errors for the observation equation describing the model fit to observed CPUE (Equation 14), which was based on CPUE and corresponding standard error values on the scale of the logarithm, which were on the order of 0 to 1.

Posterior distribution

Independent samples from the joint posterior distribution of the surplus production model were numerically simulated to estimate model parameters and make inferences. The joint posterior distribution of model parameters and unobservable states θ given the data D, *p*(θ|D), was proportional to the product of the priors of the parameters and unobservable states, and the joint likelihood of the CPUE data across all *n* years:

Equation 21:

$$p(\theta|D) = p(r)p(m)p(K)p(\varphi)p(q_i)p(\sigma_\eta^2)\prod_{t=1}^{n} p(\sigma^2_{\tau_{t,i}})\prod_{t=1}^{n} p(C_t)$$
$$\times\ p(P_1|\psi, K, \sigma_\eta^2)\prod_{t=2}^{n} p(P_t|P_{t-1}, r, m, K, \psi, \sigma_\eta^2)\prod_{t=1}^{n} p(I_{i,t}|P_t, q_i, K, \sigma^2_{\tau_{t,i}})$$

Parameter estimation for multi-parameter and nonlinear Bayesian models, such as those used in this assessment, is typically based on simulating a large number of independent samples from the posterior distribution (Gelman et al. 1995). JABBA uses MCMC simulation (Gilks et al. 1995) to numerically generate samples from the posterior distribution.

Initial starting conditions of the MCMC chains were randomly drawn from their respective prior distributions. Three chains of 150,000 samples each were then simulated from the posterior distribution. The first 50,000 samples of each simulated chain were excluded from the estimation process to remove dependence of the MCMC chain on the initial conditions and to ensure stationarity of the remaining samples in the chain. Each chain was then thinned by 10 to reduce autocorrelation, for example, every tenth sample from the posterior distribution was stored and used for inference. As a result, a total of 30,000 samples were available to summarize model results.

Prior distributions and estimated posterior distributions were compared to show whether the catch and standardized CPUE data were informative for estimating model parameters. This comparison included the priors and posteriors for the following model parameters: intrinsic growth rate, carrying capacity, ratio of initial biomass to carrying capacity, catchability, estimable observation error variance, and process error variance. Posteriors of the derived quantities MSY, B_{MSY}, and H_{MSY} were also compared to their respective derived prior distributions.

2.7.4. Convergence and Model Diagnostics

Convergence of the simulated MCMC samples to the posterior distribution was assessed via visual inspection of the trace and autocorrelation plots. Convergence was also confirmed using the Geweke convergence diagnostic (Geweke 1992), and the Heidelberger and Welch stationarity and half-width diagnostics (Heidelberger and Welch 1983). The set of convergence diagnostics were applied to key model parameters (i.e., intrinsic growth rate, production function shape, carrying capacity, ratio of initial biomass to carrying capacity, catchability coefficients, and error variances) to verify convergence of the MCMC chains to the posterior distribution.

Residuals from the base case model fit to CPUE were used to measure the goodness of fit of the production model. These log-scale observation errors $\varepsilon_{i,T}$ of observed minus predicted CPUE were

Equation 22: $\varepsilon_{i,T} = ln\left(I_{i,T}\right) - ln(q_i K P_T)$.

Nonrandom patterns in the CPUE residuals would suggest that the observed CPUE may not have conformed to one or more model assumptions. We tested for normality of the log-scale residuals using the Shapiro-Wilk test, patterns in the sign of the residuals using a runs test, and trend in the residuals by assessing if the slope of a regression of the residuals over time was significantly different than zero. All residual tests were done using a p-value of 0.05.

A retrospective analysis was conducted to assess whether there were consistent patterns in model-estimated outputs based on decreasing periods of data (Mohn 1999). The retrospective analysis was conducted by successively removing the catch and CPUE data for years 2023 to 2019 in one-year increments such that the terminal years of the model ranged from 2022 to 2018, re-estimating model parameters, and comparing the resulting biomass and harvest rate time series with the model with all data included. The magnitude of the retrospective pattern was assessed using Mohn's rho (ρ; Mohn 1999), which computes relative patterns of deviations with respect to a base case model:

Equation 23: $\rho = \sum_{y=2018}^{2022} \frac{X_{(y1:y),y} - X_{(y1:y2),y}}{X_{(y1:y2),y}}$,

where $y1$ = the start year of data and $y2$ = 2023, spanning the full data set of the base case model; X indicates either exploitable biomass or harvest rate, and y indicates the terminal year for each retrospective refitting (i.e., y from 2018 to 2022).

2.8. Projections

Estimated posterior distributions of base case assessment model parameters were used in forward projections for 2025-2029 to estimate the probability of overfishing, P^*, from 2025-2029 under alternative future catches. The projection results accounted for uncertainty in the distribution of estimates of model parameters from the posterior of the base case model.

The projections were conducted assuming each value for the future catch was constant through all projection years. The projected total catch scenarios ranged from 0 to 2 million pounds in 1000-pound increments. The projections accounted for a ratio of unreported:reported catch of 1.09. When results for projections are displayed, total catches are converted to reported catches for management purposes. To advance the model to the starting year of projections, total catch for fishing year 2024 was set equal to the average catch value from 2022-23 because these years were considered to represent a period where fishing had normalized following the greatest effects of the COVID-19 pandemic. Projections were used to compute reported catches for 2025-2029 that would produce probabilities of overfishing varying from 0% to 50% at 1% intervals. The future catch corresponding to a 50% risk of overfishing can be considered the overfishing limit (OFL). Other quantities of interest including corresponding relative biomass (B/B_{MSY}), stock status, and risks of overfishing and overfished status were also calculated.

2.9. Sensitivity Analyses

A suite of sensitivity analyses was conducted to evaluate how the base case model results would be affected if different assumptions were made regarding model structure, prior distributions, data sources, and initial conditions. The initial conditions of the MCMC sampler for the sensitivities were kept the same as for the base case, except where otherwise noted. Scenarios for sensitivity analyses are described below and in Table 13.

Sensitivity to alternative prior distribution for intrinsic growth rate

The sensitivity of base case model results to the prior distribution for intrinsic growth rate was evaluated by fitting the model using four different prior distributions for r, which were calculated as ±25% and ±50% of the mean.

Sensitivity to alternative prior distribution for carrying capacity

The sensitivity of base case model results to the prior distribution for carrying capacity was evaluated by fitting the model using four different prior distributions for K, which were calculated as ±25% and ±50% of the mean.

Sensitivity to alternative prior distribution for ratio of initial biomass to carrying capacity

The sensitivity of base case model results to the prior distribution for the ratio of initial biomass to carrying capacity was evaluated by fitting the model using four different prior distributions for ψ, which were calculated as ± 25% and ± 50% of the mean.

Sensitivity to alternative prior distribution for production shape

The sensitivity of the base case model results to the prior distribution for the production shape parameter (specified as B_{MSY}/K in JABBA) was evaluated by fitting the model using four different prior distributions for B_{MSY}/K which were calculated as ±25% and ±50% of the mean.

Sensitivity to using prior distributions based on ʻōpakapaka life history

Given the predominance of ʻōpakapaka in the Deep 7 catch, the sensitivity to using priors based on ʻōpakapaka life-history was evaluated by using an r prior with a mean equal to 0.096 with a CV of 0.14 and the mean B_{MSY}/K was equal to 0.315 with a CV of 0.25. Catch data, CPUE data, and other priors were the same as used in the base case model.

Sensitivity to excluding variation around catch

The sensitivity of excluding variability around the catch estimates was assessed using a single sensitivity. For this sensitivity, the variation around catch was removed from the model. In effect, this sensitivity analyses addressed whether including additional uncertainty in the catch time series had a strong influence on model results.

Sensitivity to including HMRFS estimates of noncommercial catch from 2004–2023

The sensitivity of using annual estimates of unsold catch from HMRFS rather than the average ratio of HMRFS unsold catch to commercial catch was evaluated. The unreported catch for 2004-2023 was calculated as the sum of annual species-specific unsold catches and annual CVs were used for the catch variation (Table 6). HMRFS estimates were not available for the 2023 fishing year; therefore, a three-year average catch and CV were calculated using the estimates from 2020–2022. Unreported catch for 1949–2003 and the corresponding CV were identical to the base case under this scenario.

Sensitivity to alternative survey relative abundance index

The sensitivity to using an alternative index of relative abundance produced by the VAST standardization approach was evaluated by fitting to the alternative index using the identical methods for fitting to the base-case index. The alternative index was produced was the best-fit VAST model that used a poisson-link, delta-gamma structure. The base case for this assessment was the best-fit model that used a logistic delta-lognormal (Ducharme-Barth 2023), which was chosen because it had the lowest marginal AIC value. This sensitivity was included to determine the effect of a different model error structure for the survey index on overall assessment results.

Sensitivity to including absolute biomass estimates from fishery-independent survey

The sensitivity of including the design-based estimates of absolute abundance (Table 11) was evaluated by adjusting the observation error model for the survey in JABBA following the 2018 benchmark and 2021 update stock assessments. The observation error model for the survey related to the relative biomass estimate from the fishery-independent survey to the estimated proportion of carrying capacity in the process equations (Equation 6) scaled by survey catchability. The observed relative fishery-independent biomass estimate (S_T) was subject to natural sampling variation which was assumed to be lognormally distributed. The observation equation for the survey was

Equation 24: $S_T = \frac{1}{q_S} P_T K \cdot \gamma_{S,T}$,

where q_S was a scalar to translate relative biomass to absolute biomass (see below), and $\gamma_{S,T}$ was a lognormal random variable with mean equal to 1.0 and variance ξ_T, which was the variance of the natural logarithm of the survey in year T. The value for ξ was calculated based on the CV of the survey on the original scale as $\xi_T=\ln[CV_{S,T}^2 + 1]$. The scalar ($q_S$) was calculated based on the number of theoretical samples within a survey grid. This was calculated based on the estimated effective radius (*rad*) of a single sample scaled to the total area within a sampling grid (250,000 m^2), then multiplied by the number of sampling grids within the sampling domain (25,892) (see Ault et al. 2018) as

Equation 25: $q_S = \frac{250{,}000}{\pi \cdot rad^2} 25{,}892$

The effective radius of a single survey sample was assigned its own prior. The prior distribution for the radius $p(rad)$ was an informative lognormal distribution with mean (μ_{rad}) and variance (σ_{rad}^2) parameters:

Equation 26: $p(rad) = \frac{1}{rad\sigma_{rad}\sqrt{2\pi}} exp\left(-\frac{(l-\mu_{rad})^2}{2\sigma_{rad}^2}\right)$.

The prior mean for *rad* was 27.6 m, based on the best estimate from (Ault et al. 2018a). As in the 2018 and 2021 assessments, a CV of 50% was used for the *rad* prior. The minimum and maximum values, respectively, for the effective radius of a single sample were assumed to be 7.5 m and 60.6 m (Ault et al. 2018a and b). As such, the prior distribution of *rad* was also constrained to be between 7.5 m and 60.6 m in the model.

Sensitivity to Schaefer Model

The sensitivity to the production function shape was evaluated with one scenario. This scenario estimated the model using a Schaefer formulation by fixing the shape parameter, m, at 2.0 so that B_{MSY}/K=0.5.

Sensitivity to initial conditions

The sensitivity of base case model results to the initial conditions used for the MCMC sampler was evaluated. In the base case, initial conditions were set using a single random draw from the prior distribution of each parameter. Consequently, for this sensitivity, the assessment model was rerun ten times with different random seeds to allow for different initial conditions. The same random seeds were used for each chain.

3. Comparison with Single-Species Models and Data

3.1. Catch, CPUE, and Survey Data for Single-Species Model

Species-specific reported (Table 3) and unreported catch (Table 5) data for ʻōpakapaka were already calculated for the Deep 7 stock assessment model and were used for total catch (Table 7) in the single species model. Similarly, a model-based estimate for ʻōpakapaka biomass (Ducharme-Barth 2023) was included for the fishery-independent survey (Table 11).

Generating a CPUE index for ʻōpakapaka required additional analyses from what was done for the Deep 7 index. Fishers do not report targeted species when reporting catch data in the fisher-reported database. The previous filtering, described in section 2.5.2 Fishery Data Filtering Steps for CPUE, represented the best information available in determining targeted Deep 7 bottomfish fishing, but distinguishing targeting among the Deep 7 species required additional analysis

To create the standardized, single-species index, we used a subset of the catch and effort reported in the fisher-reported database. Specifically, we selected only the fishing trips that targeted ʻōpakapaka, as identified by a classification and regression tree (CART) model (Breiman 1984) using the approach of Langseth and Glover (2021). CART models are a flexible method that partitions data into natural groupings based on predictor variables (Langseth and Glover 2021). These predictor variables can be both continuous and categorical. In this model, we used the species composition of catch within fishing events to predict targeting of ʻōpakapaka.

The pre-processed fisher-reported database (see section 2.5.2 Fishery Data Filtering Steps for CPUE) was further filtered for ʻōpakapaka-targeting trips using the following steps. First, the species that comprised 99% of the catch were identified and only records of those 40 species were kept. This was to remove the influence of rarely caught species on the model. Next, the dataset was divided into two groups: a training and a testing group. 70% of the data were in the training group and the remaining 30% were in the testing group. Within both groups, there was an equal proportion of ʻōpakapaka presence and absences in fishing events, 54% and 45% respectively, which was representative of the full dataset. Then, within each fishing event, the presence or absence of each of the 40 species (including ʻōpakapaka) was determined.

A regression tree model was fit to the training data, with presence/absence (1 or 0) of ʻōpakapaka as the response variable and the presences/absences of the remaining 39 species as the explanatory variables. No environmental or fishing variables were used in the model because of incomplete records. A regression tree was fit using the R package “rpart” (version 4.1.19) and the initial tree was simplified using the “prune” function (Therneau et al. 2022) to reduce overfitting. The tree was pruned using the complexity parameter that minimized the cross-validation error of the model. From the pruned model, the species whose presence/absence best predicted the targeting of ʻōpakapaka were determined (Figure 11).

The top five most influential species were *Aphareus rutilans, Hyporthodus quernus, Aprion virescens, Etelis carbunculus,* and *E. coruscans* (Figure 11). Once the model was fit to the training dataset, we used the predicted probability that a fishing event targeted ʻōpakapaka (value ranging between 0 and 1) to determine the prediction threshold. We calculated the number of incorrect predictions (e.g., fishing event that caught ʻōpakapaka was predicted as not targeting) across the range of probabilities from 0 to 1. The value that minimized the number of incorrect predictions was used as the threshold; where above that value, trips were categorized as ʻōpakapaka events, and below that value they were categorized as non-ʻōpakapaka events. We found that prediction threshold values between 0.48 and 0.7 had an equal number of incorrect predictions, therefore, we used a threshold value of 0.5. At this threshold, the model predicted events targeting ʻōpakapaka correctly (i.e., ʻōpakapaka was present in the catch for that fishing event) 69% of the time on the training dataset. The model was then fit to the testing dataset. Fishing events with a predicted probability above the threshold of 0.5 from both the training and the testing datasets were selected as ʻōpakapaka events and used for the CPUE standardization.

3.2. CPUE standardization for Single-Species Model

The same methods used to calculate the standardized index of CPUE for the Deep 7 bottomfish complex described in section 2.5.3 CPUE standardization were also used to calculate the standardized index of ʻōpakapaka CPUE. The change in AIC, log-likelihood, and degrees of freedom for each predictor from the Bernoulli and lognormal processes for both time periods are provided in Appendix C (Table C1).

The best-fit ʻōpakapaka model for the early time period varied slightly from the best-fit model for the Deep 7 bottomfish complex. The difference for both the Bernoulli and lognormal processes was that pounds of uku was not selected (Table C1). The best fit model for the Bernoulli process reduced deviance by 22% relative to the null model (i.e., intercept only) and 4.6% from the model with only year and fisher. The best fit model for the lognormal process reduced deviance by 12.8% relative to the null model and 2.4% from the model with only year and fisher.

3.3. Assessment Model for Single-Species

Parameter values were changed within the Bayesian surplus production model to reflect the species being assessed. The prior distribution for carrying capacity was calculated similarly to the complex-level model, using eight times the highest catch for ‘ōpakapaka (mean = 6.1 million lb, CV=1.0), and the prior distribution for the initial proportion of carrying capacity was the same as for the base case model for the complex (mean = 0.75, CV=0.5). The prior distributions for r and the shape parameter, m, were informed by the species-specific estimates from the ASEM procedure. The lognormal prior for r had a mean = 0.096 and CV=0.28, and the prior for m corresponded to B_{MSY}/K = 0.315 with a CV=0.25. The CV for catch was 0.21, which was the average CV for ‘ōpakapaka from annual HMRFS estimates. The number of iterations, chains, and burn-in period were the same as in the Deep 7 complex JABBA model.

In addition to the single-species production model, an integrated, age-structured assessment model was developed and evaluated for ‘ōpakapaka using Stock Synthesis version 3.30.21 (Methot and Wetzel 2013; Supplemental Material A). One of the key advantages of adopting the Stock Synthesis framework is its capacity to facilitate the separation of data into distinct categories. This division enhances analytical clarity by methodically organizing information into well-defined groups, making it significantly easier to discern and understand underlying patterns. Specifically, the data inputs include commercial and non-commercial catch for ‘ōpakapaka, CPUE from the fisher reporting database (see section 3.1 Catch, CPUE, and Survey Data for Single-Species Model), and an index of relative abundance from the BFISH camera survey. Additionally, it has a separate index of relative abundance from the BFISH research fishing survey (Ducharme-Barth 2023), weight composition data from the commercial fisher reporting database, and length composition data from the BFISH camera and research fishing surveys (Figure 12).

In addition to auxiliary data, information from the most recent life history studies on natural mortality, growth, longevity, fecundity, and reproduction (Table 2) were fixed within the model. A third major difference from the JABBA model was the parameterization of fleet-specific selectivity. Logistic selectivity was estimated for the commercial fishery and the BFISH camera survey, while it was fixed for the non-commercial fishery. A double normal selectivity curve was fitted to the BFISH research fishing length data and the descending limb was freely estimated, allowing for dome-shaped selectivity. From the model we estimated biomass, harvest rate, and stock status and compared the estimates with the single-species JABBA model.

4. Results

4.1. Diagnostics

4.1.1. Convergence Diagnostics

Convergence diagnostics indicated that the MCMC simulation to estimate the posterior distribution of production model parameters converged to stationary distributions (Table 14). In particular, the Geweke diagnostic for all estimated parameters passed at a confidence level of α=0.05, indicating that the burn-in period was sufficient for removing any non-stationarity from the MCMC chains. The Gelman and Rubin potential scale reduction factors were equal to unity, confirming convergence to the posterior distribution. The Heidelberger and Welch stationarity and half-width diagnostic tests were also passed by every parameter at a confidence level of α=0.05. Autocorrelation was low for all parameters, and visual inspection of trace plots did not reveal any convergence issues. Overall, the diagnostics indicated that the base case assessment model reached convergence.

4.1.2. Model Diagnostics

Including estimable observation error for the FRS CPUE time series resulted in values of total observation error (i.e., the sum of uncertainty in the standardized CPUE estimates and the uncertainty estimated within the production model) that were fairly similar to values for the BFISH CPUE (Figure 13). Residual diagnostics, RMSE, and runs tests indicated the production model generally provided a good fit to the FRS standardized CPUE and BFISH index of relative abundance (Figure 14). The combined RMSE of residuals was very low, 9.6% (Figure 15), and the commercial CPUE time series passed the runs test with only one year residual falling greater than three standard deviations away from the expected value (Figure 16). While the BFISH index of relative abundance failed the runs tests, this is not surprising with a short time series and residuals were fairly small (Figure 16). Model residuals did not exhibit significant trends for the commercial (p=0.87) or fishery-independent (p=0.095) indices. Residuals were also normal fits to the commercial (p=0.27) and fishery-independent (p=0.27) indices.

Comparisons of assumed prior distributions and estimated posterior distributions showed that the priors were more informative for the intrinsic growth rate and ψ, but less informative for carrying capacity (Figure 17). The posterior mean (0.095) for intrinsic growth rate was 1% smaller than the prior mean (0.096). The posterior mean for B_{MSY}/K (0.329) was 0.3% larger than the prior mean for B_{MSY}/K (0.328). The posterior mean for ψ (0.878) was 17% larger than the prior mean (0.75). The similarity between the prior means and posterior means for intrinsic growth rate, B_{MSY}/K, and ψ—

which are all moderately informative priors—suggested that the priors were more informative for these parameters. The posteriors for estimated catch matched the priors for all years while accounting for uncertainty. Alternatively, the posterior mean for carrying capacity (21.08 million pounds) was 131% larger than the prior mean (9.14 million pounds).

Additionally, posterior distributions for catchability, process error, and observation errors were substantially different from prior distributions, which were chosen to be vague priors. Although prior distributions for quantities—such as MSY, B_{MSY}, and H_{MSY}—were derived from the priors of intrinsic growth rate, carrying capacity, and shape, the priors were less informative for MSY and B_{MSY} but more informative for H_{MSY}. The posterior mean for MSY (747 thousand pounds) was 108% larger than the prior mean (360 thousand pounds). The posterior mean for B_{MSY} (6.94 million pounds) was 129% larger than the prior mean (3.03 million pounds). However, the posterior mean for H_{MSY} (0.119) was only 0.8% larger than the prior mean (0.118). Overall, the observed data appeared to contain enough information to adjust the implied prior estimates for derived quantities.

Parameter correlations did not indicate a problem in model estimation (Figure 18). The scaling parameters for the indices of abundance were highly correlated with each other (0.98 magnitude) and correlations among carrying capacity, parameters affecting the scaling of relative indices (q_1 and q_2), and intrinsic growth rate (r) were high (up to 0.77 magnitude). However, the remaining correlations were less than 0.42 in magnitude.

Retrospective analysis of the estimated biomass and harvest rates from the assessment model indicated that model outputs did not exhibit substantial retrospective patterns in biomass (Figure 19) or harvest rate (Figure 20). There was a negative retrospective pattern for biomass (Mohn's rho = -0.12) and a positive retrospective pattern for harvest rate (Mohn's rho = 0.15). However, both Mohn's rho values were within acceptable range (Hurtado-Ferro et al. 2015). Data conflicts are the probable cause for the discrepancies in scale noted in the retrospective analysis. There is an apparent disparity between the commercial data CPUE trend and that from the survey. This data conflict, combined with the reliance on the data through high prior CVs for model parameters, led to the scaling differences observed in the retrospective analysis.

4.2. Stock Status

Production model estimates indicated that H_{MSY} was 11% and that B_{MSST} was 5.66 million pounds of exploitable Deep 7 bottomfish biomass with an associated MSY of 6.54 thousand pounds (Table 12).

- Median estimates of the MSY-based biological reference points of maximum sustainable yield for the reported catch (MSY ± one standard deviation, expressed in units of reported catch), the harvest rate to produce MSY (H_{MSY} ± one standard deviation) and the exploitable biomass to produce MSST (B_{MSST} ± one standard deviation) were: MSY = 709 thousand pounds (± 207 thousand pounds)
- H_{MSY} = 11% (± 5.1%)
- B_{MSST} = 5.66 million pounds (± 2.1 million pounds).

Deep 7 biomass exhibited a gradual long-term decline from near pristine conditions in the 1950s, until the early 2000s (Table 15, Figure 21). After 2006, the biomass began steadily increasing for the next 12 years and is currently stable. Stock biomass has not decreased below B_{MSST} at any point since 1949 and the harvest rate has not exceeded MFMT (Figure 22). The probability that the stock is overfished or overfishing is occurring in 2023 is extremely low, 9.6×10^{-3} and 9.3×10^{-3}, respectively (Table 15).

4.3. Stock Projections

The constant five-year catch projection scenarios showed the distribution of outcomes in the probability of overfishing, biomass, harvest rates, and the probability of depletion of Deep 7 bottomfish that would likely occur under alternative reported catch scenarios in the main Hawaiian Islands during 2025–2029 (Table 16, Table 17; Figure 23–Figure 26). Projections indicated that the amount of reported Deep 7 catch that would produce approximately a 50% chance of overfishing for each year from 2025 through 2029 was between 507 to 612 thousand pounds (Table 16; Figure 23). For comparison, the smallest Deep 7 reported catch that would lead to a roughly 40% chance of overfishing was 550 thousand pounds through 2025, 521 thousand pounds through 2026, 498 thousand pounds through 2027, 478 thousand pounds through 2028, and 464 thousand pounds through 2029 (Table 16; Figure 23). The reported catch required to achieve a 25% chance of overfishing from 2025 through 2029 varied across years from 397 to 459 thousand pounds (Table 16).

4.4. Sensitivity Analyses

Sensitivity of the model results varied by the parameter prior distributions, data inputs, or model assumptions that were tested. The model results were sensitive to the assumed prior mean for parameters B_{MSY}/K, K, and r and the use of prior distributions from the single-species ‘ōpakapaka model. The model was insensitive to the assumed prior mean for parameters ψ, the treatment of catch, and the treatment of the relative index of abundance, Of the 19 sensitivity runs, stock status in the terminal year relative to overfishing (H_{2023}/H_{CR}) and overfished (B_{2023}/B_{MSST}) reference points did not change in any of the models. Details on each sensitivity are provided below and summarized in Table 18.

4.4.1. Sensitivity to alternative prior distributions of intrinsic growth rate

The model results were sensitive to changes in the prior mean of the intrinsic growth rate ® parameter; reduced prior means had a greater impact on the results than increased prior means. Decreasing the prior mean by 25% and 50% resulted in the posterior estimate for *r* decreasing by 7% and 29.9% respectively (Table 18). Increasing the prior mean by 25% and 50% resulted in the posterior estimate for *r* increasing by 9.7% and 17.55% respectively. Estimates of *K* were inversely related, decreasing the prior mean by 25% and 50% led to *K* posterior estimates increasing by 3.72% and 25.32% respectively. Assuming higher prior means resulted in lower biomass estimates and increased harvest rate estimates compared to the base model (Figure 27). When the prior mean decreased by 50%, B_{MSST} increased by 24.7% and H_{MSY} decreased by 28.57%. This resulted in the probability of overfishing in 2023 to increase 137.8% and the probability of being overfished in 2023 to decreased 9.69%. However, status in the terminal year relative to reference points remained unchanged for all scenarios (Figure 28).

4.4.2. Sensitivity to alternative prior distributions of carrying capacity

The model results were sensitive to changes in the prior distribution of carrying capacity (Figure 29). Decreasing the mean of the prior by 50% resulted in a 13.1% decrease in exploitable biomass in the terminal year (Table 18). When the prior mean for *K* was increased by 25%, the posterior estimate for *K* was relatively unchanged (increased by less than 1%). However, when the prior mean was increased by 50%, the posterior estimates for the parameter *K* increased by 11.95% (Table 18). When the prior mean was reduced by 25% and 50%, the posterior estimates decreased by 9.75% and 14.34% respectively (Table 18). When the prior mean was reduced by 50%, estimates of B_{MSST} decreased by 14.2% and estimates of H_{MSY} increased by 15.2% (Table 18). Therefore, the probability of overfishing in 2023 decreased by 47.5% and the probability of being overfished in 2023 decreased to 61.6% (Table 18). The time series of B/B_{MSST} and H/H_{CR} were similar between sensitivity scenarios and the status in the terminal year relative to reference points did not change (Figure 30).

4.4.3. Sensitivity to alternative prior distributions of initial proportion of carrying capacity

The model results were insensitive to changes in the assumed prior mean for initial proportion of carrying capacity (ψ). For scenarios where the prior mean of ψ was decreased by 25% and 50%, posterior estimates of ψ decreased by 3.5% and 13% respectively. Posterior estimates from scenarios where the prior mean of ψ was increased by 25%, and 50% increased by 3.8% and 6.6% respectively (Table 18). For all scenarios of the prior mean of ψ, *K* estimates were increased from the base model (Table 18), except when ψ was decreased by 25%.

For all scenarios of the prior mean of ψ, *r* posterior mean estimates decreased from the base model except for when ψ prior mean was increased by 50%. Decreasing ψ prior mean by 50% resulted in terminal biomass and B_{MSST} estimates that were 5.2% lower and 3.5% higher respectively from the base model estimates (Table 18). Terminal harvest rates and H_{MSY} were 5.5% higher and 9.8% lower respectively than estimates from the base model (Table 18). This led to the probability of overfishing in 2023, and the probability of being overfished in 2023 increasing by 148% each. Overall, the time series of H/H_{CR} was higher for models with decreased prior mean ψ values (Figure 31). Additionally, the time series of B/B_{MSST} was lower for scenarios with decreased prior mean ψ values than the base model (Figure 31). However, based on these scenarios, changing the prior mean of ψ by up to ±50% would not change the status in the terminal year relative to reference points (Figure 32).

4.4.4. Sensitivity to alternative prior distributions of B_{MSY}/K

The model results were sensitive to changes in the assumed prior means for B_{MSY}/K, decreasing B_{MSY}/K by 50% had the greatest impact on model results (Figure 33). For scenarios where the prior mean of B_{MSY}/K was decreased by 25% and 50%, posterior estimates of B_{MSY}/K decreased by 14.6% and 33.54% respectively. When the prior mean of B_{MSY}/K was increased by 25% and 50%, posterior estimates increased by 11.9% and 22.3% respectively. For all scenarios, posterior estimates of *r* increased from the base model by 0.88% to 8.47% (Table 18).

Varying the prior mean for B_{MSY}/K had very little impact on the posterior estimate of ψ with estimates ranging from 0.83% to 2.01% larger than the base model estimate (Table 18). Varying the prior mean for B_{MSY}/K also had little impact on the posterior estimates of *K* compared to the base model (Table 18). Decreasing the prior mean for B_{MSY}/K by 50% resulted in 6.7% decrease in terminal biomass and a 40.1% decrease in B_{MSST} (Table 18). Terminal harvest rates and H_{MSY} were 7.2% lower and 101.8% higher respectively than estimates from the base model (Table 18).

This led to a decrease of 100% for both the probability of overfishing and of being overfished respectively compared to the base model (Table 18). The sensitivities decreasing B_{MSY}/K by 50% had the greatest impact on terminal year stock status relative to reference points, however, none of the sensitivity scenarios led to a change in terminal year status (Figure 34).

4.4.5. Sensitivity to ‘Ōpakapaka life history parameters

The model results were sensitive to using ‘ōpakapaka-specific priors (Table 18). Posterior estimates of the parameters r, K, B_{MSY}/K, and ψ varied from the base case model by 4.62%, 16.1%, 3.35%, and 6.6%, respectively (Table 18). The estimates of exploitable biomass and harvest rate were similar to the base model with the terminal year biomass being 26.6% higher (Table 18, Figure 35) and terminal year harvest rate being 1.8% higher than the base model estimates (Table 18, Figure 35). The probability of overfishing in 2023 and of being overfished in 2023 were 81.6% and 72.3% lower when using ‘ōpakapaka-specific priors than the base model. Overall, the status in 2023 relative to the reference points did not change when ‘ōpakapaka-specific priors were used (Figure 36).

4.4.6. Sensitivity to excluding catch variability and including HMRFS estimates of non-commercial catch

Model results were not sensitive to scenarios with no variability on catch or where annual HMRFS catch estimates were used for unreported catch estimates (Figure 37). Parameter estimates (r, K, B_{MSY}/K, and ψ) for the scenarios with no variability on catch and HMRFS annual estimates were less than 3% different from the base model (Table 18).

Terminal year biomass and B/B_{MSST} were 2.9% and 2.2% higher respectively for the scenario with no catch error compared to the base model (Table 18). Terminal year biomass and B/B_{MSST} for the scenario with annual HMRFS estimates (with catch error) were 3.2% and 1% higher respectively than the base model. Terminal harvest rates were 2.8% lower and 10.8% lower than the base model (Table 18) for the scenarios with no error on catch and annual HMRFS estimates respectively. Terminal H/H_{CR} estimates were 4.1% and 10.8% lower than the base model (Table 18).

Overall, the probability of overfishing in 2023 was decreased by 33.4% for no error on catch and decreased by 82% with annual HMRFS estimates scenarios, and the probability of being overfished in 2023 decreased by 36.7% for no error on catch and decreased by 85.8% with annual HMRFS estimates scenarios. The status in 2023 relative to reference points was not changed by either of the catch scenarios (Figure 38).

4.4.7. Sensitivity to treatment of the relative index of abundance

Model results were not sensitive to the treatment of the relative index of abundance (Table 18, Figure 39). Model results changed only slightly when a different standardization model was used (poisson-link, delta-gamma); parameter estimates (r, *K, B_{MSY}/K,* and ψ) were less than 3% different from the base case estimates, and final year estimates of biomass, harvest rate, and stock status were within 2% of the base case estimates (Table 18). When the index was treated as an absolute abundance in the model, parameter estimates (*r, K, B_{MSY}/K,* and ψ) varied slightly more from the base model, between 2.38% to 9.14%.

Terminal biomass and *B/B_{MSST}* for the model fitting to absolute estimates of abundance were 2.85% higher and 6.55% lower, respectively, than the base model and harvest rate and *H/H_{CR}* were 7.14% lower and 4.97% higher, respectively, than the base model estimates (Table 18). Estimates of *B/B_{MSST}* were similar for both BFISH index scenarios compared to the base model, however, a declining trend during the terminal two years was apparent when fitting to the absolute BFISH estimates.

These scenarios show that while treating the index as an index of absolute abundance does produce a different trend in the later portion of the time series, neither of the alternative models result in an overfished or overfishing status in the terminal year (Figure 40).

4.4.8. Sensitivity to the Schaefer model

The model results were sensitive to changes in the production model shape when *B_{MSY}/K* was fixed at 0.5 (Figure 41). When *B_{MSY}/K* was fixed at 0.5, the posterior estimate of *B_{MSY}/K* increased by 52.4% from the base model. Posterior estimates of parameters *r, K,* and ψ increased from the base model by 31.7%, 2.6%, and 2.5% respectively (Table 18). When *B_{MSY}/K* was fixed at 0.5, the terminal year biomass and harvest rates were similar to the base model, 6% greater and 5.7% smaller, respectively. However, estimates of *B_{MSST}* and *H_{MSY}* were 57% greater and 47.3% smaller, respectively (Table 18).

This led to an increase of 681.6% and 441.9% in the probability of overfishing and of being overfished respectively compared to the base model (Table 18). The sensitivities decreasing by 50% and Fixing *B_{MSY}/K* at 0.5 had an impact on terminal year stock status relative to reference points, however, it did not lead to a change in terminal year status (Figure 42).

4.4.9. Sensitivity to initial conditions

The model results were not sensitive to initial conditions. Estimates of *B_{2023}/B_{MSST}* varied from the base model by -1.63% to 2.68% and estimates of *H_{2023}/H_{CR}* varied from the base model by -5.3% to 3.2%. The stock status for the Deep 7 complex in 2023 relative to the reference points did not change when initial conditions were changed (Figure 43).

4.5. Single-species Modeling

Diagnostics indicated that the MCMC simulation to estimate the posterior distribution of production model parameters for the ʻōpakapaka model converged for all parameters (Table 19). Residual analyses indicated that the production model provided a good fit to the standardized CPUE observations and that residuals were normal and without trend (Figure 44–Figure 46). Posterior means of model parameters *r, m,* and ψ were all less than 15% different from the posterior mean estimates from the Deep 7 complex model (Table 19). Parameter and derived quantities related to the model scale, including *K,* B_{MSST}, and *MSY* were approximately proportional to the corresponding value in the Deep 7 complex model by 40–50%. This ratio corresponds to the estimated ratio of ʻōpakapaka biomass to Deep 7 biomass from the fishery-independent survey (42%, Table 19).

Of the remaining parameters and derived quantities, process error and catchability parameters differed the most from the complex model, ranging from 45% to 87% larger than the complex model estimates (Table 19). Posterior median and 95% credible intervals for biomass scaled to 44% of the biomass for the Deep 7 bottomfish complex over all the years (Figure 47). Posterior median of harvest rate for ʻōpakapaka averaged 30% greater than for the Deep 7 complex across all years (Figure 48). Given commensurate changes in the reference points B_{MSST} and H_{MSY} for the ʻōpakapaka model the relative status for ʻōpakapaka was similar to the status for Deep 7 bottomfish as a complex (Figure 49).

The age-structured model for ʻōpakapaka converged (final gradient of 8.7x 10^{-5} and positive definite hessian), fit the data well based the model diagnostics, and it provided estimates of biomass, harvest rate, and stock status similar to that of the ʻōpakapaka JABBA model (Supplemental Material A). The trend in exploitable biomass from the age-structured model is smoother (less annual variability) than the time series of biomass from the JABBA model, however the general trajectory is similar (Figure 50).

For both models, the stock shows a gradual decline from the beginning of the time series until the early 2000s when the biomass began increasing. The biomass estimates from the JABBA model are slightly lower than the estimates from the age-structured model, particularly in the initial few years and last few years of the model. However, the biomass estimates from the age-structured model are well within the 95% credible intervals from the JABBA model.

Harvest rate estimates from the age-structured model were similar to the estimates from the JABBA model (Figure 50). H_{MSY} estimates were different with H_{MSY} estimates from the age-structured model being slightly higher than the estimate from the JABBA model. Overall, the stock status from the age-structured model was the same as from the

JABBA model (i.e., not overfished and not undergoing, Figure 51), and terminal year status was slightly more optimistic from the age-structured model compared to the JABBA model.

5. Discussion

The Deep 7 bottomfish stock complex in the main Hawaiian Islands was categorized as not being overfished and not experiencing overfishing in 2023. The 2024 benchmark assessment produced estimates of biomass, harvest rate, and biological reference points that were indicative of a decreased scale in exploitable biomass but increased productivity relative to the previous assessment (Syslo et al. 2021). The terminal year for biomass estimation in the 2021 assessment update was 2018. Biomass in 2018 was estimated at 11.31 million pounds in the 2024 benchmark, a decrease from the corresponding estimate of 21.88 million pounds in 2018 from the 2021 update. The decrease in estimated biomass scale and increase in productivity were the result of numerous improvements in the data preparation and modeling approaches, which are highlighted throughout this report and discussed below, but are also influenced by the declining trend in the fishery-independent survey.

Projections produced reported catch values corresponding to probabilities of overfishing that were relatively similar to those in the 2021 update despite the difference in biomass scale. The amount of reported catch that would yield a 50% probability of overfishing was 507–612 thousand pounds in the 2024 assessment compared to 556–618 thousand pounds in the 2021 assessment. The smallest Deep 7 projected reported catch that would lead to a roughly P^*=40% chance of overfishing was about 464 thousand pounds in the 2024 assessment compared to 486 thousand pounds in the 2021 update. Forty percent was approximately the risk of overfishing chosen by the WPRFMC for setting the annual catch limit (ACL) for fishing years 2021–2024 based on the 2021 benchmark assessment. The 464 thousand pounds catch value is about 5.7% less than the ACL for the 2021–24 fishing season (492,000 pounds), based on the 2021 update. The fact that projected reported catches for a given probability of overfishing declined 5.7% despite a nearly 50% decline in biomass is largely explained by the difference in the shape of the surplus production function, which resulted in higher productivity at lower biomass ratios for the 2024 assessment.

This assessment included several improvements in data treatment and modeling over the framework used for the 2018 benchmark, which were implemented to address comments from reviewers of the 2018 benchmark and 2021 update assessments. Improvements fall under the general categories of changes to the treatment of noncommercial catch, changes to the treatment of relative abundance indices, and improvements to the modeling framework, which includes improved incorporation of prior information.

Previous reviews identified noncommercial catch as the greatest uncertainty influencing assessment results (Martell et al. 2017; Franklin 2021). Previous assessments used assumed ratios of noncommercial to reported commercial catch that were based on the best available studies and were not updated after the 2011 benchmark stock assessment (Brodziak et al. 2011). The current assessment made extensive use of the existing HMRFS framework to improve our understanding of the magnitude of unreported catch. Although annual species-specific estimates of noncommercial catch from HMRFS were variable (Ma et al. 2023), estimates converged on stable and plausible values as multiple years of data were aggregated.

The overall ratio of total noncommercial Deep 7 catch estimated from HMRFS to reported commercial catch was 1.09, which was similar to the value used for the final years of the assessment model and projections in the 2021 update (1.11). The inclusion of HMRFS data also provided a quantitative basis for variation in catch that could be propagated through the production model.

The current assessment re-evaluated the assumed ratio of noncommercial to commercial catch for the time period before HMRFS data became available. In an effort to reconstruct the previously used ratios, it was determined that they were based on a total commercial catch from the Hawaiian archipelago (including the northwestern Hawaiian Islands; Appendix B) to scale up (estimate) recreational/unreported catch and applying them to the main Hawaiian Islands resulted in values for total catch that were biased high. The use of the corrected ratios is one of the factors contributing to a decrease in the scale of stock biomass relative to previous assessments.

The commercial CPUE index was improved by evaluating the construct validity and influence of covariates—including cumulative experience, wind speed and direction, and hours reported (as a unit of effort)—which ultimately allowed the estimation of a single CPUE index from 1949 to 2023. Previous reviewers have recommended the development of a single CPUE series (Martell, S. et al. 2017; Franklin, E.C. 2020) given that splitting the series results in separate estimates of biomass scale, which reduces the amount of information on abundance and potentially leads to bias (Hoyle et al. 2024). We also improved the generation of the CPUE index from the best fit standardization models by using the predict-then aggregate approach (Hoyle et al. 2024). This approach is analogous to using marginal means, and is less influenced by the balance of the data among strata when compared to the approach used in previous Deep 7 assessments.

Previous review panels recommended further research into the effective radius of the camera gear for the fishery-independent survey, because the prior distribution for this quantity was influential for scaling biomass in previous assessments. The current assessment was fit to the survey as a relative index of abundance to avoid this scaling issue. The previous methodology for fitting the survey to absolute (i.e., design-based) biomass estimates was included as a sensitivity. The design-based estimates of biomass exhibited a steeper decline over time than the model-based estimates.

However, the effect on model results was minimal given the relatively large amount of uncertainty in the effective radius of the camera gear. The decline in BFISH abundance indices contrasted with the relative abundance trend in the fishery-dependent CPUE, and occurred during a period of relatively low catch.

Several covariates hypothesized to affect the abundance or catchability of bottomfish to the BFISH survey were investigated during the development of the model-based index (Ducharme-Barth2023). Covariates included in the model-based estimates (depth, substrate hardness, complexity, slope, island group) were similar to those used to stratify the design-based estimates (Ducharme-Barth 2023; Richards 2023). Additional covariates were ultimately not included. Ducharme-Barth (2023) discusses logistical issues with including environmental covariates such as wind and ocean conditions, which also exhibit spatial correlation, and potential topics for future research.

The model-based approach to developing the BFISH index is not sensitive to the swept-area assumption of the camera gear that is required for the design-based estimator. However, it is sensitive to the estimated relative difference in fishing power between the research fishing and camera gears (Ducharme-Barth 2023). Gear calibration trials were conducted during BFISH survey development (Richards et al. 2016), but have not been revisited. Refinements to the annual BFISH sampling have generally focused on reducing the CV for the design-based estimator, which requires a certain number of grids to be sampled (Richards 2023). Given finite resources, increasing the number of grids simultaneously sampled with both camera and research fishing gear will result in a decrease in the total number of grids that can be sampled in a given year. Thus, the tradeoff between reducing the total number of grids sampled (i.e., increasing the BFISH CV) and conducting enough grids with paired samples should be quantified.

The fishery-independent survey was designed for the three Deep 7 species that produce the majority of catch (ʻōpakapaka, ehu, and onaga; Richards 2023), which is a reasonable approach given that participants in the Deep 7 fishery generally target ʻōpakapaka or a combined group of ehu/onaga (Langseth and Glover 2021). The depth domain for the Deep 7 complex is from about 75 to 400 m. ʻŌpakapaka occupy the shallower portion of this range (≤250 m), ehu occur at a wider range of depths but mostly deeper than ʻōpakapaka, and onaga are found at at greater depths (>200 m; Langseth and Glover 2021; Ducharme-Barth, N. 2023).

The camera gear is only effective at depths up to 250 m (Richards 2023). Thus, researchers at PIFSC are assessing the feasibility of extending the depth range of cameras to that of the research fishing gear (i.e., 400 m) using non-obtrusive artificial lighting. Artificial lighting may also prove useful if the BFISH survey can be extended into evening and nighttime hours. Much of the fishing for ʻōpakapaka occurs during nighttime hours (Langseth and Glover 2021); therefore, the feasibility of conducting survey sampling during this time should continue to be investigated to determine whether daytime sampling is representative.

The current assessment incorporated the most-recent life-history information available for the Deep 7 species, a recommendation from reviews of previous assessments (Appendix A). This information was used within an age-structured modeling framework to develop the priors for the intrinsic rate of increase and shape of the production function. While the *r* prior generated from the ASEM approach was similar to previous assessments, the shape of the production function resulted in higher productivity at lower values of biomass. For the current assessment, B_{MSY}/K occurred at 0.329, whereas MSY was estimated to occur at 0.57 of *K* in the previous assessment.

We believe that the revision to the shape of the production function reflect the best available information for Deep 7 species as it incorporates growth, maturity, longevity, stock-recruit steepness, and fishery selectivity. This estimate also agrees with the available literature. For example, in a meta-analysis applying production models to 147 fish stocks across several taxa, Thorson et al (2012) found that the average value of B_{MSY}/K was 0.40 and the average estimate of B_{MSY}/K for perciformes was 0.353.

This assessment used JABBA to overcome several limitations with the previous modeling framework implemented in WinBUGS, which included a lower bound for the shape of the production function at 0.37 and difficulties associated with incorporating process error into projections. Transitioning the assessment to the JABBA framework involved incorporating additional differences that resulted in a decreased biomass scale (in addition to the production function shape, mentioned above). The first of these is a penalty that restricts the estimated proportion of carrying capacity from greatly exceeding *K* in any given year to avoid unrealistic estimates of biomass (Winker et al. 2018). A second difference was that the previous assessment reported the mean estimated biomass in a given year, whereas JABBA reports the median biomass, which is likely a more appropriate measure of central tendency for a variable assumed to be lognormally distributed.

This assessment continued the development of a single-species production-model for ʻōpakapaka that was first included in the 2018 benchmark stock assessment and added an age-structured model for ʻōpakapaka (Supplemental Material A). The review panel for the 2021 update recommended (as medium priority) that future stock assessments continue to present both the Deep 7 complex and single-species assessments for important species with sufficient information (Franklin 2021).

To meet this recommendation, we developed an integrated, age-structured model for ʻōpakapaka that incorporated the most recent life-history information and the full extent of suitable, available data for the species. The age-structured model revealed more nuanced information about the stock's biomass and dynamics. Additionally, it allowed us to understand the impacts of the limitations of a production model on the estimates of exploitable biomass and stock status relative to management reference points. The results of the age-structured model showed general agreement with the estimates from the single-species production model and adds support to the conclusions of the Deep 7 complex production model.

The panel also recommended further data collection and life history studies for other species in the complex (Franklin 2021). The ASEM approach used to evaluate the r prior and ratio of B_{MSY}/K for this assessment represents a step towards developing single species assessment models for Deep 7 species. The differences in life history characteristics among the species (Andrews et al. 2012; Andrews et al. 2019; Nichols 2019; Andrews and Scofield 2021) and in their spatial distribution and vulnerability to the fishery (Langseth and Glover 2021) highlight the need to reevaluate the continued use of a single complex to assess and manage Deep 7 species. Through the development of research track assessments, we plan to continue investigating the optimal assessment complexity, including the number of groups to divide the complex into, and the complexity of the assessment model to apply for each grouping based on the quality and quantity of available data (Syslo et al. 2022).

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Tables

Table 1. List of bottomfish species in the Hawaiian bottomfish management unit species complex. The current stock assessment provides an assessment of the status of the set of Deep 7 bottomfish species.

Common name	Local name	Scientific name	Deep7 species
Pink snapper	‘Ōpakapaka	*Pristipomoides filamentosus*	X
Longtail snapper	Onaga	*Etelis coruscans*	X
Squirrelfish snapper	Ehu	*Etelis carbunculus*	X
Sea bass	Hapu‘upu‘u	*Hyporthodus quernus*	X
Grey jobfish	Uku	*Aprion virescens*	-
Snapper	Gindai	*Pristipomoides zonatus*	X
Snapper	Kalekale	*Pristipomoides sieboldii*	X
Blue stripe snapper	Ta‘ape	*Lutjanus kasmira*	-
Yellowtail snapper	Yellowtail kalekale	*Pristipomoides auricilla*	-
Silver jaw jobfish	Lehi	*Aphareus rutilans*	X
Amberjack	Kahala	*Seriola dumerili*	-
Thick lipped trevally	Butaguchi	*Pseudocaranx dentex*	-
Giant trevally	White ulua	*Caranx ignobilis*	-
Black jack	Black ulua	*Caranx lugubris*	-

Table 2. . Life history parameters and sources for Deep 7 species. Parameters are theoretical maximum length (L_{inf}), growth coefficient (K), theoretical age when length is 0 (A0), length-weight relationship intercept (*LW A*) and slope (*LW B*), maximum age (A_{max}), natural mortality (M), length at 50% maturity (L_{m50}), length at 50% selectivity (L_{s50}), and steepness (h).

Species	Sex	L_{inf}	*K*	*A0*	*LW A*	*LW B*	A_{max}	*M*	L_{m50}	L_{s50}	*h*	Growth source	A_{max} source	Maturity source
Aphareus rutilans	F	83.3	0.13	-0.6	5.82E-05	2.77	30	0.18	46	33	0.72	Nadon et al. (2023)	Stepwise; Nadon et al. (2023)	Nadon et al. (2023)
Aphareus rutilans	M	83.3	0.13	-0.6	5.82E-05	2.77	30	0.18	46	33	0.72	Nadon et al. (2023)	Stepwise; Nadon et al. (2023)	Nadon et al. (2023)
Etelis carbunculus	F	54.9	0.23	0	1.59E-05	3.03	32	0.17	25.3	29	0.60	Nichols (2019)	Nichols (2019)	DeMartini (2016); Mean of MHI/NWHI
Etelis carbunculus	M	42.2	0.17	0	1.59E-05	3.03	32	0.17	25.3	29	0.60	Nichols (2019)	Nichols (2019)	DeMartini (2016); Mean of MHI/NWHI
Etelis coruscans	F	89.9	0.11	-1.5	4.25E-05	2.75	55	0.10	62.2	32	0.64	Andrews et al. (2021)	Andrews et al. (2021)	Reed et al. (2023)
Etelis coruscans	M	84.0	0.12	-1.6	4.25E-05	2.75	55	0.10	62.2	32	0.64	Andrews et al. (2021)	Andrews et al. (2021)	Reed et al. (2023)
Hyporthodus quernus	F	95.8	0.08	-2.4	2.00E-05	3.00	50	0.11	58	47	0.75	Andrews et al. (2019)	Andrews et al. (2019); Could be as high as 70 yr.	DeMartini et al.(2011)
Hyporthodus quernus	M	95.8	0.08	-2.4	2.00E-05	3.00	50	0.11	58	47	0.75	Andrews et al. (2019)	Andrews et al. (2019); Could be as high as 70 yr.	DeMartini et al. (2011)

Species	Sex	L_{inf}	*K*	*A0*	*LW A*	*LW B*	A_{max}	*M*	L_{m50}	L_{s50}	*h*	Growth source	A_{max} source	Maturity source
Pristipomoides filamentosus	F	67.5	0.24	-0.3	1.75E-05	2.99	40	0.14	40.7	36	0.76	Andrews et al. (2012)	Andrews et al. (2012); 44+ yrs.	Leurs et al. (2018)
Pristipomoides filamentosus	M	67.5	0.24	-0.3	1.75E-05	2.99	40	0.14	40.7	36	0.76	Andrews et al. (2012)	Andrews et al. (2012); 44+ yrs.	Luers et al. (2018)
Pristipomoides sieboldii	F	44.4	0.09	-6.3	2.43E-05	2.91	38	0.14	23.8	39	0.75	Uehara et al.(2020)	Uehara et al. (2020)	DeMartini (2016)
Pristipomoides sieboldii	M	44.4	0.09	-6.3	2.43E-05	2.91	38	0.14	23.8	39	0.75	Uehara et al. (2020)	Uehara et al. (2020)	DeMartini (2016)
Pristipomoides zonatus	F	42.5	0.38	-1.2	1.80E-05	3.04	30	0.18	29.7	23	0.73	Andrews et al. (2021)	Schemmel et al. (2021)	Andrews and Scofield (2021); Sex-pooled together
Pristipomoides zonatus	M	42.5	0.38	-1.2	1.80E-05	3.04	30	0.18	29.7	23	0.73	Andrews et al. (2021)	Schemmel et al.(2021)	Andrews and Scofield (2021); Sex-pooled together

Table 3. Reported catch (units are 1000 lb) by Deep 7 species, 1949–2023 fishing years.

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1949	29.5	36.8	112.0	103.4	66.6	5.7	0.2	354.1
1950	18.6	29.1	113.6	75.2	60.0	4.6	0.8	301.8
1951	20.5	32.1	124.4	65.6	72.2	2.8	2.0	319.5
1952	27.8	45.3	118.8	52.0	44.5	9.5	2.7	300.5
1953	19.8	32.4	100.6	50.4	49.5	2.8	2.0	257.5
1954	16.7	40.2	102.3	40.8	65.5	3.9	1.9	271.2
1955	18.4	28.5	80.8	30.1	61.7	1.1	2.6	223.4
1956	23.3	32.5	104.5	39.7	68.8	3.8	3.7	276.2
1957	17.4	29.7	147.2	36.8	76.1	8.7	2.1	318.2
1958	17.5	17.5	92.6	26.8	52.2	2.4	2.0	211.1
1959	15.7	19.2	77.8	22.8	65.0	2.1	1.4	204.1
1960	12.4	19.3	70.6	19.3	39.0	1.6	1.2	163.3
1961	6.2	19.6	56.1	12.9	32.9	1.0	0.4	129.0
1962	9.7	16.3	75.7	15.3	48.5	1.7	0.7	167.8
1963	12.3	18.2	92.4	23.7	60.7	2.7	0.8	210.8
1964	10.6	23.5	92.4	24.7	47.1	1.0	2.3	201.6
1965	10.5	14.6	103.6	20.3	59.6	1.3	0.9	210.7
1966	11.6	13.6	71.4	18.1	64.1	2.0	0.8	181.6
1967	10.6	9.6	121.2	18.4	68.4	2.4	0.8	231.4
1968	11.3	6.9	85.1	19.9	69.5	2.2	0.8	195.6
1969	10.9	4.2	85.9	16.2	53.9	5.8	0.5	177.5
1970	20.1	5.1	69.7	15.9	43.5	2.7	1.4	158.4
1971	14.5	4.3	59.1	15.3	39.3	1.8	0.9	135.2
1972	17.6	8.1	117.9	21.3	58.7	4.4	1.2	229.3
1973	14.9	5.1	93.4	14.6	35.6	4.5	1.3	169.3
1974	14.6	4.9	135.3	21.1	43.6	4.9	1.5	225.8
1975	23.2	6.0	116.2	21.9	45.1	8.5	1.4	222.2

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1976	22.4	7.9	105.5	31.3	80.2	10.3	1.2	258.9
1977	30.1	8.6	106.3	35.7	84.8	7.3	1.5	274.3
1978	28.6	9.8	154.6	35.7	66.5	9.8	2.6	307.6
1979	29.6	7.8	146.0	22.5	53.0	12.1	2.9	273.8
1980	17.7	7.0	151.1	17.0	31.3	17.7	2.4	244.3
1981	17.0	8.2	197.4	21.3	42.9	19.9	1.9	308.5
1982	21.7	8.1	177.7	24.5	65.9	30.0	1.6	329.5
1983	32.7	15.0	230.5	28.5	72.8	28.5	2.7	410.6
1984	27.0	13.4	158.7	35.9	86.5	16.8	3.5	341.9
1985	31.1	22.2	196.9	40.4	163.4	25.5	4.5	484.1
1986	24.9	25.0	173.7	60.7	194.9	27.7	3.5	510.4
1987	28.7	28.4	258.2	48.6	174.0	38.7	3.3	579.9
1988	11.0	18.2	301.6	43.0	159.2	38.2	2.0	573.3
1989	15.5	11.2	309.6	42.0	146.4	44.7	1.8	571.1
1990	14.6	15.5	210.0	38.2	143.1	34.9	2.8	459.1
1991	16.6	19.7	136.5	31.9	104.2	19.0	3.6	331.6
1992	15.8	28.1	172.5	31.9	91.8	17.2	5.1	362.3
1993	12.0	17.2	134.2	23.8	52.6	11.1	3.6	254.5
1994	11.0	18.4	173.6	23.0	68.1	11.7	3.8	309.6
1995	16.1	29.2	188.3	25.8	68.9	13.6	4.0	345.9
1996	9.8	18.7	139.8	28.1	65.4	9.7	2.9	274.3
1997	14.1	22.8	160.5	26.1	61.3	11.8	3.0	299.7
1998	12.7	24.6	149.7	26.4	70.8	9.4	3.4	296.9
1999	9.9	11.1	104.0	19.5	59.4	8.8	2.3	215.0
2000	13.1	15.9	167.0	26.7	72.7	11.2	3.2	309.8
2001	15.5	15.3	125.0	26.5	63.0	11.5	3.6	260.4
2002	9.2	11.4	108.0	17.3	60.5	11.8	2.5	220.7
2003	9.5	12.3	128.8	16.5	70.4	8.6	2.2	248.2
2004	7.9	8.1	88.4	19.2	75.8	4.9	2.1	206.3

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai	Total
2005	10.4	7.8	104.4	22.6	89.8	7.0	2.0	244.0
2006	7.2	5.5	76.0	18.9	74.4	6.3	1.6	190.0
2007	7.5	6.1	92.4	19.5	85.5	8.4	2.3	221.8
2008	6.6	5.5	96.2	18.3	55.9	11.0	2.8	196.3
2009	7.9	9.6	133.4	24.5	59.2	16.8	3.6	255.1
2010	8.3	8.2	105.4	24.7	58.0	6.1	2.8	213.5
2011	8.2	10.0	149.4	24.7	67.8	11.5	3.1	274.8
2012	9.9	11.3	105.5	25.5	53.2	7.9	3.7	216.9
2013	10.6	12.3	96.0	30.1	66.8	13.0	3.4	232.2
2014	10.6	19.0	160.8	31.3	75.7	8.4	3.8	309.5
2015	9.3	17.8	154.6	32.1	79.9	12.5	2.8	309.1
2016	10.2	13.1	137.5	31.4	62.1	7.7	1.9	263.9
2017	8.3	10.6	135.3	25.9	47.2	9.1	2.2	238.8
2018	9.5	12.4	119.3	21.8	66.5	8.7	2.6	240.8
2019	5.8	9.9	70.1	23.0	58.8	7.0	2.7	177.3
2020	6.6	11.4	64.4	26.6	43.0	8.8	5.6	166.3
2021	4.0	10.6	56.8	28.4	43.3	8.9	5.6	157.6
2022	4.2	12.0	81.9	30.6	47.7	7.6	5.7	189.7
2023	3.7	10.3	98.3	27.6	48.2	8.2	5.6	201.8

Table 4. Ratio of unreported to reported catch for Deep 7 species, 1949–2023 fishing years.

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai
1949	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1950	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1951	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1952	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1953	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1954	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1955	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1956	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1957	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1958	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1959	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1960	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1961	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1962	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1963	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1964	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1965	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1966	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1967	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1968	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1969	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1970	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1971	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1972	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1973	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1974	0.39	0.01	1.80	0.52	0.59	0.09	0.14
1975	0.47	0.02	1.47	0.72	0.54	0.03	0.09

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai
1976	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1977	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1978	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1979	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1980	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1981	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1982	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1983	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1984	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1985	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1986	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1987	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1988	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1989	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1990	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1991	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1992	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1993	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1994	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1995	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1996	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1997	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1998	0.47	0.02	1.47	0.72	0.54	0.03	0.09
1999	0.47	0.02	1.47	0.72	0.54	0.03	0.09
2000	0.47	0.02	1.47	0.72	0.54	0.03	0.09
2001	0.47	0.02	1.47	0.72	0.54	0.03	0.09
2002	0.47	0.02	1.47	0.72	0.54	0.03	0.09
2003	0.47	0.02	1.47	0.72	0.54	0.03	0.09
2004	1.56	1.55	1.03	1.57	0.86	0.71	2.09

Year	Hapu‘upu‘u	Kalekale	‘Ōpakapaka	Ehu	Onaga	Lehi	Gindai
2005	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2006	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2007	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2008	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2009	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2010	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2011	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2012	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2013	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2014	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2015	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2016	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2017	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2018	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2019	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2020	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2021	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2022	1.56	1.55	1.03	1.57	0.86	0.71	2.09
2023	1.56	1.55	1.03	1.57	0.86	0.71	2.09

Table 5. Unreported catch (units are 1000 lb) of Deep 7 by species, 1949–2023 fishing years.

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1949	11.4	0.5	201.6	53.7	39.1	0.5	0.0	306.8
1950	7.2	0.4	204.5	39.1	35.2	0.4	0.1	286.9
1951	7.9	0.4	224.0	34.1	42.4	0.2	0.3	309.3
1952	10.7	0.6	214.0	27.0	26.1	0.8	0.4	279.6
1953	7.7	0.4	181.1	26.2	29.1	0.2	0.3	244.9
1954	6.5	0.5	184.3	21.2	38.5	0.3	0.3	251.5
1955	7.1	0.4	145.6	15.7	36.2	0.1	0.4	205.4
1956	9.0	0.4	188.2	20.6	40.4	0.3	0.5	259.4
1957	6.7	0.4	265.2	19.1	44.7	0.8	0.3	337.1
1958	6.8	0.2	166.7	13.9	30.7	0.2	0.3	218.8
1959	6.1	0.2	140.1	11.8	38.2	0.2	0.2	196.8
1960	4.8	0.2	127.1	10.0	22.9	0.1	0.2	165.4
1961	2.4	0.2	101.1	6.7	19.3	0.1	0.1	129.9
1962	3.7	0.2	136.3	7.9	28.5	0.1	0.1	176.9
1963	4.8	0.2	166.5	12.3	35.7	0.2	0.1	219.7
1964	4.1	0.3	166.5	12.8	27.6	0.1	0.3	211.7
1965	4.1	0.2	186.5	10.5	35.0	0.1	0.1	236.5
1966	4.5	0.2	128.6	9.4	37.6	0.2	0.1	180.6
1967	4.1	0.1	218.3	9.5	40.2	0.2	0.1	272.5
1968	4.4	0.1	153.2	10.3	40.8	0.2	0.1	209.1
1969	4.2	0.1	154.8	8.4	31.7	0.5	0.1	199.7
1970	7.8	0.1	125.5	8.2	25.6	0.2	0.2	167.6

Year	Hapu‘upu‘u	Kalekale	‘Ōpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1971	5.6	0.1	106.5	7.9	23.1	0.2	0.1	143.4
1972	6.8	0.1	212.3	11.1	34.5	0.4	0.2	265.3
1973	5.8	0.1	168.1	7.6	20.9	0.4	0.2	203.0
1974	5.7	0.1	243.7	11.0	25.6	0.4	0.2	286.6
1975	11.0	0.1	170.3	15.7	24.2	0.2	0.1	221.6
1976	10.6	0.2	154.6	22.4	43.1	0.3	0.1	231.2
1977	14.2	0.2	155.8	25.6	45.5	0.2	0.1	241.7
1978	13.5	0.2	226.7	25.6	35.7	0.3	0.2	302.2
1979	14.0	0.2	214.0	16.1	28.5	0.3	0.3	273.3
1980	8.4	0.1	221.4	12.2	16.8	0.5	0.2	259.6
1981	8.0	0.2	289.3	15.3	23.0	0.5	0.2	336.4
1982	10.2	0.2	260.5	17.6	35.4	0.8	0.1	324.8
1983	15.4	0.3	337.8	20.4	39.1	0.7	0.2	414.1
1984	12.8	0.3	232.7	25.8	46.5	0.4	0.3	318.7
1985	14.7	0.5	288.5	29.0	87.8	0.7	0.4	421.5
1986	11.8	0.5	254.6	43.5	104.7	0.7	0.3	416.1
1987	13.6	0.6	378.5	34.8	93.5	1.0	0.3	522.2
1988	5.2	0.4	442.0	30.9	85.5	1.0	0.2	565.1
1989	7.3	0.2	453.7	30.1	78.6	1.1	0.2	571.3
1990	6.9	0.3	307.8	27.4	76.8	0.9	0.3	420.4
1991	7.9	0.4	200.1	22.9	56.0	0.5	0.3	288.0
1992	7.5	0.6	252.8	22.9	49.3	0.4	0.5	333.9
1993	5.7	0.4	196.7	17.0	28.2	0.3	0.3	248.6

Year	Hapu‘upu‘u	Kalekale	‘Ōpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1994	5.2	0.4	254.5	16.5	36.6	0.3	0.3	313.8
1995	7.6	0.6	276.0	18.5	37.0	0.3	0.4	340.4
1996	4.7	0.4	204.9	20.1	35.1	0.2	0.3	265.7
1997	6.6	0.5	235.2	18.7	32.9	0.3	0.3	294.6
1998	6.0	0.5	219.4	18.9	38.0	0.2	0.3	283.4
1999	4.7	0.2	152.4	14.0	31.9	0.2	0.2	203.7
2000	6.2	0.3	244.8	19.1	39.0	0.3	0.3	310.1
2001	7.3	0.3	183.2	19.0	33.8	0.3	0.3	244.3
2002	4.3	0.2	158.4	12.4	32.5	0.3	0.2	208.3
2003	4.5	0.3	188.8	11.8	37.8	0.2	0.2	243.6
2004	12.4	12.5	91.4	30.2	65.5	3.5	4.4	219.9
2005	16.3	12.2	108.1	35.5	77.5	5.0	4.2	258.7
2006	11.3	8.6	78.7	29.6	64.3	4.5	3.4	200.4
2007	11.8	9.6	95.6	30.6	73.9	6.0	4.8	232.2
2008	10.3	8.6	99.6	28.7	48.3	7.9	5.9	209.2
2009	12.4	14.9	138.0	38.6	51.2	12.0	7.6	274.6
2010	12.9	12.7	109.1	38.9	50.1	4.4	5.8	233.9
2011	12.9	15.5	154.7	38.8	58.6	8.2	6.5	295.1
2012	15.4	17.5	109.2	40.1	46.0	5.6	7.7	241.4
2013	16.5	19.1	99.4	47.4	57.7	9.3	7.0	256.4
2014	16.5	29.5	166.4	49.2	65.3	6.0	8.0	340.9
2015	14.6	27.7	160.0	50.4	69.0	9.0	6.0	336.6
2016	16.0	20.4	142.2	49.4	53.7	5.5	4.0	291.1

Year	Hapu‘upu‘u	Kalekale	‘Ōpakapaka	Ehu	Onaga	Lehi	Gindai	Total
2017	13.0	16.5	140.0	40.7	40.8	6.5	4.7	262.3
2018	14.9	19.3	123.5	34.3	57.4	6.2	5.4	261.0
2019	9.1	15.4	72.5	36.2	50.7	5.0	5.6	194.6
2020	10.3	17.8	66.6	41.8	37.2	6.3	11.7	191.5
2021	6.3	16.5	58.7	44.6	37.4	6.3	11.8	181.7
2022	6.6	18.6	84.8	48.0	41.2	5.4	12.0	216.6
2023	5.8	15.9	101.8	43.3	41.7	5.8	11.7	226.0

Table 6. Estimates of noncommercial catch for all Deep 7 species summed from HMRFS (million lb) and coefficient of variation (CV) by fishing year, 2004–2022.

Fishing Year	Total	CV
2004	0.163	0.29
2005	0.207	0.26
2006	0.197	0.24
2007	0.185	0.50
2008	0.183	0.29
2009	0.233	0.26
2010	0.209	0.24
2011	0.333	0.26
2012	0.320	0.22
2013	0.384	0.21
2014	0.445	0.19
2015	0.325	0.25
2016	0.183	0.23
2017	0.160	0.24
2018	0.100	0.37
2019	0.441	0.22
2020	0.200	0.23
2021	0.112	0.26
2022	0.264	0.18

Table 7. Total catch (units are 1000 lb) of Deep 7 bottomfish by species in the main Hawaiian Islands by fishing year (July 1–June 30), 1949-2023, as calculated from the sum of reported catch (Table 3) and unreported catch (Table 5).

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1949	41.0	37.2	313.6	157.1	105.6	6.2	0.2	660.9
1950	25.8	29.4	318.1	114.3	95.2	5.0	0.9	588.7
1951	28.4	32.5	348.3	99.7	114.7	3.1	2.2	628.8
1952	38.5	45.8	332.8	79.1	70.6	10.3	3.1	580.2
1953	27.5	32.8	281.7	76.5	78.6	3.1	2.2	502.4
1954	23.1	40.7	286.6	61.9	104.0	4.2	2.1	522.7
1955	25.5	28.9	226.4	45.8	97.9	1.2	3.0	428.8
1956	32.3	32.9	292.6	60.3	109.1	4.2	4.2	535.6
1957	24.1	30.1	412.4	55.9	120.8	9.5	2.5	655.3
1958	24.3	17.7	259.3	40.8	82.9	2.6	2.3	429.8
1959	21.8	19.5	217.9	34.7	103.2	2.3	1.6	400.9
1960	17.2	19.5	197.7	29.4	61.8	1.7	1.4	328.7
1961	8.6	19.8	157.2	19.7	52.2	1.0	0.5	258.9
1962	13.4	16.5	211.9	23.2	77.0	1.8	0.8	344.7
1963	17.1	18.5	258.9	36.0	96.4	2.9	0.9	430.5
1964	14.6	23.8	258.9	37.6	74.7	1.1	2.6	413.3
1965	14.5	14.8	290.1	30.8	94.6	1.4	1.1	447.3
1966	16.1	13.7	200.0	27.5	101.7	2.2	0.9	362.2
1967	14.7	9.7	339.5	27.9	108.6	2.6	0.9	503.9
1968	15.7	7.0	238.3	30.2	110.3	2.4	0.9	404.7
1969	15.2	4.2	240.7	24.6	85.6	6.3	0.5	377.2

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1970	27.8	5.2	195.2	24.1	69.1	2.9	1.6	326.0
1971	20.1	4.4	165.6	23.3	62.3	2.0	1.0	278.6
1972	24.5	8.2	330.1	32.4	93.2	4.8	1.4	494.6
1973	20.7	5.2	261.5	22.1	56.5	4.9	1.4	372.4
1974	20.3	4.9	379.0	32.1	69.2	5.3	1.7	512.4
1975	34.2	6.1	286.5	37.6	69.2	8.7	1.5	443.8
1976	33.0	8.1	260.1	53.7	123.3	10.6	1.3	490.1
1977	44.3	8.8	262.2	61.4	130.3	7.5	1.7	516.1
1978	42.1	10.0	381.3	61.3	102.2	10.0	2.8	609.8
1979	43.5	7.9	360.0	38.6	81.5	12.4	3.1	547.2
1980	26.1	7.2	372.5	29.3	48.2	18.1	2.6	503.9
1981	25.0	8.4	486.6	36.6	65.9	20.4	2.0	644.9
1982	31.9	8.2	438.2	42.1	101.4	30.7	1.8	654.4
1983	48.1	15.3	568.3	48.9	111.9	29.2	2.9	824.7
1984	39.8	13.7	391.4	61.7	133.0	17.2	3.9	660.6
1985	45.8	22.7	485.4	69.4	251.2	26.2	5.0	905.6
1986	36.7	25.5	428.3	104.2	299.6	28.4	3.8	926.5
1987	42.2	29.0	636.7	83.4	267.5	39.7	3.6	1102.1
1988	16.3	18.5	743.6	73.9	244.7	39.2	2.2	1138.4
1989	22.8	11.4	763.3	72.0	224.9	45.9	2.0	1142.3
1990	21.6	15.8	517.8	65.6	219.9	35.7	3.0	879.5
1991	24.5	20.1	336.6	54.8	160.2	19.5	4.0	619.6
1992	23.3	28.7	425.3	54.8	141.0	17.6	5.6	696.3

Year	Hapuʻupuʻu	Kalekale	ʻŌpakapaka	Ehu	Onaga	Lehi	Gindai	Total
1993	17.6	17.5	330.9	40.8	80.8	11.4	4.0	503.1
1994	16.2	18.8	428.1	39.5	104.7	11.9	4.1	623.4
1995	23.7	29.8	464.3	44.3	105.9	13.9	4.4	686.4
1996	14.5	19.0	344.6	48.2	100.5	9.9	3.1	540.0
1997	20.7	23.3	395.7	44.9	94.3	12.1	3.3	594.3
1998	18.7	25.1	369.1	45.3	108.9	9.6	3.7	580.3
1999	14.6	11.3	256.4	33.5	91.3	9.0	2.5	418.7
2000	19.4	16.2	411.9	45.9	111.7	11.5	3.5	619.9
2001	22.8	15.6	308.2	45.4	96.8	11.8	4.0	504.7
2002	13.5	11.6	266.4	29.7	93.0	12.1	2.7	429.0
2003	14.0	12.5	317.7	28.3	108.2	8.8	2.4	491.9
2004	20.3	20.6	179.8	49.4	141.3	8.4	6.4	426.2
2005	26.7	20.0	212.5	58.0	167.3	11.9	6.2	502.7
2006	18.6	14.1	154.7	48.5	138.7	10.8	5.0	390.3
2007	19.3	15.7	188.0	50.0	159.4	14.4	7.1	454.0
2008	16.9	14.1	195.7	47.0	104.1	18.9	8.7	405.5
2009	20.3	24.5	271.4	63.1	110.4	28.8	11.2	529.6
2010	21.2	20.9	214.5	63.6	108.1	10.5	8.6	447.5
2011	21.1	25.5	304.1	63.5	126.4	19.7	9.6	569.9
2012	25.3	28.8	214.7	65.6	99.2	13.5	11.4	458.3
2013	27.1	31.5	195.4	77.5	124.5	22.3	10.4	488.6
2014	27.1	48.4	327.2	80.5	141.0	14.5	11.8	650.4
2015	23.9	45.5	314.6	82.5	148.9	21.5	8.8	645.8

Year	Hapu‘upu‘u	Kalekale	‘Ōpakapaka	Ehu	Onaga	Lehi	Gindai	Total
2016	26.2	33.5	279.7	80.8	115.8	13.2	5.9	555.1
2017	21.4	27.2	275.3	66.5	88.0	15.7	6.9	501.1
2018	24.4	31.7	242.8	56.1	123.9	14.9	8.0	501.8
2019	14.8	25.4	142.6	59.2	109.5	12.0	8.3	371.8
2020	16.9	29.2	131.0	68.3	80.2	15.1	17.2	357.8
2021	10.4	27.2	115.5	73.1	80.6	15.2	17.4	339.3
2022	10.7	30.6	166.7	78.6	88.9	13.0	17.7	406.3
2023	9.4	26.2	200.1	70.9	89.9	14.0	17.3	427.8

Table 8. Proportion of records with individual name information from 1949–1990. All records had individual name information after 1990.

Year	Proportion	Year	Proportion
1949	0.8	**1970**	0.7
1950	0.7	**1971**	0.6
1951	0.7	**1972**	0.7
1952	0.7	**1973**	0.6
1953	0.6	**1974**	0.7
1954	0.3	**1975**	0.6
1955	0.3	**1976**	0.0
1956	0.3	**1977**	1.0
1957	0.3	**1978**	1.0
1958	0.3	**1979**	1.0
1959	0.5	**1980**	1.0
1960	0.6	**1981**	1.0
1961	0.7	**1982**	0.9
1962	0.6	**1983**	1.0
1963	0.6	**1984**	0.9
1964	0.7	**1985**	0.9
1965	0.6	**1986**	1.0
1966	0.6	**1987**	1.0
1967	0.6	**1988**	1.0
1968	0.6	**1989**	1.0
1969	0.6	**1990**	1.0

Table 9. Summary of log-likelihood values and reduction in AIC (ΔAIC = AIC previous model – AIC proposed model) during model selection for the best-fit model for the Bernoulli and lognormal processes using maximum likelihood. Each parameter is added to the model with all previously selected parameters included. The year predictor was included in all baseline models and was added first among fixed effects in model selection.

Process	Selected predictor	Delta	Log-likelihood	Number of parameters
Bernoulli				
	null	0	-35836.4	1
	+fisher	14274.3	-28698.3	2
	+year	1685.4	-27781.6	76
	+area	1129.6	-27129.8	163
	+area:year	1890	-26183.8	164
	+uku	852.4	-25756.6	165
	+quarter	509.6	-25498.8	168
Lognormal				
	null	0	-203702.3	1
	+fisher	47315	-180043.8	3
	+year	1713.5	-179113.1	77
	+area	3461.4	-177245.4	164
	+area:year	3066.1	-175711.4	165
	+quarter	1738.4	-174839.1	168
	+uku	1249.4	-174213.4	169

Table 10. Annual index of commercial standardized CPUE and standard error (SE).

Year	Estimated CPUE	SE	Year	Estimated CPUE	SE
1949	116.99	7.41	**1983**	72.84	2.61
1950	109.84	6.75	**1984**	62.01	2.37
1951	124.43	7.90	**1985**	82.18	2.75
1952	149.48	9.80	**1986**	79.46	2.63
1953	152.69	9.37	**1987**	89.69	2.69
1954	148.91	9.83	**1988**	99.26	3.14
1955	151.37	10.19	**1989**	99.45	3.58
1956	105.61	7.15	**1990**	87.72	3.29
1957	109.09	7.50	**1991**	84.44	2.77
1958	79.53	5.58	**1992**	77.55	2.45
1959	90.65	6.51	**1993**	76.55	2.85
1960	107.00	7.13	**1994**	78.00	2.68
1961	122.33	9.26	**1995**	78.99	2.61
1962	141.16	9.35	**1996**	70.48	2.26
1963	117.30	6.59	**1997**	77.74	2.58
1964	117.61	7.57	**1998**	71.17	2.44
1965	149.82	9.73	**1999**	74.10	2.68
1966	107.52	6.86	**2000**	86.46	2.96
1967	113.88	7.11	**2001**	76.04	2.75
1968	109.73	7.68	**2002**	68.81	2.69
1969	118.15	7.36	**2003**	69.52	2.38
1970	96.91	6.10	**2004**	62.08	2.37
1971	103.34	6.58	**2005**	70.10	3.04
1972	139.50	8.31	**2006**	71.69	3.42
1973	106.92	6.38	**2007**	72.09	3.65
1974	125.33	6.06	**2008**	79.03	3.53
1975	115.50	6.43	**2009**	69.14	2.87
1976	-	-	**2010**	64.71	2.94
1977	85.48	3.93	**2011**	72.07	2.57
1978	106.20	5.07	**2012**	69.35	2.65
1979	107.60	4.95	**2013**	75.72	3.37
1980	93.35	3.92	**2014**	79.40	3.23
1981	86.73	3.88	**2015**	95.00	3.90
1982	89.94	3.64	**2016**	84.61	3.53

Year	Estimated CPUE	SE	Year	Estimated CPUE	SE
2017	88.87	3.69	**2021**	78.57	3.40
2018	89.82	3.77	**2022**	82.45	3.44
2019	86.56	3.61	**2023**	90.64	3.80
2020	81.60	3.59	**-**	-	-

Table 11. Biomass estimates from the fishery-independent survey for fishing years 2017–2023 for the Deep 7 complex and ‘ōpakapaka only. Model-based estimates were used for the base-case model and fit as a relative index (i.e., estimating a catchability coefficient). Design-based estimates were included as a sensitivity and fit as absolute biomass values following the 2018 and 2021 assessments. ‘Ōpakapaka model-based estimates were used for the ‘ōpakapaka -only model.

Fishing Year	Model-based biomass (million lb)	Model-based CV	Design-based biomass (million lb)	Design-based CV
Deep 7 Complex				
2017	7.173	0.17	12.910	0.23
2018	8.496	0.15	12.130	0.12
2019	5.000	0.17	7.505	0.15
2020	5.377	0.16	7.978	0.14
2021	5.556	0.14	7.648	0.13
2022	5.428	0.13	7.640	0.19
2023	4.804	0.12	4.832	0.11
‘Ōpakapaka				
2017	5.235	0.22	-	-
2018	5.889	0.20	-	-
2019	3.092	0.24	-	-
2020	3.817	0.21	-	-
2021	2.575	0.28	-	-
2022	3.282	0.18	-	-
2023	2.823	0.17	-	-

Table 12. Prior distributions and parameter estimates for the 2024 base case assessment model for the main Hawaiian Islands Deep 7 bottomfish stock complex. Parameters are intrinsic growth rate (r), carrying capacity (K), ratio of B_{MSY} to K (B_{MSY}/K), initial pro proportion of carrying capacity (ψ), catchability for commercial (q_{comm}) and survey (q_{surv}) CPUE series, observation error for commercial CPUE (τ_{comm}^2), process error (σ^2), and annual total catch (C). Derived quantities are maximum sustainable yield (MSY) for total and reported catch, harvest rate at MSY (H_{MSY}), biomass at MSY (B_{MSY}), and biomass at minimum stock size threshold (B_{MSST}). Biomass and MSY are reported in millions of pounds.

-	**Prior distributions**				**Parameter estimates**	
Parameter	**Distribution**	**Mean**	**CV**	**Bounds**	**Median**	**SD**
r	lognormal	0.096	66%	-	0.090	0.037
K	lognormal	9.2 million lb	100%	-	20.01 million lb	7.07 million lb
B_{MSY}/K	lognormal	0.329	20%	-	0.328	0.035
m	lognormal	0.806	20%	-	0.800	0.163
ψ	lognormal	0.75	50%	-	0.878	0.183
q_{comm}	uniform	-	-	$[10^{-10},10^{10}]$	7.289	2.89
q_{surv}	uniform	-	-	$[10^{-10},10^{10}]$	0.497	0.200
τ_{comm}^2	Inverse gamma	0.083*	-	-	0.012	0.003
σ^2	Inverse gamma	0.001*	-	-	0.005	0.002
C	lognormal	C	13%	-	-	-
MSY (total catch)	-	-	-	-	0.709 million lb	0.207 million lb
H_{MSY}	-	-	-	-	0.112	0.051
B_{MSY}	-	-	-	-	6.539 million lb	2.476 million lb
B_{MSST}	-	-	-	-	5.67 million lb	2.15 million lb

*Value is mode rather than mean

Table 13. Summary of sensitivity scenarios evaluated for the Deep 7 bottomfish surplus production model as described in detail in section 2.9 Sensitivity Analyses. Symbols are for intrinsic growth rate (*r*), carrying capacity (*K*), initial proportion of carrying capacity (*K*), initial proportion of carrying capacity (ψ), biomass that produces maximum sustainable yield (B_{MSY}), and shape parameter (m).

Value	Number of scenarios	Type of change	Description
r	4	Distribution mean	Prior mean adjusted by ± 25% and 50%
K	4	Distribution mean	Prior mean adjusted by ± 25% and +50%
ψ	4	Distribution mean	Prior mean adjusted by ± 25% and 50%
B_{MSY}/K	4	Distribution mean	Prior mean adjusted by ± 25% and 50%
‘Ōpakapaka priors	1	Priors (r, B_{MSY}/K)	Use priors determined from ‘ōpakapaka life history only
Total catch variation	1	Data	Exclude variation around catch
HMRFS annual catch	1	Data	Use annual HMRFS estimates and CVs for unreported catch 2004–2023
Alternative survey CPUE	1	Data	Fit to alternative index produced by VAST
Absolute survey biomass	1	Model and data	Fit to survey biomass absolute abundance (i.e., catchability = effective radius)
Schaefer	1	Model	Fit model as Schaefer (i.e., B_{MSY}/K = 0.5; m = 2.0)
Jitter	10	Initial conditions	Evaluate sensitivity to initial parameter values used in MCMC

Table 14. MCMC convergence diagnostics for model parameter estimates.

Parameters	Gelman and Rubin	Geweke	HW stationarity	HW half-width	Lag1 auto-correlation	Lag5 auto-correlation
B_{MSY}	1.06	0.514	0.514	passed	-	-
H_{CR}	1.01	0.234	0.234	passed	-	-
MSY	1.03	0.574	0.574	passed	-	-
K	1.07	0.702	0.167	passed	-0.014	-0.003
R	1.01	0.553	0.715	passed	-0.001	-0.009
Ψ	1.01	0.927	0.321	passed	0.007	0.001
m	1.00	0.341	0.797	passed	0.010	0.002
q_1	1.03	0.447	0.111	passed	-0.006	-0.008
q_2	1.03	0.323	0.102	passed	-0.006	-0.005
σ^2	1.00	0.824	0.768	passed	-0.006	-0.007
τ^2	1.00	0.693	0.497	passed	0.007	0.004

Table 15. Estimates of mean exploitable biomass (*B*) in million lb, median relative exploitable biomass (B/B_{MSST}), probability of being overfished (B/B_{MSY}<0.867), median harvest rate (*H*), relative median harvest rate (H/H_{CR}), and probability of overfishing (H/H_{CR}>1).

Year	*B*	B/B_{MSST}	Probability of being overfished	*H*	H/H_{CR}	Probability of overfishing
1949	17.07	3.10	0.00	0.04	0.35	0.01
1950	16.95	3.07	0.00	0.03	0.31	0.00
1951	17.56	3.20	0.00	0.04	0.32	0.00
1952	18.37	3.34	0.00	0.03	0.28	0.00
1953	18.68	3.40	0.00	0.03	0.24	0.00
1954	18.40	3.35	0.00	0.03	0.26	0.00
1955	17.57	3.20	0.00	0.02	0.22	0.00
1956	16.00	2.91	0.00	0.03	0.30	0.00
1957	14.96	2.72	0.00	0.04	0.39	0.02
1958	13.88	2.51	0.00	0.03	0.28	0.00
1959	14.09	2.56	0.00	0.03	0.26	0.00
1960	14.88	2.70	0.00	0.02	0.20	0.00
1961	15.88	2.88	0.00	0.02	0.15	0.00
1962	16.67	3.03	0.00	0.02	0.19	0.00
1963	16.55	3.02	0.00	0.03	0.23	0.00
1964	16.63	3.02	0.00	0.02	0.22	0.00
1965	16.89	3.07	0.00	0.03	0.24	0.00
1966	16.01	2.92	0.00	0.02	0.20	0.00
1967	15.76	2.87	0.00	0.03	0.29	0.00
1968	15.45	2.81	0.00	0.03	0.24	0.00
1969	15.35	2.79	0.00	0.02	0.22	0.00
1970	15.00	2.72	0.00	0.02	0.20	0.00
1971	15.31	2.78	0.00	0.02	0.16	0.00
1972	16.05	2.92	0.00	0.03	0.28	0.00
1973	15.63	2.84	0.00	0.02	0.21	0.00
1974	15.71	2.86	0.00	0.03	0.29	0.00
1975	15.14	2.75	0.00	0.03	0.26	0.00
1976	14.43	2.62	0.00	0.03	0.31	0.00

Year	*B*	*B*/B_{MSST}	Probability of being overfished	*H*	*H*/H_{CR}	Probability of overfishing
1977	13.69	2.49	0.00	0.04	0.34	0.01
1978	13.81	2.51	0.00	0.04	0.40	0.02
1979	13.53	2.46	0.00	0.04	0.36	0.01
1980	12.91	2.34	0.00	0.04	0.35	0.01
1981	12.35	2.24	0.00	0.05	0.47	0.04
1982	11.83	2.15	0.01	0.06	0.50	0.05
1983	11.07	2.01	0.01	0.07	0.67	0.17
1984	10.61	1.93	0.01	0.06	0.56	0.09
1985	11.23	2.04	0.01	0.08	0.72	0.22
1986	11.55	2.10	0.01	0.08	0.72	0.22
1987	12.12	2.20	0.00	0.09	0.82	0.31
1988	12.43	2.26	0.00	0.09	0.82	0.32
1989	12.27	2.23	0.00	0.09	0.84	0.33
1990	11.65	2.12	0.01	0.08	0.68	0.18
1991	11.21	2.04	0.01	0.06	0.50	0.05
1992	10.91	1.98	0.01	0.06	0.57	0.09
1993	10.68	1.94	0.01	0.05	0.42	0.02
1994	10.68	1.95	0.01	0.06	0.52	0.06
1995	10.55	1.92	0.01	0.07	0.59	0.10
1996	10.25	1.86	0.02	0.05	0.47	0.04
1997	10.34	1.88	0.01	0.06	0.52	0.06
1998	10.22	1.86	0.02	0.06	0.51	0.06
1999	10.28	1.87	0.02	0.04	0.37	0.01
2000	10.60	1.93	0.01	0.06	0.53	0.06
2001	10.19	1.86	0.02	0.05	0.45	0.03
2002	9.80	1.78	0.02	0.04	0.39	0.02
2003	9.65	1.76	0.02	0.05	0.46	0.04
2004	9.44	1.72	0.03	0.05	0.41	0.02
2005	9.66	1.76	0.02	0.05	0.47	0.04
2006	9.77	1.78	0.02	0.04	0.36	0.01

Year	B	B/B_{MSST}	Probability of being overfished	H	H/H_{CR}	Probability of overfishing
2007	9.95	1.81	0.02	0.05	0.41	0.02
2008	10.06	1.83	0.02	0.04	0.36	0.01
2009	9.90	1.80	0.02	0.05	0.48	0.04
2010	9.75	1.77	0.02	0.05	0.41	0.02
2011	10.01	1.82	0.02	0.06	0.51	0.06
2012	10.16	1.85	0.02	0.05	0.41	0.02
2013	10.67	1.94	0.01	0.05	0.41	0.02
2014	11.25	2.05	0.01	0.06	0.52	0.06
2015	11.82	2.15	0.01	0.05	0.49	0.05
2016	11.91	2.17	0.01	0.05	0.42	0.02
2017	12.31	2.24	0.00	0.04	0.37	0.01
2018	12.41	2.26	0.00	0.04	0.36	0.01
2019	11.67	2.12	0.01	0.03	0.29	0.00
2020	11.34	2.06	0.01	0.03	0.28	0.00
2021	11.20	2.04	0.01	0.03	0.27	0.00
2022	11.23	2.05	0.01	0.04	0.32	0.01
2023	11.22	2.04	0.01	0.04	0.34	0.01

Table 16. Projection results for mean probability of overfishing (H/H_{CR}>1) and corresponding annual reported catch where the probability of overfishing is reached. The mean probability the stock is overfished (B/B_{MSY}<0.867), median harvest rate, and mean biomass are the values in each fishing year (July 1–June 30) that correspond to the specified reported catch values.

Probability of overfishing (H/H_{CR}>1)	**0**	**0.01**	**0.05**	**0.1**	**0.15**	**0.2**	**0.25**	**0.3**	**0.35**	**0.4**	**0.45**	**0.5**
Reported catch (millions of lbs), constant from 2025 through terminal year												
2025	0.00	0.21	0.29	0.34	0.39	0.43	0.46	0.49	0.52	0.55	0.58	0.61
2026	0.00	0.20	0.29	0.33	0.38	0.41	0.44	0.46	0.49	0.52	0.55	0.57
2027	0.00	0.20	0.28	0.33	0.36	0.39	0.42	0.45	0.47	0.50	0.52	0.55
2028	0.00	0.20	0.28	0.32	0.35	0.38	0.41	0.43	0.45	0.48	0.50	0.53
2029	0.00	0.20	0.27	0.32	0.34	0.37	0.40	0.42	0.44	0.46	0.48	0.51
Probability stock is overfished (B/B_{MSY}<0.867)												
2025	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
2026	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.02
2027	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.02	0.02	0.02	0.03	0.03
2028	0.00	0.01	0.01	0.02	0.02	0.02	0.02	0.03	0.03	0.04	0.04	0.05
2029	0.00	0.01	0.01	0.02	0.02	0.03	0.03	0.04	0.04	0.05	0.06	0.07
Biomass (millions of lb)												
2025	11.89	11.89	11.89	11.89	11.89	11.89	11.89	11.89	11.89	11.89	11.89	11.89
2026	12.48	12.03	11.84	11.73	11.63	11.56	11.50	11.44	11.37	11.30	11.24	11.18
2027	13.04	12.17	11.82	11.61	11.46	11.33	11.21	11.08	10.97	10.87	10.76	10.66
2028	13.57	12.34	11.82	11.55	11.33	11.14	10.99	10.84	10.69	10.53	10.37	10.21
2029	14.04	12.46	11.83	11.48	11.24	10.99	10.79	10.59	10.43	10.22	10.06	9.86

Probability of overfishing (H/H_{CR}>1)	**0**	**0.01**	**0.05**	**0.1**	**0.15**	**0.2**	**0.25**	**0.3**	**0.35**	**0.4**	**0.45**	**0.5**
						Harvest rate						
2025	0.00	0.33	0.47	0.56	0.63	0.69	0.75	0.79	0.85	0.90	0.95	1.00
2026	0.00	0.32	0.47	0.55	0.63	0.68	0.73	0.78	0.84	0.89	0.95	0.99
2027	0.00	0.32	0.46	0.55	0.61	0.67	0.73	0.79	0.84	0.89	0.94	0.99
2028	0.00	0.31	0.45	0.54	0.60	0.66	0.72	0.77	0.82	0.88	0.94	1.00
2029	0.00	0.30	0.44	0.53	0.59	0.66	0.71	0.77	0.82	0.88	0.93	1.00

Table 17. Probability of overfishing in terminal year (H/H_{CR}>1) and projected reported catch (millions of lb) by fishing year (July 1–June 30). Catch values for a given probability of overfishing in a given year were applied in all previous years (i.e., 2025 to terminal year).

P(Overfishing)	2025	2026	2027	2028	2029
0	0.000	0.000	0.000	0.000	0.000
0.01	0.206	0.201	0.201	0.196	0.196
0.02	0.234	0.234	0.230	0.230	0.225
0.03	0.258	0.254	0.249	0.249	0.244
0.04	0.278	0.273	0.268	0.263	0.258
0.05	0.292	0.287	0.282	0.278	0.273
0.06	0.306	0.301	0.292	0.287	0.282
0.07	0.316	0.311	0.301	0.297	0.292
0.08	0.325	0.321	0.311	0.306	0.301
0.09	0.340	0.330	0.321	0.311	0.306
0.1	0.344	0.335	0.330	0.321	0.316
0.11	0.354	0.344	0.335	0.330	0.321
0.12	0.364	0.354	0.344	0.335	0.330
0.13	0.373	0.364	0.349	0.340	0.335
0.14	0.378	0.368	0.359	0.349	0.340
0.15	0.388	0.378	0.364	0.354	0.344
0.16	0.397	0.383	0.368	0.359	0.354
0.17	0.402	0.388	0.378	0.364	0.359
0.18	0.411	0.397	0.383	0.373	0.364
0.19	0.416	0.402	0.388	0.378	0.368
0.2	0.426	0.407	0.392	0.383	0.373
0.21	0.431	0.411	0.402	0.388	0.378
0.22	0.435	0.421	0.407	0.392	0.383
0.23	0.445	0.426	0.411	0.397	0.388
0.24	0.450	0.431	0.416	0.402	0.392
0.25	0.459	0.435	0.421	0.407	0.397
0.26	0.464	0.445	0.426	0.411	0.402
0.27	0.469	0.450	0.431	0.416	0.407
0.28	0.478	0.455	0.435	0.421	0.411
0.29	0.483	0.459	0.440	0.426	0.416
0.3	0.488	0.464	0.450	0.431	0.421

P(Overfishing)	2025	2026	2027	2028	2029
0.31	0.498	0.469	0.455	0.435	0.426
0.32	0.502	0.474	0.459	0.440	0.431
0.33	0.507	0.483	0.464	0.445	0.435
0.34	0.512	0.488	0.469	0.450	0.435
0.35	0.522	0.493	0.474	0.455	0.440
0.36	0.526	0.498	0.478	0.459	0.445
0.37	0.531	0.502	0.483	0.464	0.450
0.38	0.536	0.507	0.488	0.469	0.455
0.39	0.545	0.517	0.493	0.474	0.459
0.4	0.550	0.522	0.498	0.478	0.464
0.41	0.555	0.526	0.502	0.483	0.469
0.42	0.565	0.531	0.507	0.488	0.474
0.43	0.569	0.536	0.512	0.493	0.474
0.44	0.574	0.541	0.517	0.498	0.478
0.45	0.584	0.550	0.522	0.502	0.483
0.46	0.589	0.555	0.526	0.507	0.488
0.47	0.593	0.560	0.531	0.512	0.493
0.48	0.598	0.565	0.536	0.517	0.498
0.49	0.608	0.569	0.541	0.522	0.502
0.5	0.612	0.574	0.545	0.526	0.507

Table 18. Sensitivity of production model results to 25% and 50% increases and decreases in prior mean for intrinsic growth rate (*r*), carrying capacity (*K*), initial proportion of carrying capacity (ψ), and *BMSY/K*, and the use of ʻōpakapaka-specific prior values, treatments of the fishery-independent index of relative abundance, treatments of catch variability and noncommercial catch, and using a Schaefer model. Results are expressed as percent change relative to base case model for *r, K, m*, ψ, maximum sustainable yield (MSY), biomass as MSY (B_{MSY}), biomass at minimum stock size threshold (B_{MSST}), harvest rate at MSY (H_{MSY}), terminal year exploitable biomass (B_{2023}) and harvest rate (H_{2023}), exploitable biomass in 2023 relative to B_{MSY} and B_{MSST} (B_{2023}/B_{MSY}, B_{2023}/B_{MSST}), harvest rate in 2023 relative to H_{CR} (H_{2023}/H_{CR}) and the probability of overfishing in 2023 (*poflH* 2023) and probability of being overfished in 2023 (*poflB* 2023).

Scenario	*r*	*K*	B_{MSY}/ *K*	ψ	MSY	B_{MSY}	B_{MSST}	H_{MSY}	B_{2023}	H_{2023}	B_{2023}/ B_{MSY}	B_{2023}/ B_{MSST}	H_{2023}/ H_{CR}	*poflH* 2023	*poflB* 2023
r = 0.072	-6.98	3.72	-0.3	-0.78	-2.68	3.41	3.41	-6.25	4.59	-4.39	-1.07	-1.09	4.39	-27.34	-63.67
r = 0.12	9.7	-6.08	0.3	2.58	4.65	-5.83	-5.83	9.82	-2.42	2.48	2.43	2.4	-6.43	-78.78	-86.51
r = 0.048	-29.87	25.32	-0.61	-7.18	-10.44	24.7	24.7	-28.57	16.79	-14.38	-7.23	-7.24	22.51	137.77	-9.69
r = 0.144	17.55	-10.88	0.3	3.14	6.21	-10.25	-10.25	16.96	-7.03	7.56	3.11	3.08	-9.06	-83.45	-83.39
K = 6.85	6.03	-9.75	-0.3	-3.07	-2.4	-9.47	-9.47	6.25	-11.76	13.33	-3.22	-3.25	5.85	-24.82	-45.67
K = 11.42	-0.08	0.51	0	1.45	1.27	0.35	0.35	0	1.96	-1.92	1.02	0.99	-2.05	-17.27	-34.26
K = 4.57	14.62	-14.34	0	-1.1	-0.85	-14.18	-14.18	15.18	-13.19	15.19	-1.02	-1.03	2.05	-47.48	-61.59
K = 13.71	-7.23	11.95	0.3	3.55	3.39	12.16	12.16	-7.14	15.56	-13.46	2.6	2.58	-5.85	-54.68	-78.89
ψ = 0.56	-1.67	-0.17	0	-3.49	-0.85	-0.66	-0.66	-1.79	-0.68	0.68	-2.03	-2.06	2.92	-10.79	-22.49
ψ = 0.94	-0.01	1.38	0	3.76	2.12	1.76	1.76	0	4.85	-4.62	2.26	2.27	-4.39	-25.18	-45.67
ψ = 0.38	-10.18	3.32	0	-13.05	-6.21	3.53	3.53	-9.82	-5.18	5.46	-9.49	-9.51	18.42	148.2	148.44
ψ = 1.125	2.45	0.49	0	6.55	3.95	0.86	0.86	1.79	5.81	-5.49	3.78	3.77	-7.6	-69.42	-94.46
B_{MSY}/*K* = 0.28	1.73	-7.02	-14.63	0.83	9.87	-20.35	-20.35	35.71	-4.91	5.16	17.79	17.77	-23.1	-96.4	-92.39
B_{MSY}/*K* = 0.37	7.31	-1.84	11.89	2.01	-2.4	10.32	10.32	-13.39	1.95	-1.91	-9.15	-9.15	12.87	-51.8	-38.41
B_{MSY}/*K* = 0.22	0.88	-9.69	-33.54	1.51	22.14	-40.14	-40.14	101.79	-6.69	7.17	52.23	52.22	-46.78	-100	-100
B_{MSY}/*K* = 0.40	8.47	2.04	22.26	1.93	-7.05	24.65	24.65	-26.79	5.14	-4.89	-17	-16.99	29.82	27.7	45.33

Scenario	r	K	B_{MSY}/K	ψ	MSY	B_{MSY}	B_{MSST}	H_{MSY}	B_{2023}	H_{2023}	B_{2023}/B_{MSY}	B_{2023}/B_{MSST}	H_{2023}/H_{CR}	*pofIH* 2023	*pofIB* 2023
ʻŌpakapaka priors	-4.62	16.09	-3.35	6.6	16.5	12.66	12.66	1.79	26.56	-20.99	10.16	10.43	-23.1	-81.65	-72.32
No catch variability	1.33	-0.52	0	2.35	1.97	-0.2	-0.2	0.89	2.94	-2.86	2.15	2.13	-4.09	-33.45	-36.68
HMRFS annual estimates	0.16	1.58	0	1.38	2.12	1.51	1.51	0	3.22	-10.8	1.02	1.02	-10.8	-82	-85.8
BFISH absolute	-6.9	9.14	0	2.38	2.26	9.67	9.67	-7.14	2.85	-2.77	-6.55	-6.53	4.97	88.13	42.21
BFISH PLDG	2.05	-1.02	0	0.49	0.99	-0.8	-0.8	1.79	-1.05	1.06	-1.64	-1.62	1.17	194.24	177.85
Schaefer Model	31.73	2.64	52.44	2.47	-15.66	57.01	57.01	-47.32	6.06	-5.71	-32.52	-32.55	78.36	681.65	441.87

Table 19. Posterior median of select model parameters and derived quantities from the ‘ōpakapaka production model. The percent difference between these values and the Deep 7 bottom complex production model are also shown.

Parameters or quantities	**‘Ōpakapaka Model**	**Percent difference (%) from Deep 7 bottomfish complex**
K	10.27	-51.3
r	0.09	-0.52
m	0.77	-5.36
ψ	0.75	-14.62
q_1	11.17	44.78
q_2	0.90	69.31
σ^2	0.01	87.51
F_{MSY}	0.13	7.36
B_{MSY}	3.28	-52.78
MSY	0.39	-47.51
B_{MSY}/K	0.32	-3.34
B_{MSST}	2.32	-59.08

Figures

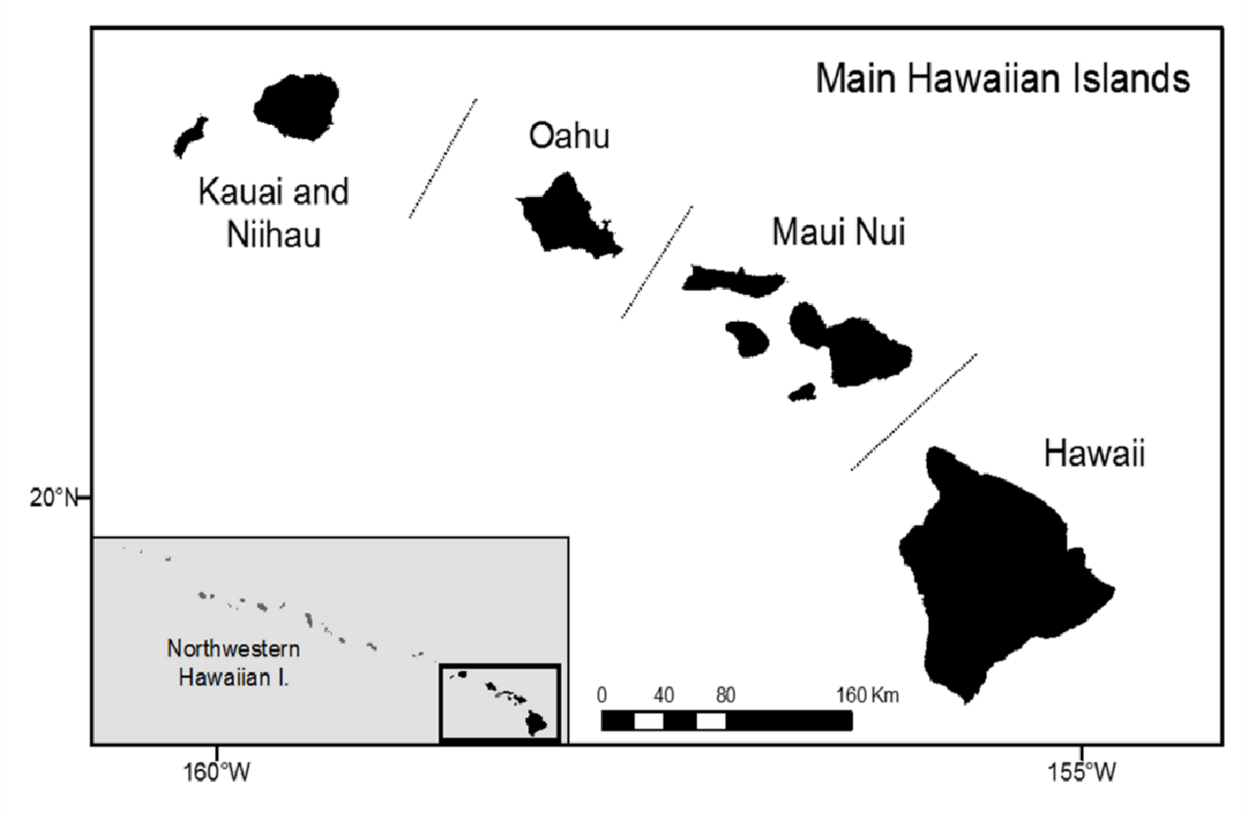


Figure 1. Main Hawaiian Islands and its four sub regions. The entire archipelago is visible in the inset, including the Northwestern Hawaiian Islands.

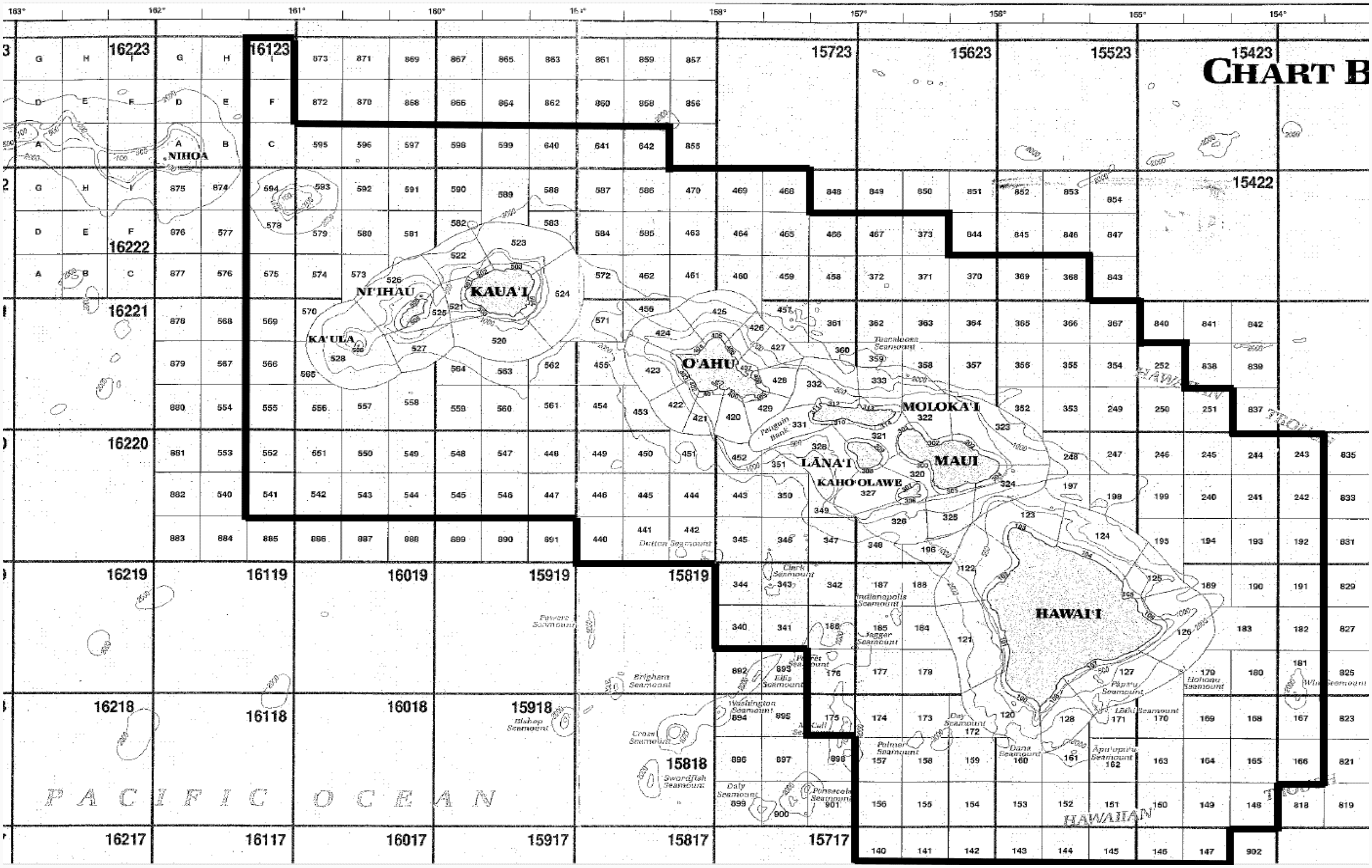
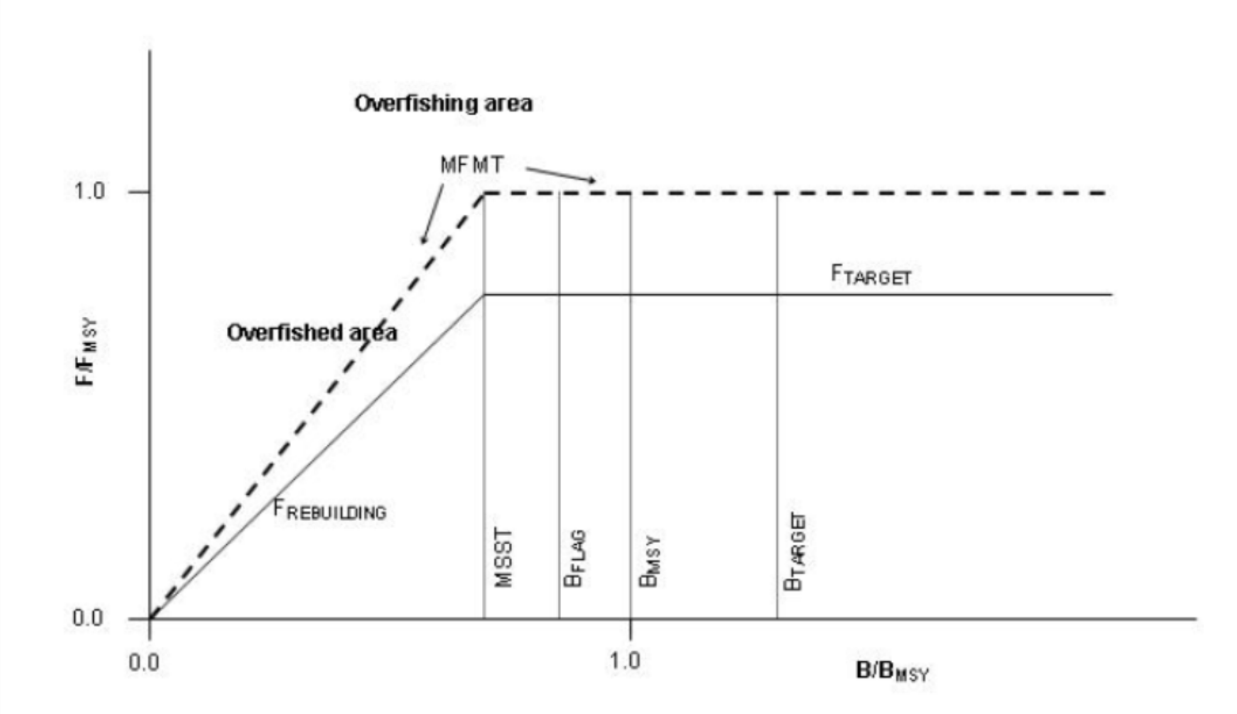


Figure 2. Boundary of the stock area in the main Hawaiian Islands for the 2024 Deep 7 benchmark stock assessment.



MFMT	MSST	B_{FLAG}
$F(B) = \frac{F_{MSY} B}{c\, B_{MSY}}$ for $B \leq c\, B_{MSY}$ $F(B) = F_{MSY}$ for $B > c\, B_{MSY}$	$c\, B_{MSY}$	B_{MSY}
where $c = \max(1\text{-}M, 0.5)$		

Figure 3. Harvest control rule for main Hawaiian Islands Deep 7 bottomfish (reproduced from WPRFMC 2009). *F* and F_{MSY} in the figure are equivalent to harvest rate (*H*) and harvest rate that produces maximum sustainable yield (MSY; H_{MSY}) in the 2024 benchmark assessment. The harvest control rule determines the threshold for overfishing (defined as H_{CR} in the 2024 assessments) as a function of H_{MSY}, biomass (*B*), the biomass that produces maximum sustainable yield (B_{MSY}), and one minus the rate of natural mortality (*M*; assumed to be 0.134).

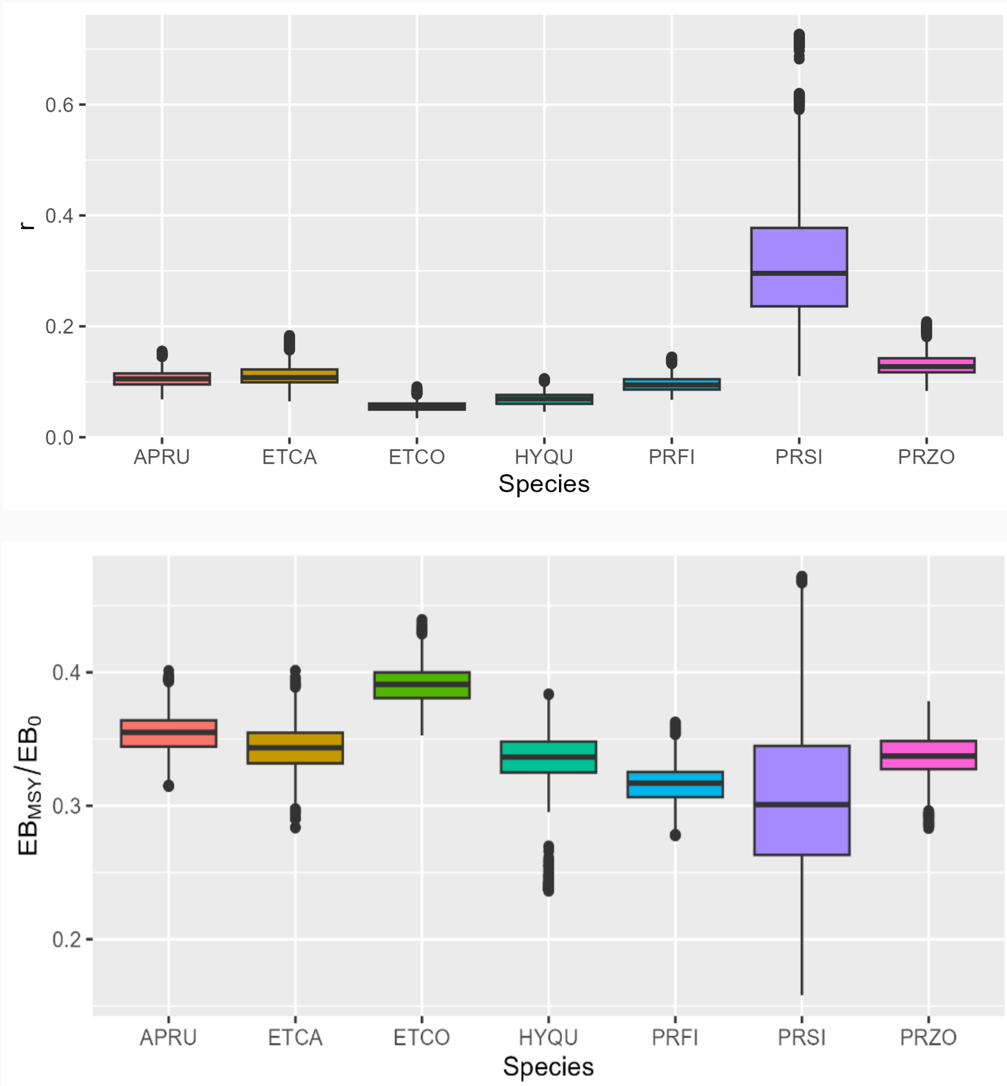


Figure 4. Species-specific estimates of intrinsic growth rate (*r*; top panel) and exploitable biomass at maximum sustainable yield relative to unfished exploitable biomass (EB_{MSY}/EB_0; bottom panel) estimated from the age structured equilibrium model.

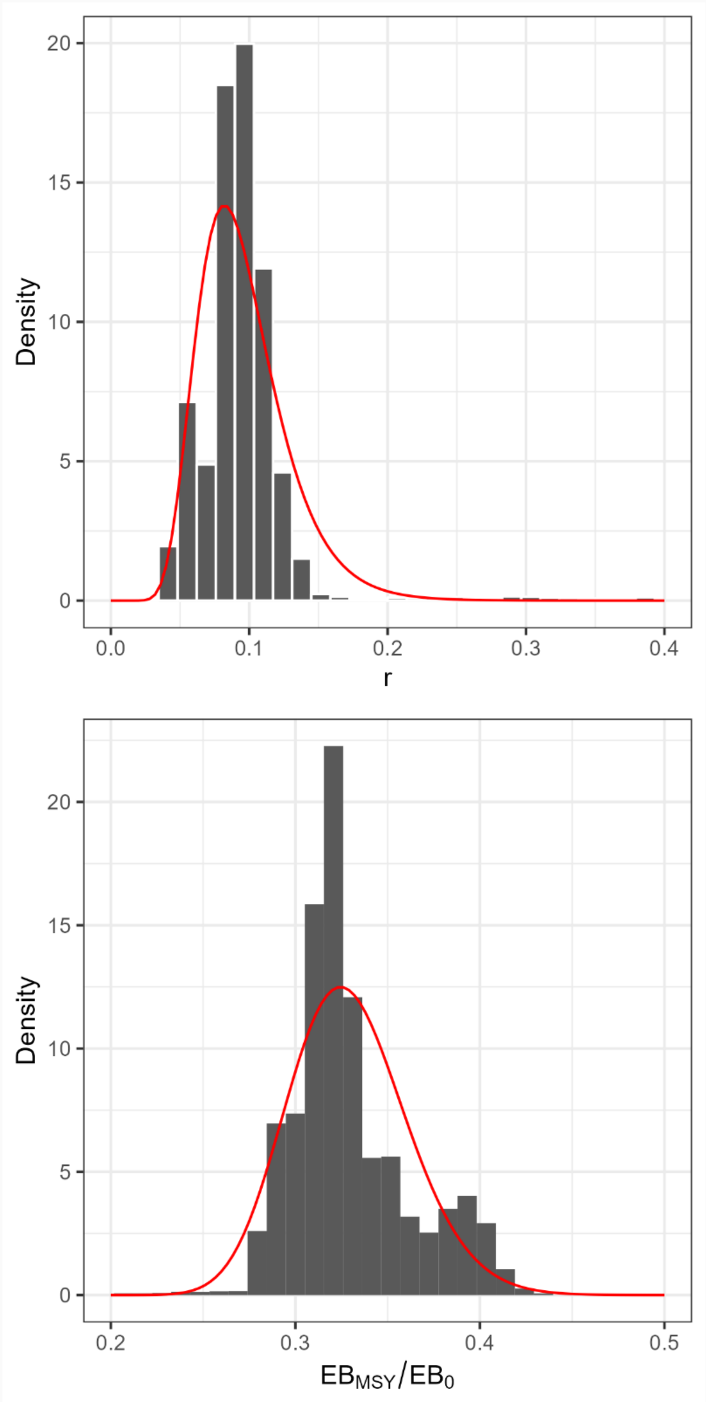


Figure 5. Probability distribution of the intrinsic growth rate (*r*; top panel) and exploited biomass ratio (EB_{MSY}/EB_0; bottom panel) for all species using the age-structured equilibrium model. The red line represents the best-fit, log-normal distribution.

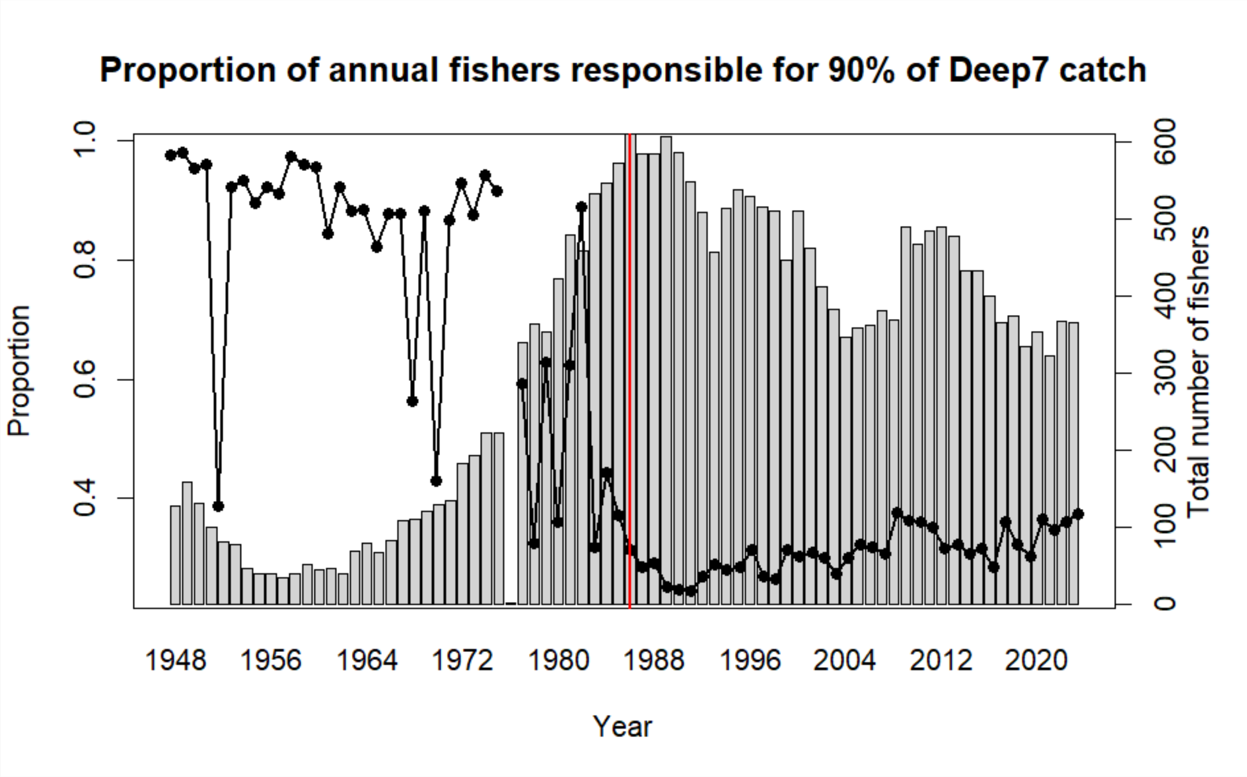


Figure 6. Proportion of annual fishers responsible for 90% of Deep 7 catch (points) and total number of fishers reporting Deep 7 catch (bars) from 1948–2023. The vertical red line delineates 1986, the first year with 100% complete fisher license information.

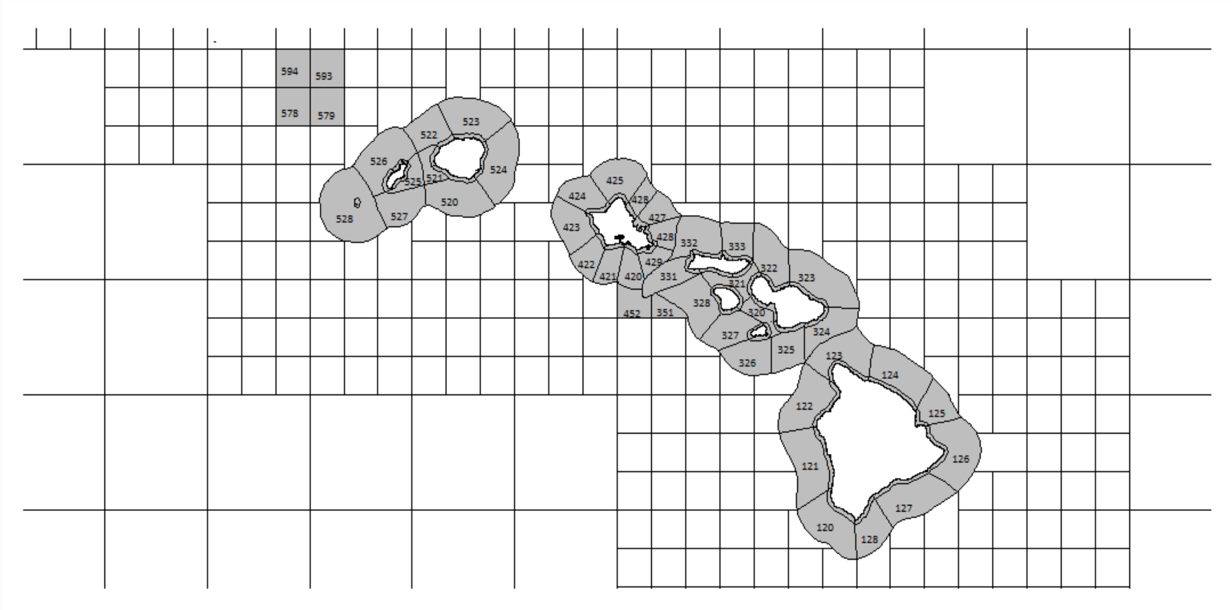


Figure 7. Areas retained (gray) for CPUE analysis based on having at least 15 years of catch observations during 1949–2023.

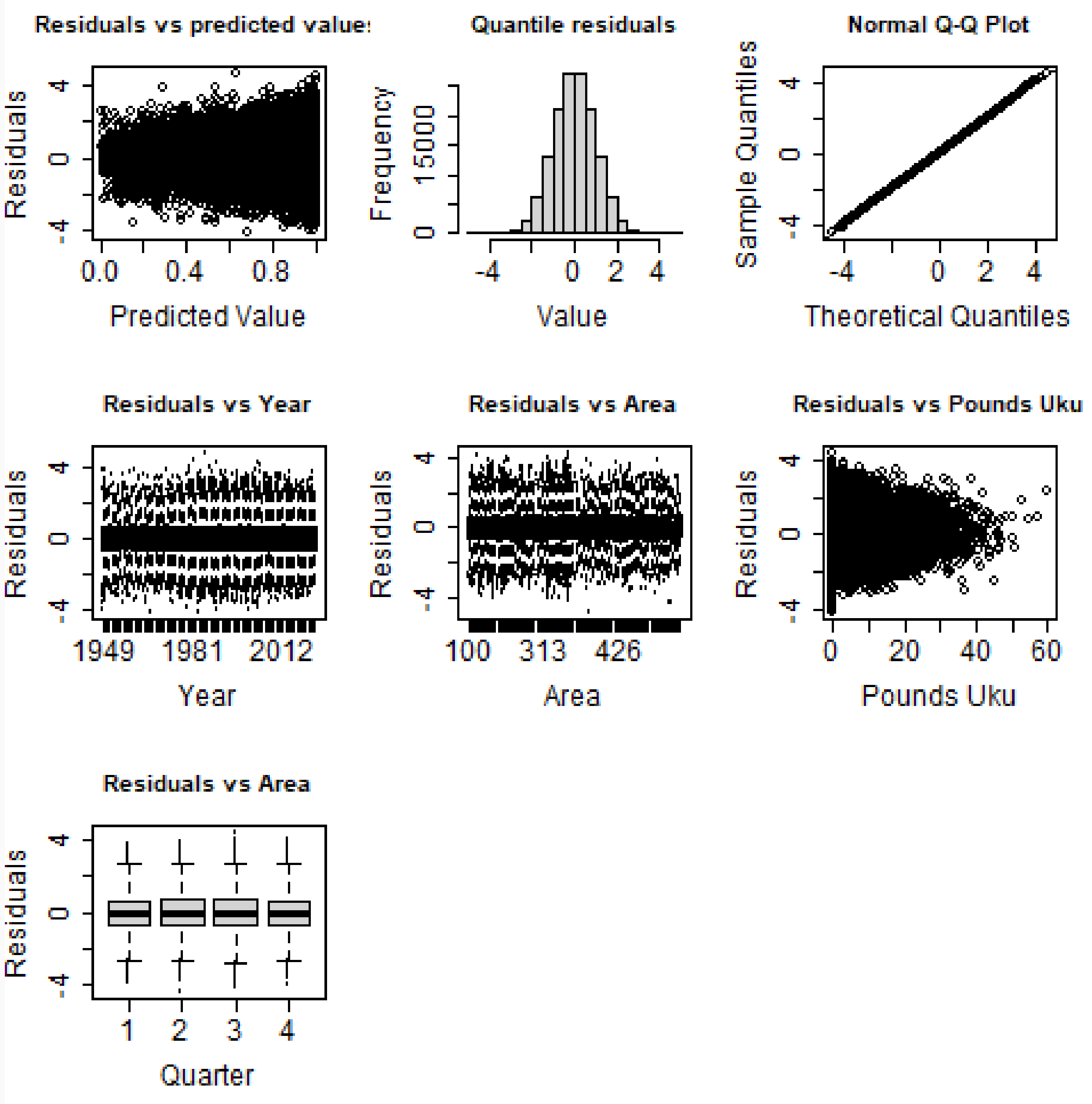


Figure 8. Diagnostic plots for the best-fit Bernoulli model.

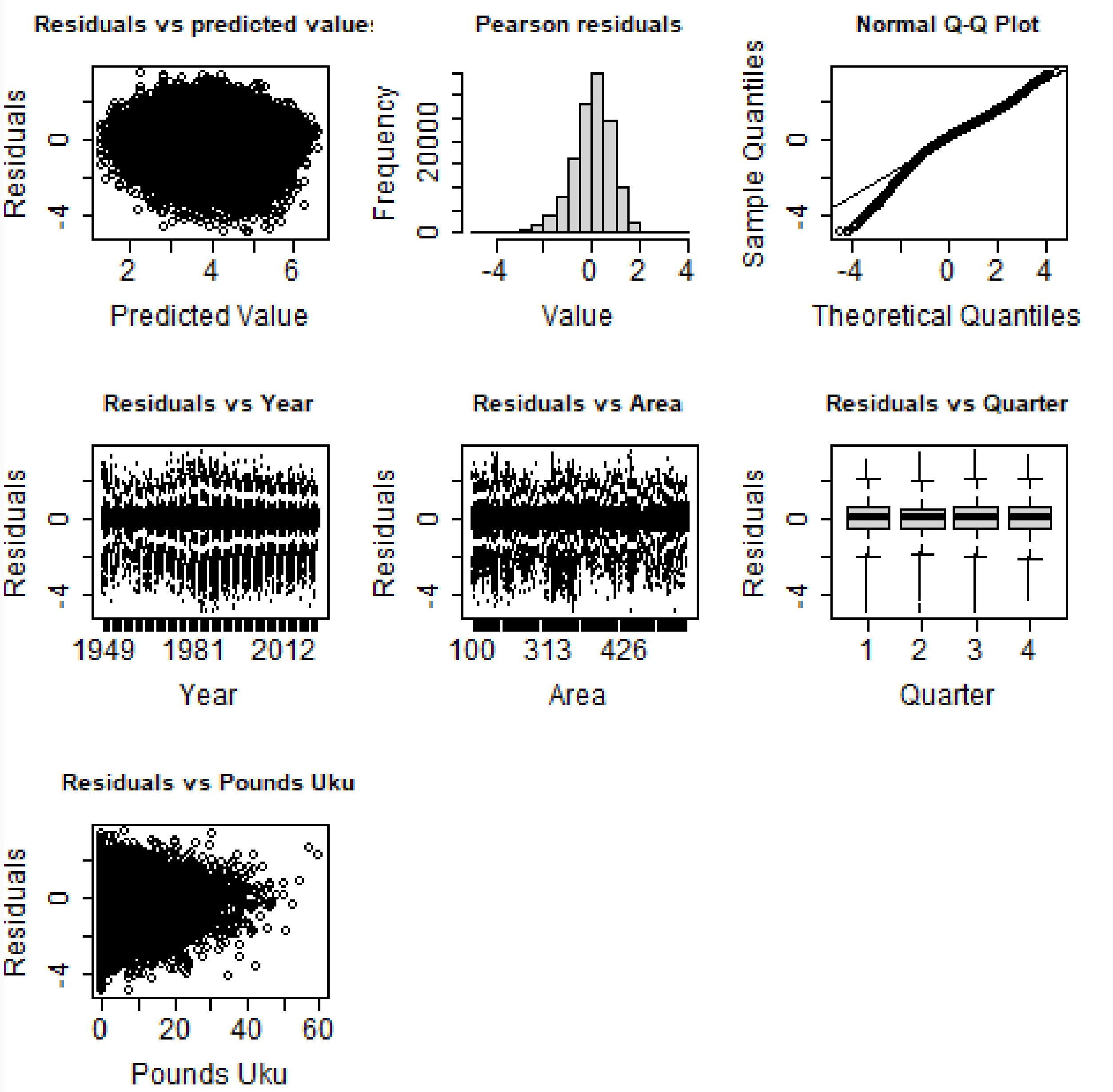


Figure 9. Diagnostic plots for the best-fit lognormal model.

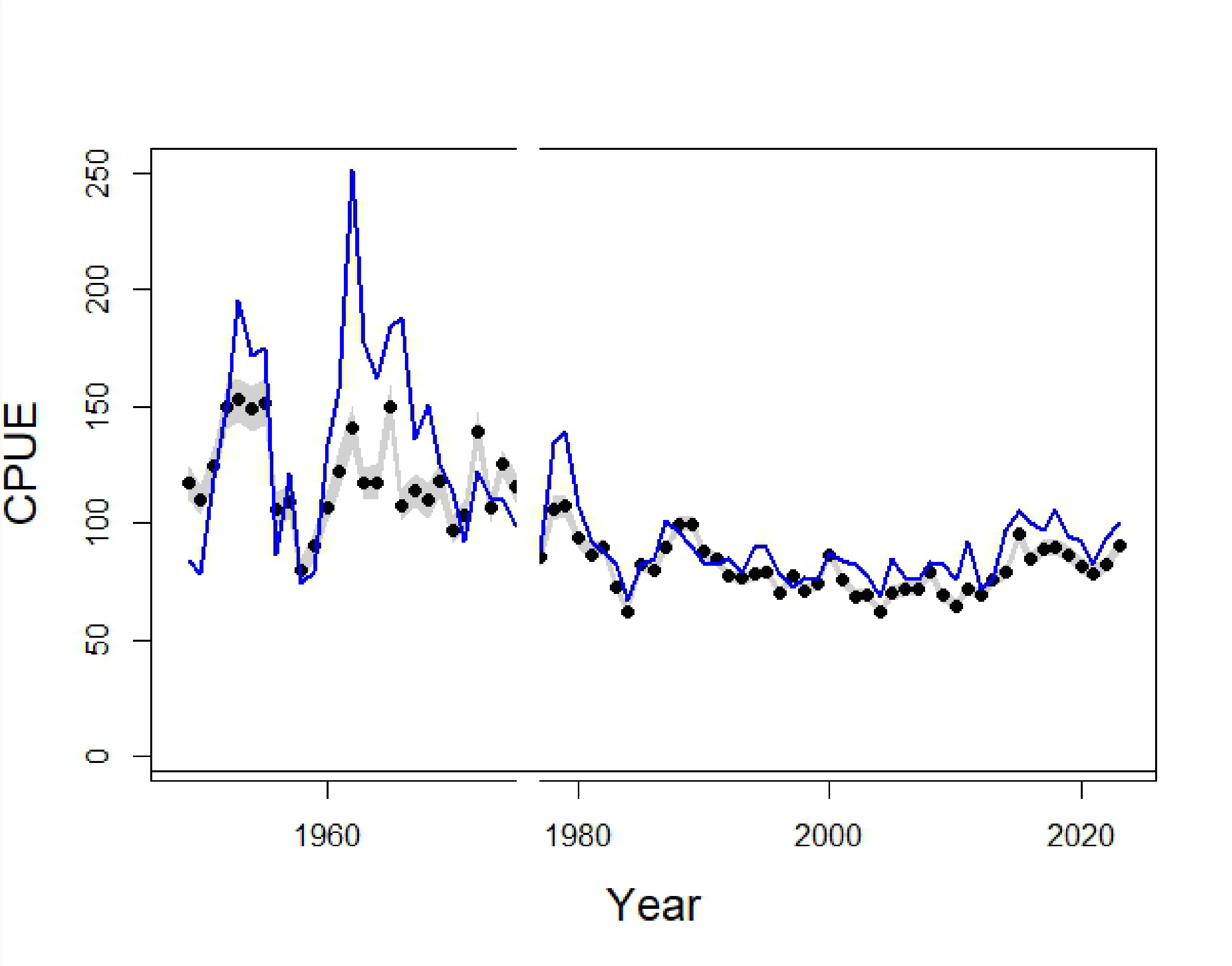


Figure 10. Standardized commercial CPUE ± 1 SE (points and gray shading) and mean nominal CPUE (blue line) for 1949–2023, excluding 1976.

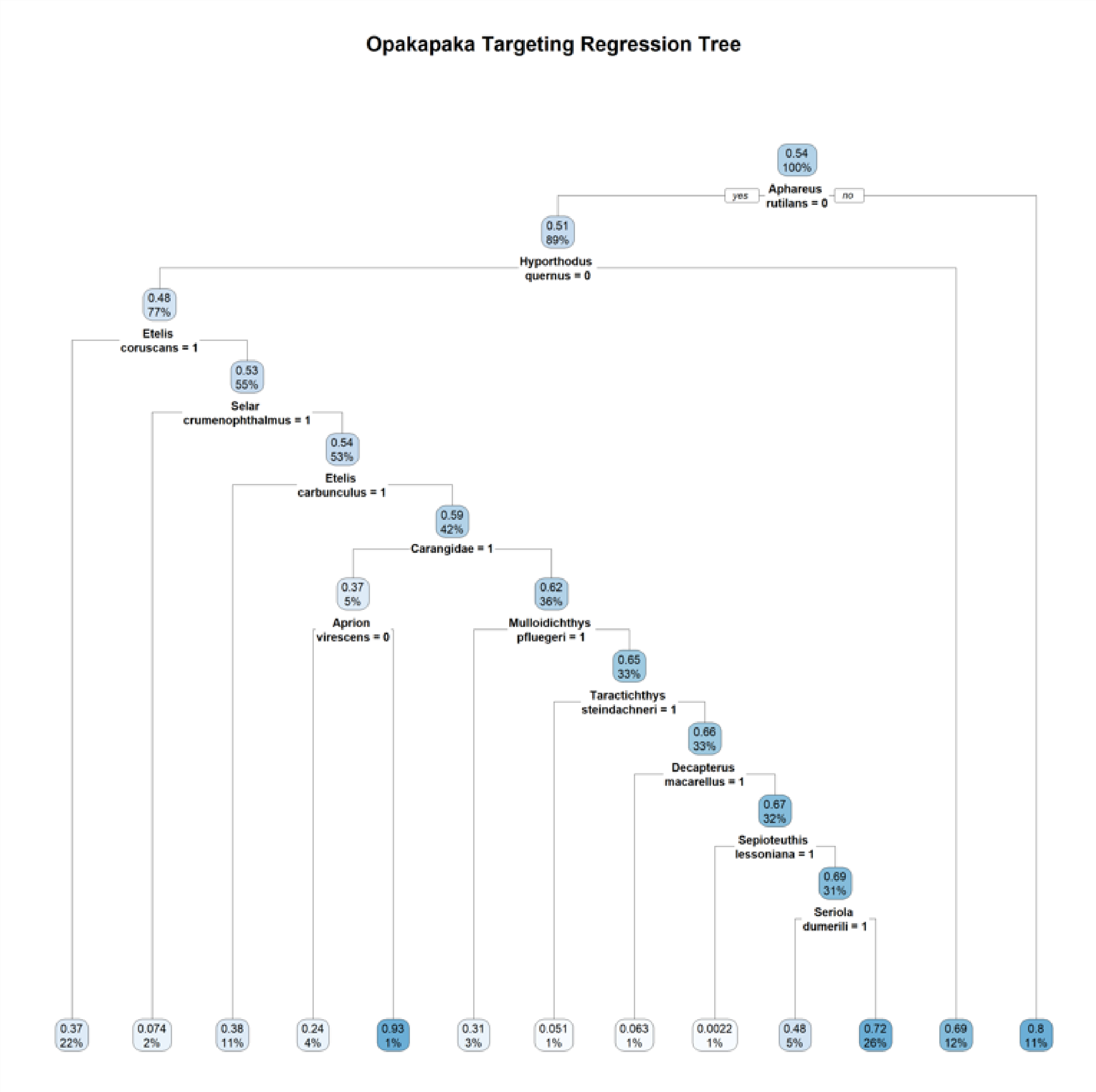


Figure 11. Regression tree for most influential species in predicting ‘ōpakapaka targeting.

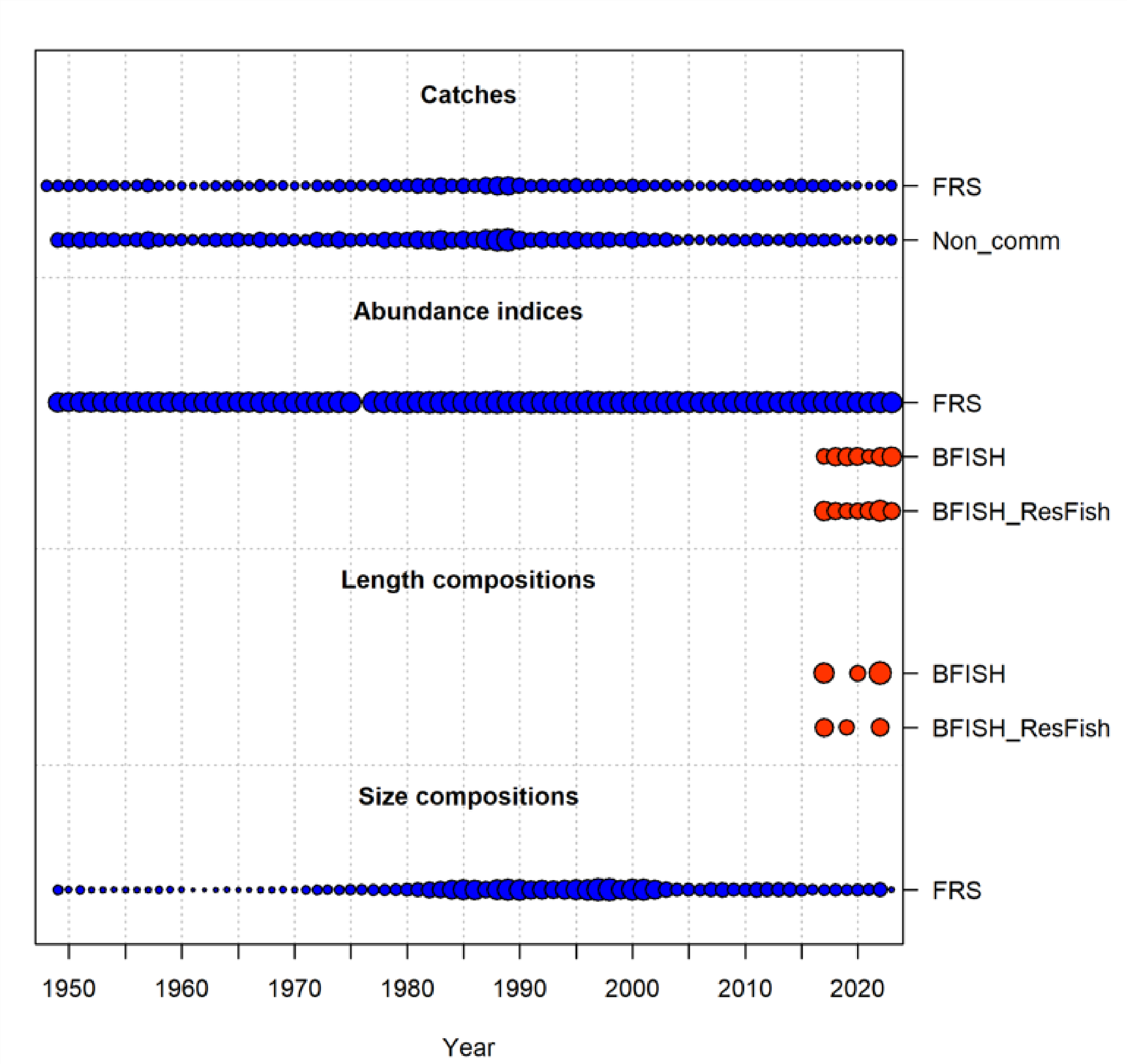


Figure 12. Data inputs for the single-species ‘ōpakapaka stock synthesis model. Inputs included catch from commercial (FRS) and non-commercial (Non_comm) fisheries; CPUE from the commercial (FRS) fishery and indices of relative abundance from the BFISH camera (BFISH) and research fishing (BFISH_ResFish) surveys; length composition from the BFISH camera (BFISH) and research fishing (BFISH_ResFish) surveys; and weight composition from the commercial (FRS) fishery. Dots represent years with the respective data and size of dots indicates the quantity of data for each year relative to the total time series of that data source.

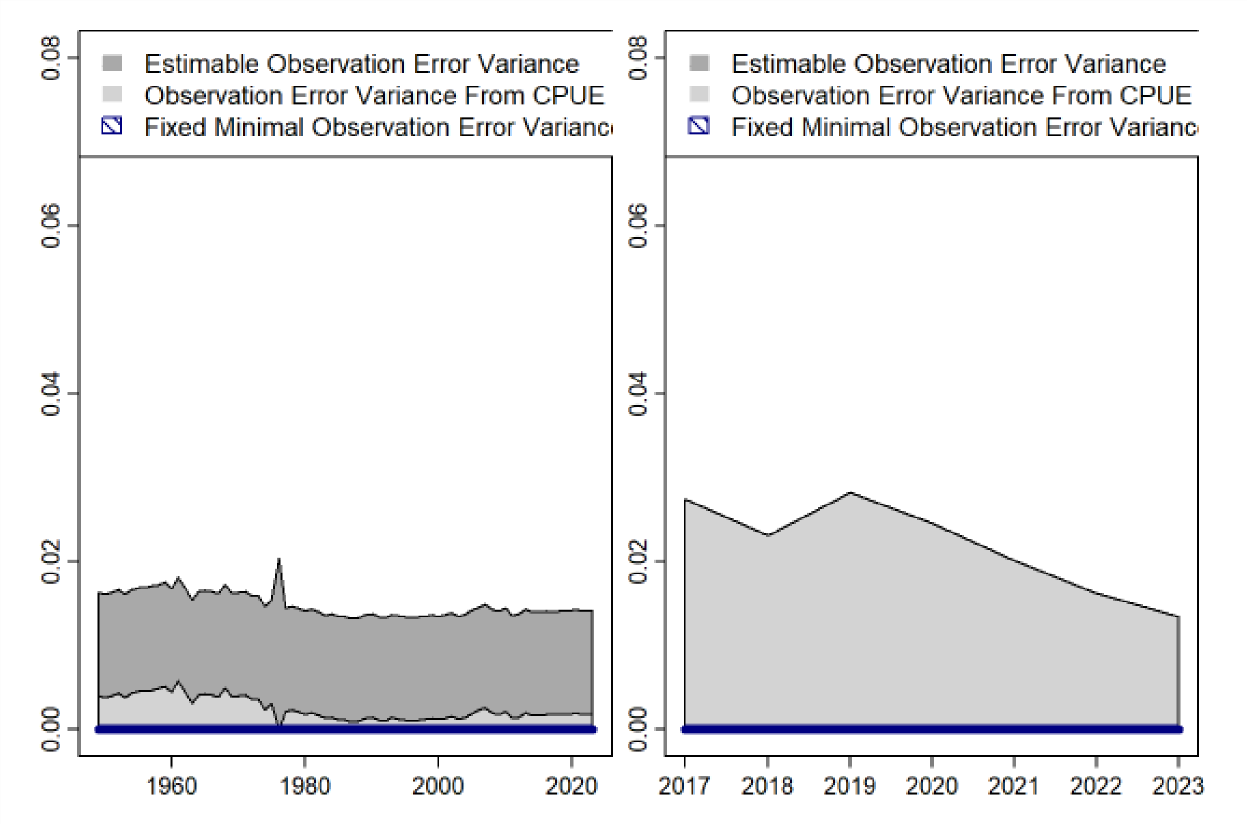


Figure 13. Total observation error variance for the FRS (left) and BFISH (right), partitioned into minimum observation error (set to 0), observation error from CPUE (light gray), and estimable observation error (dark gray).

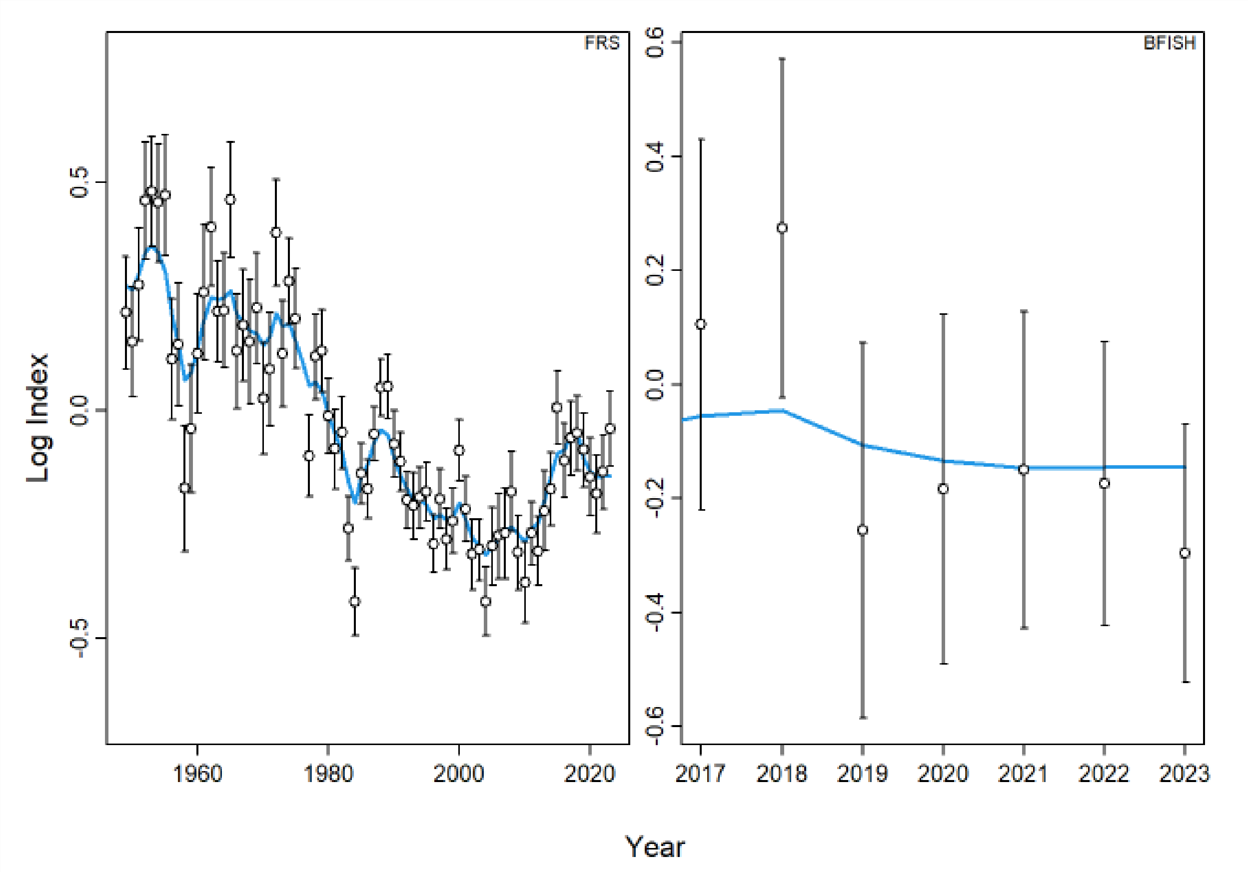


Figure 14. Fits of the FRS CPUE and BFISH indices of abundance on the log scale. Open circles are observations, vertical lines are the associated log standard error, and the blue line is the expected values from the model fit.

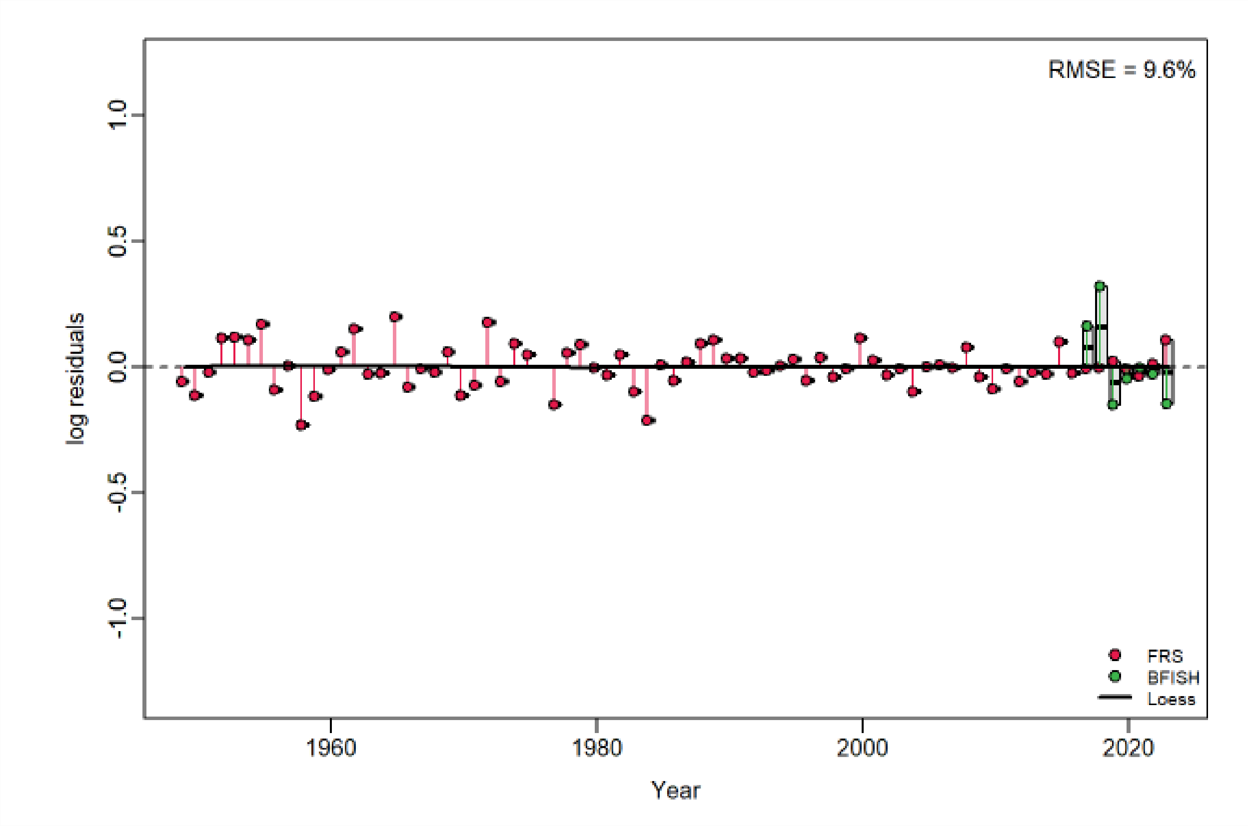


Figure 15. Residuals for CPUE (FRS) and BFISH indices of abundance and combined RMSE (9.6%).

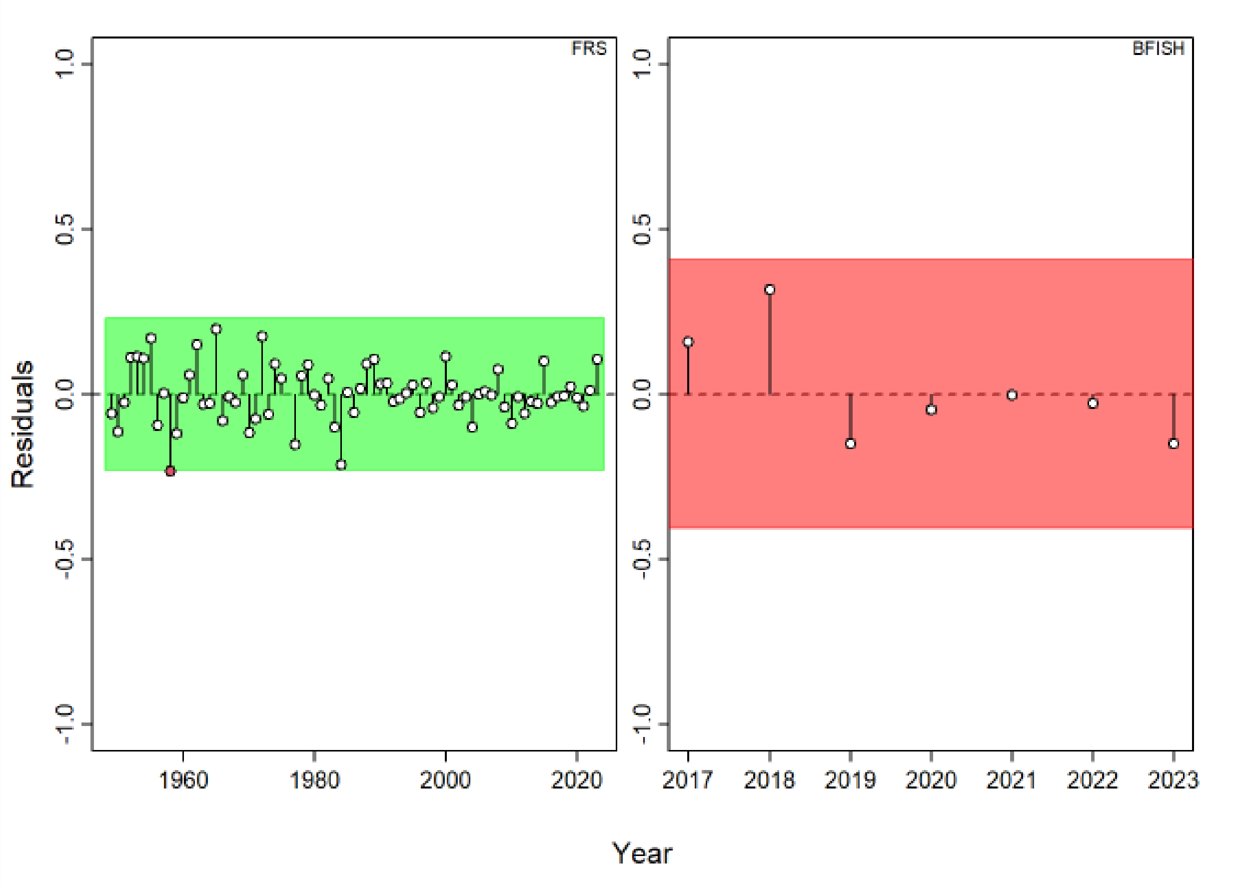


Figure 16. Runs test for FRS CPUE (left) and BFISH index of abundance (right). FRS CPUE passed the runs tests (green color) but the BFISH index of abundance did not (red color). One year residual from FRS CPUE in 1959 was greater than three standard deviations away from the mean as indicated by the red circle.

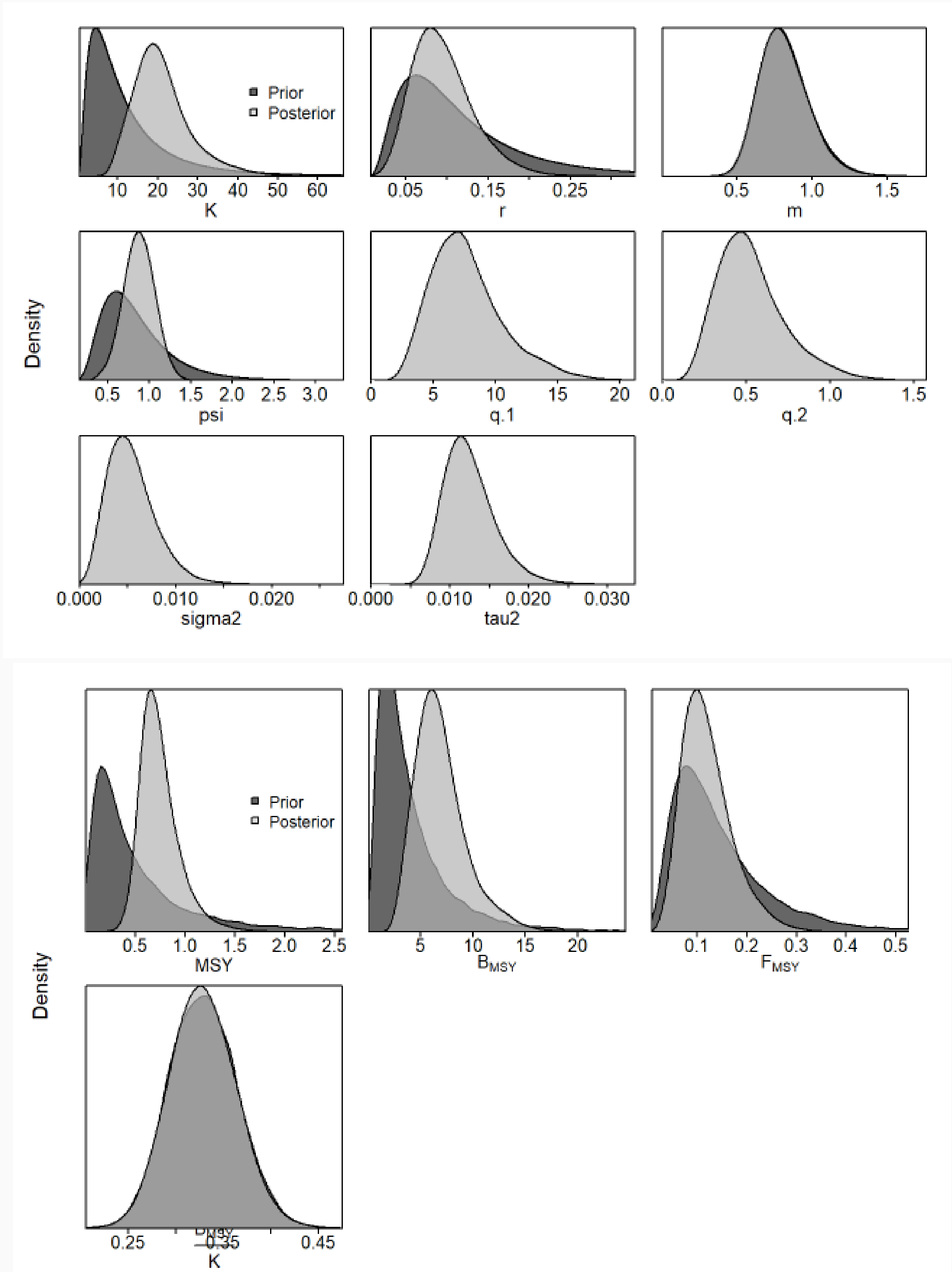


Figure 17. Prior and posterior distributions of parameters and derived quantities.

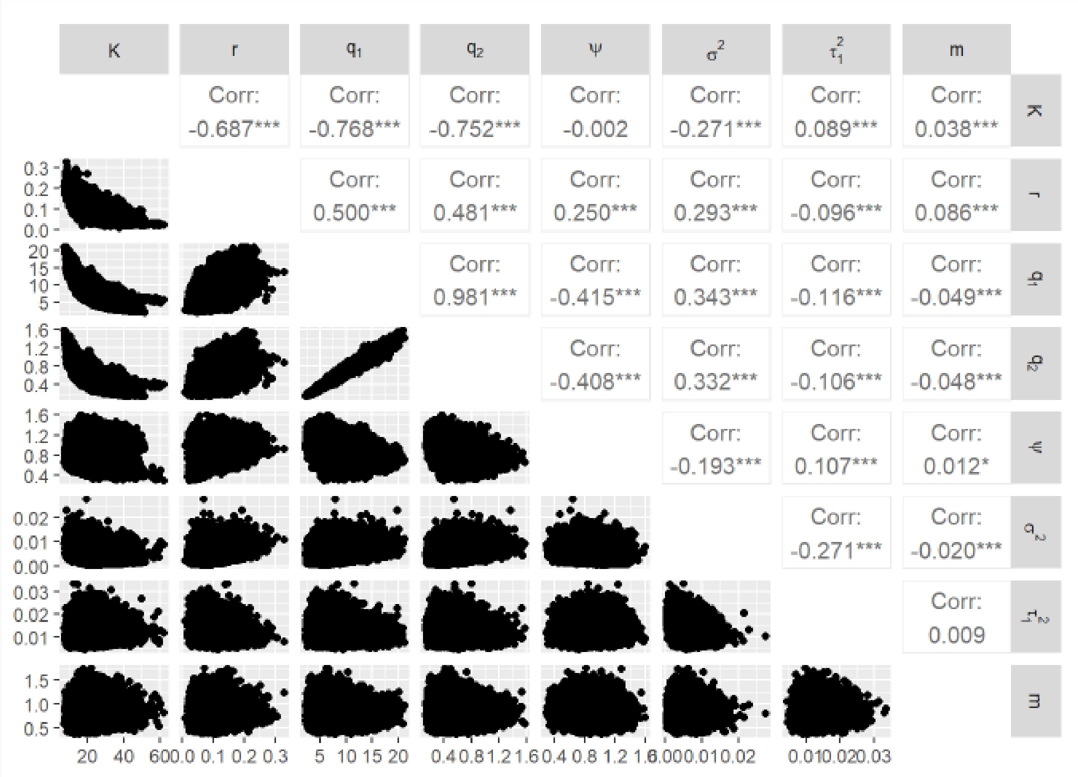


Figure 18. Pairwise scatterplots of parameter estimates. Parameters are carrying capacity (*K*), intrinsic growth rate (*r*), catchability for FRS CPUE (*q*1) and BFISH index of abundance (*q*2), initial proportion of carrying capacity (*P*1), process error (σ2), and observation error (τ2).

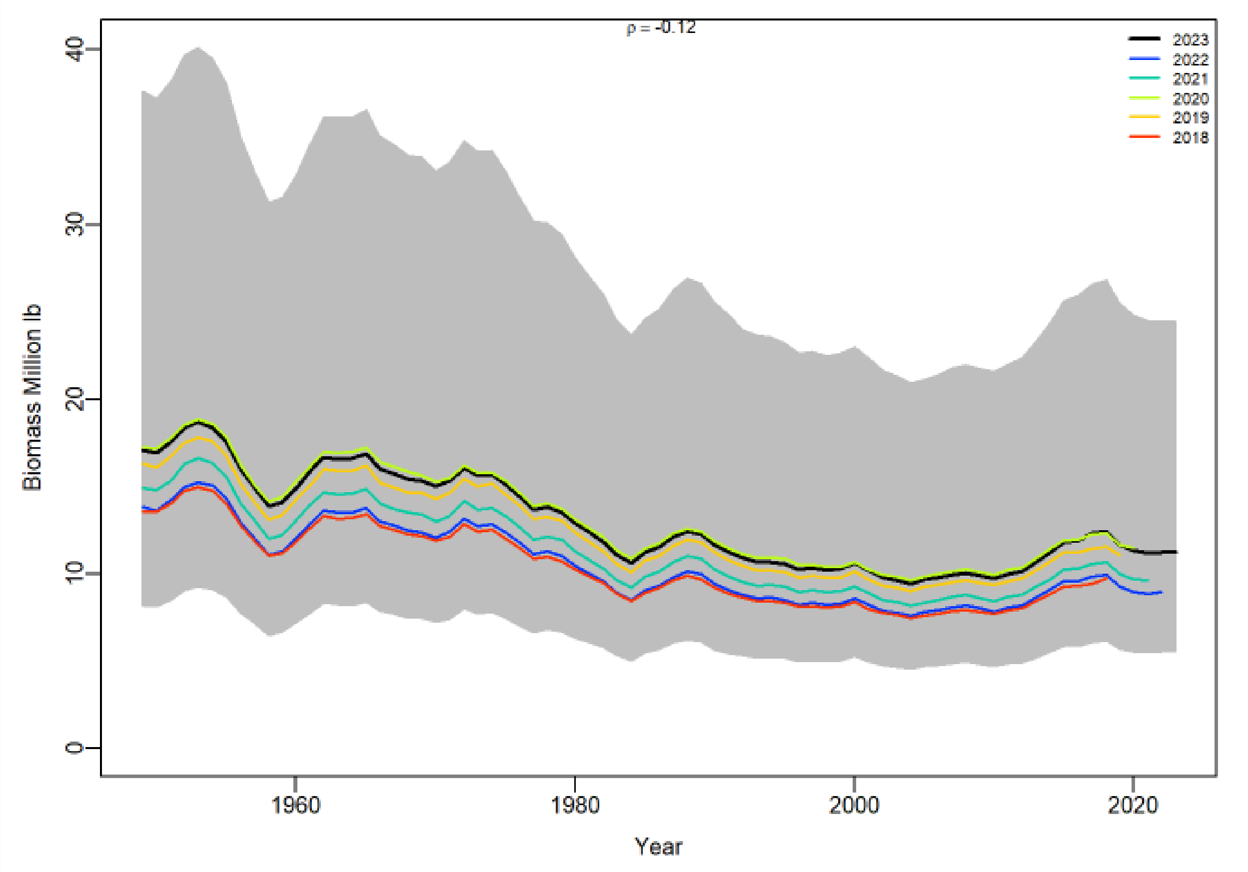


Figure 19. Exploitable biomass estimates from the retrospective analysis with terminal year set as fishing year 2023 through 2018. A slightly negative trend was observed (Mohn's rho = -0.12), but it was within the acceptable range.

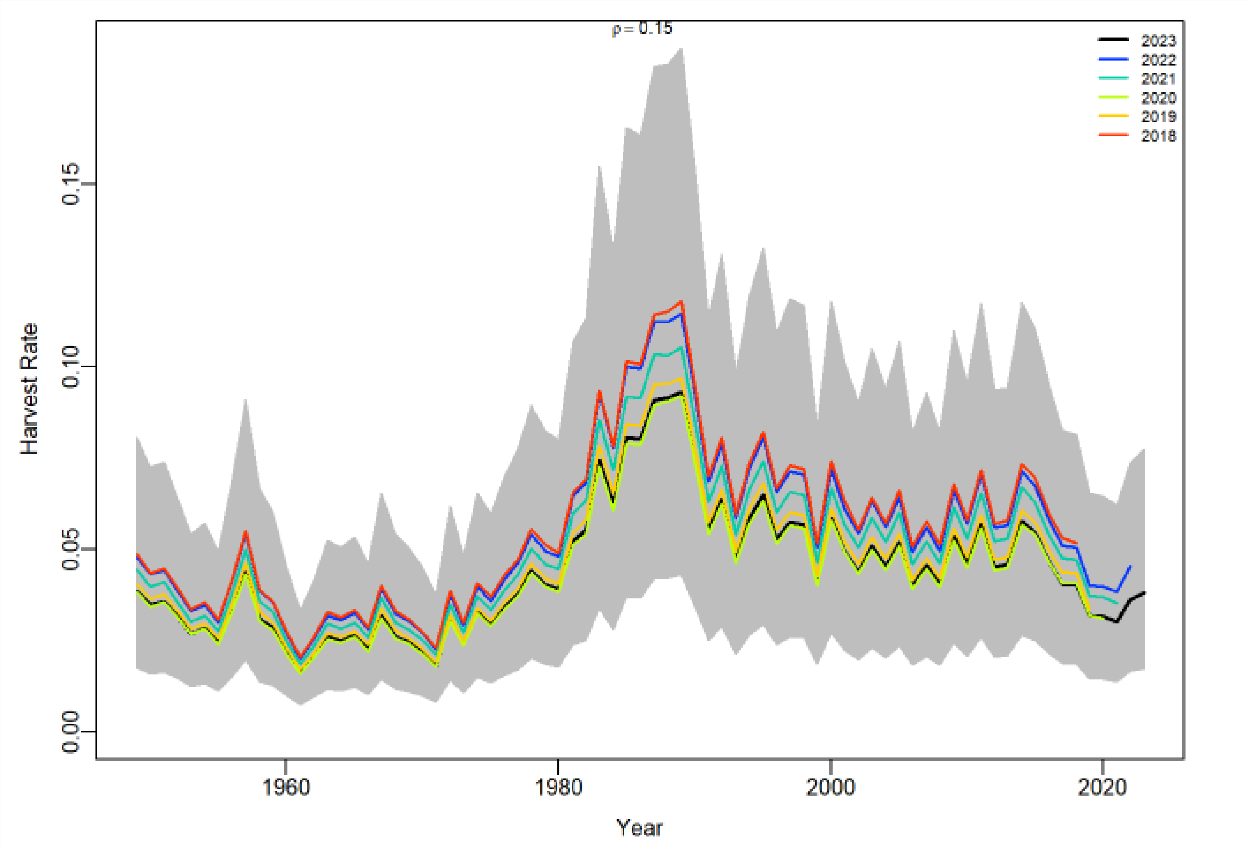


Figure 20. Harvest rate mortality estimates from the retrospective analysis with terminal year set as fishing year 2023 through 2018. A slightly positive bias was observed (Mohn's rho = 0.15), but still within the acceptable range.

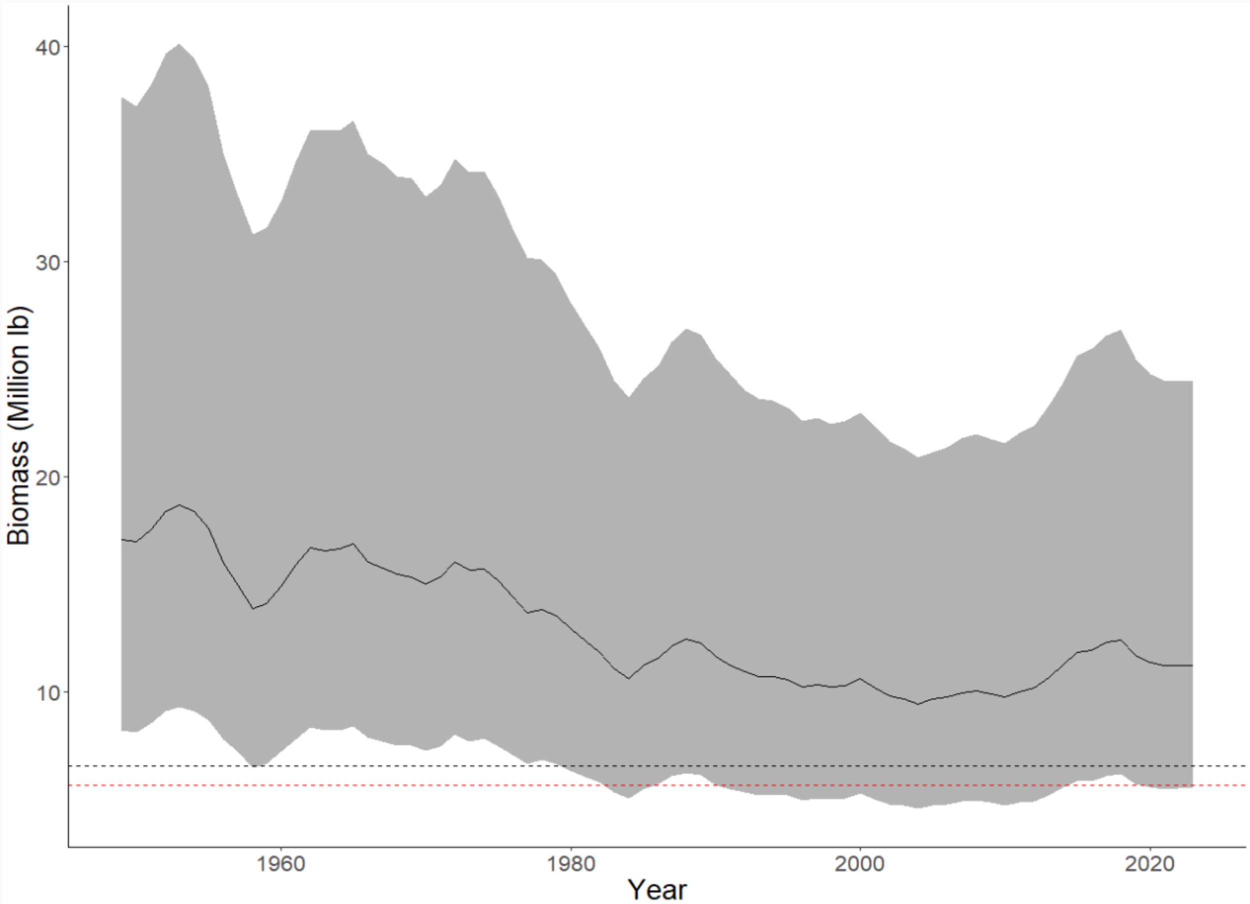


Figure 21. Estimated exploitable biomass (solid line) and 95% credible intervals (gray shaded area) for the Deep 7 bottomfish complex in the main Hawaiian Islands from 1949 through 2023. The black dashed line delineates B_{MSY} and red dashed line indicates B_{MSST}.

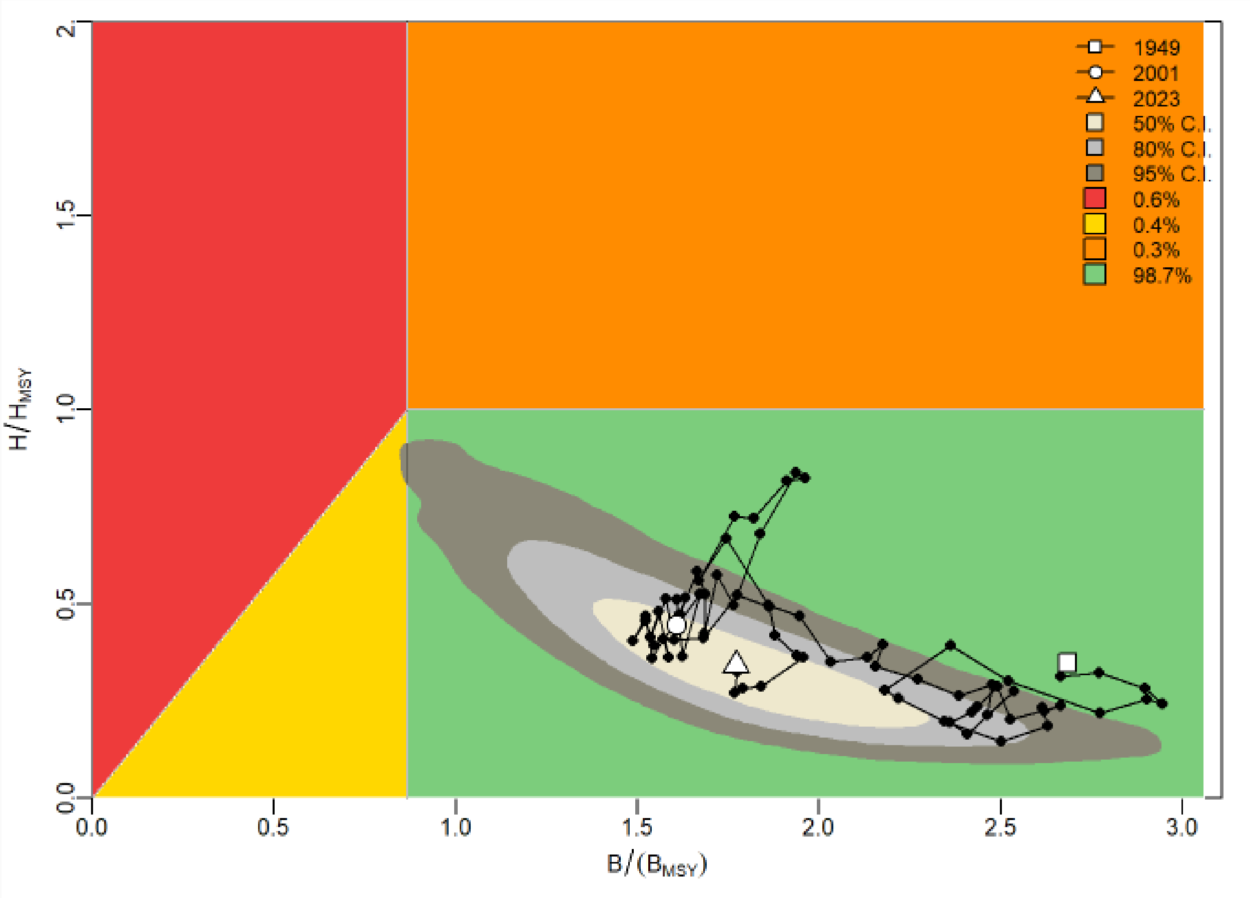


Figure 22. Estimated status for the Deep 7 Bottomfish complex in the main Hawaiian Islands from 1949 through 2023. The white square indicates the first year of the model (1949), the circle indicates the year 70% through the time series (2001), and the triangle indicates the terminal year (2023). The shaded areas represent the 50%, 80%, 95% credible intervals for the terminal year estimate.

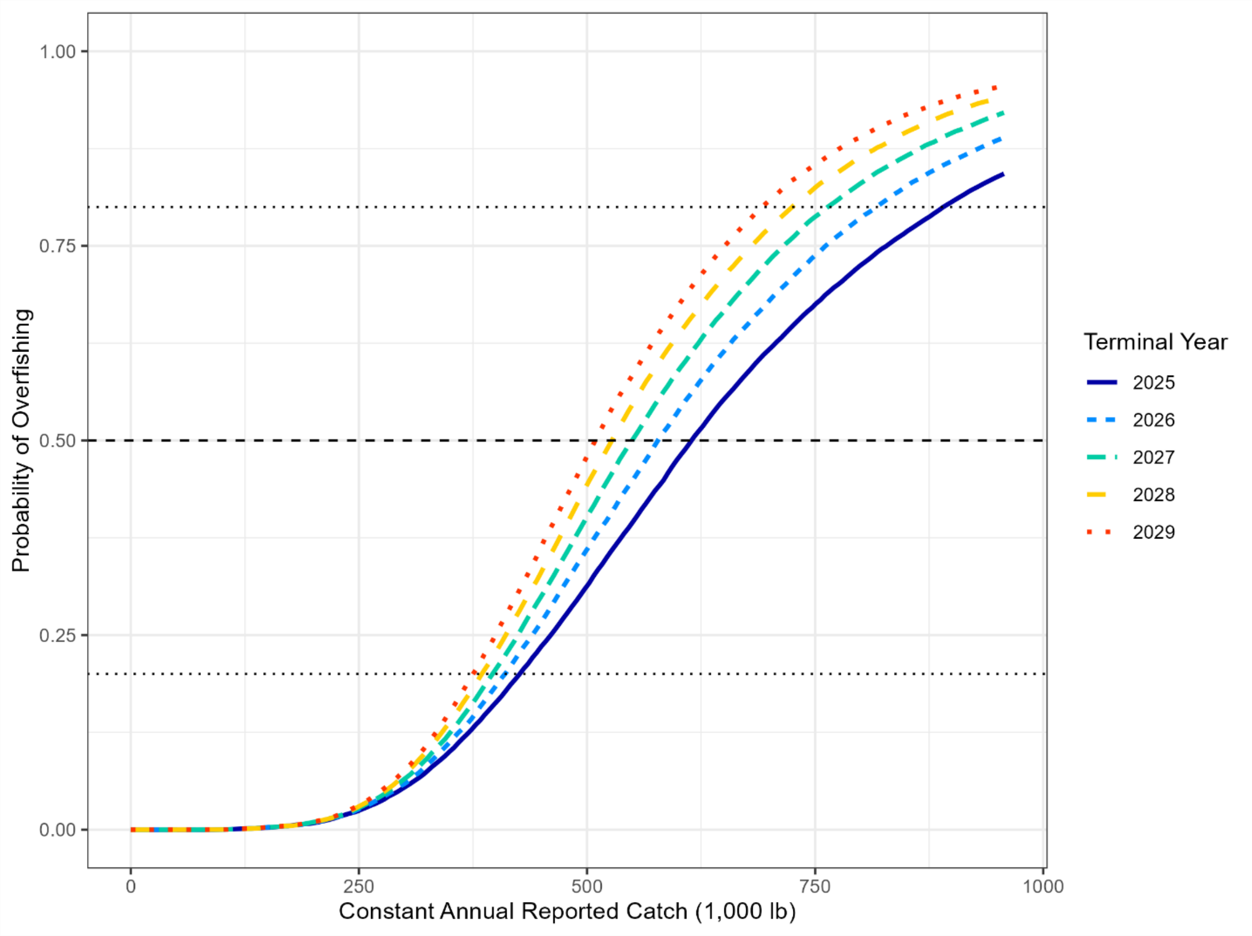


Figure 23. Probability of overfishing (i.e., H/H_{MSY}>1) Deep 7 bottomfish complex in the main Hawaiian Islands assuming fixed, constant reported catch through the terminal years (2025 through 2029, represented by line color and type). Reported catch ranged from zero to one million pounds.

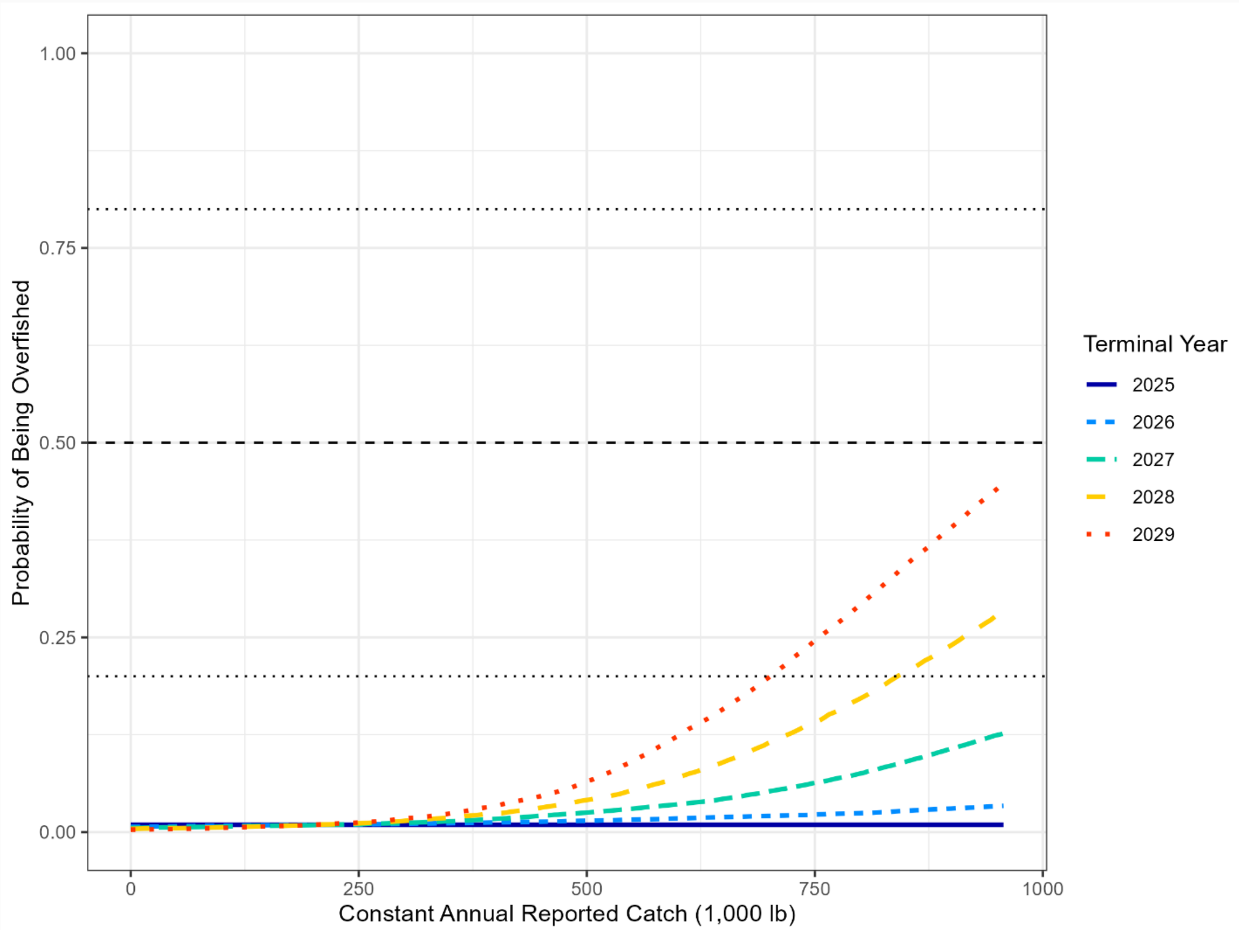


Figure 24. Probability of the stock being overfished (i.e., B/B_{MSY}<0.867) for Deep 7 bottomfish complex in the main Hawaiian Islands assuming fixed, constant reported catch through the terminal years (2025 through 2029, represented by line color and type). Reported catch ranged from zero to one million pounds.

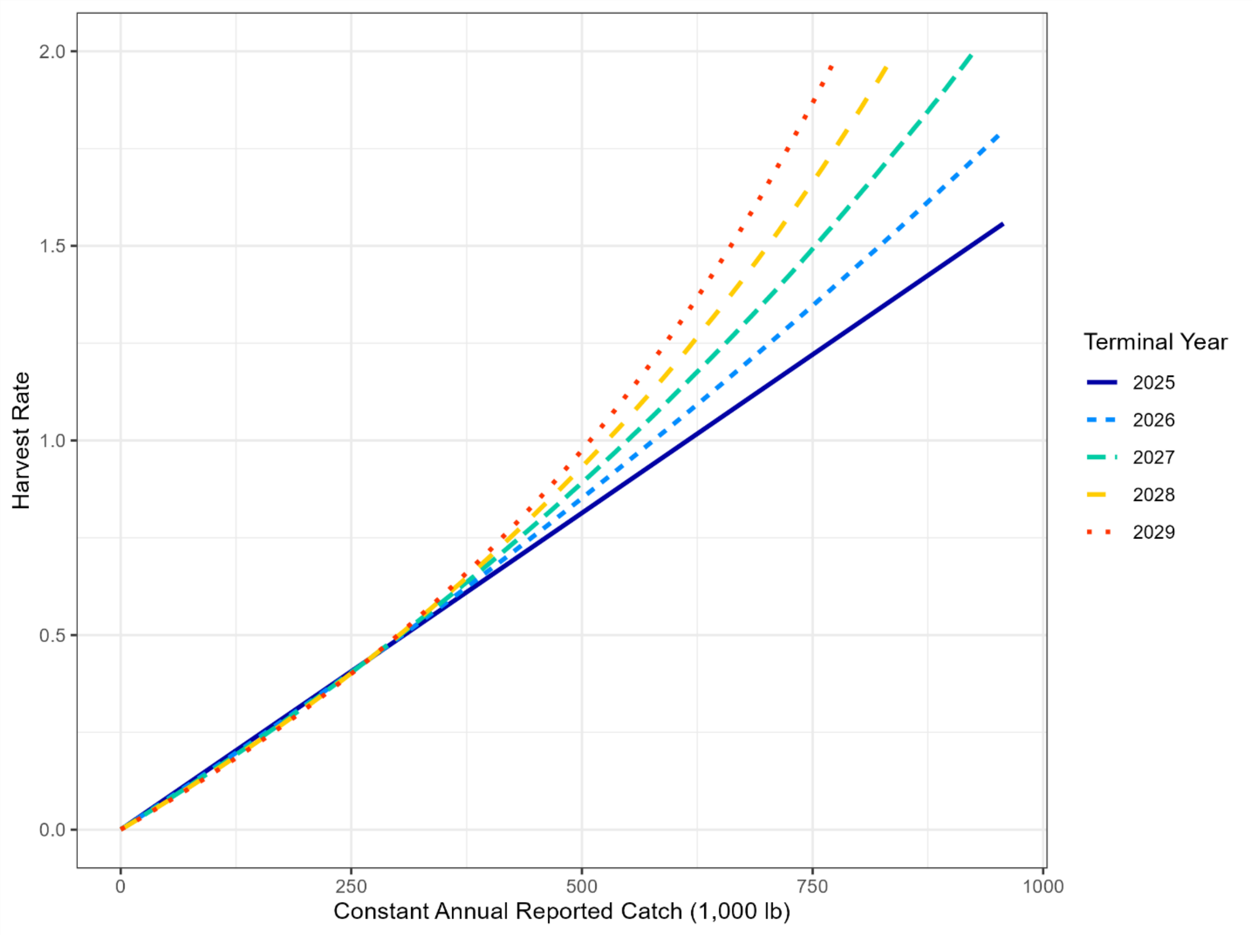


Figure 25. Median harvest rate for Deep 7 bottomfish complex in the main Hawaiian Islands assuming fixed, constant reported catch through the terminal years (2025 through 2029, represented by line color and type). Reported catch ranged from zero to one million pounds.

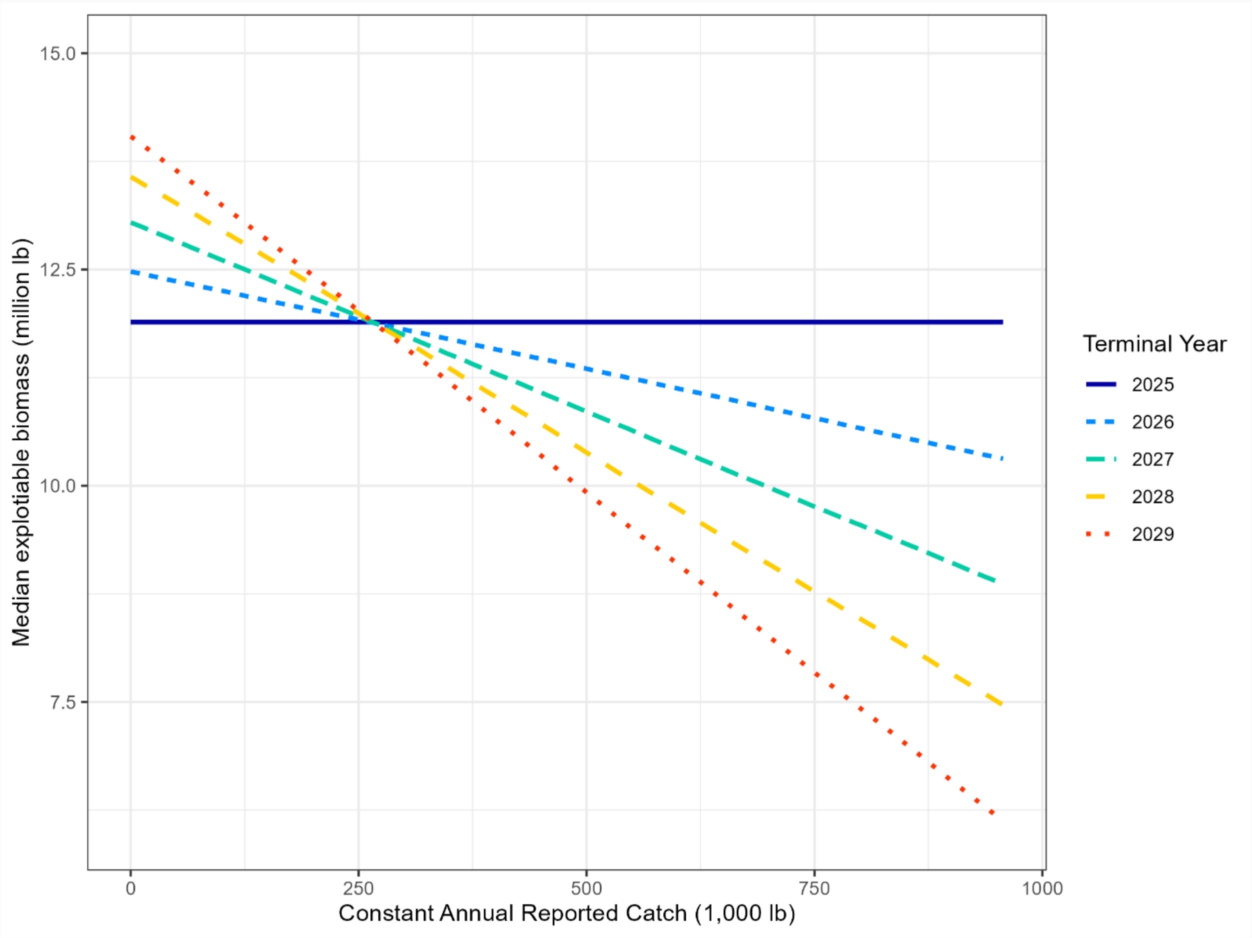


Figure 26. Median exploitable biomass for Deep 7 bottomfish complex in the main Hawaiian Islands assuming fixed, constant reported catch through the terminal years (2025 through 2029, represented by line color and type). Reported catch ranged from zero to one million pounds.

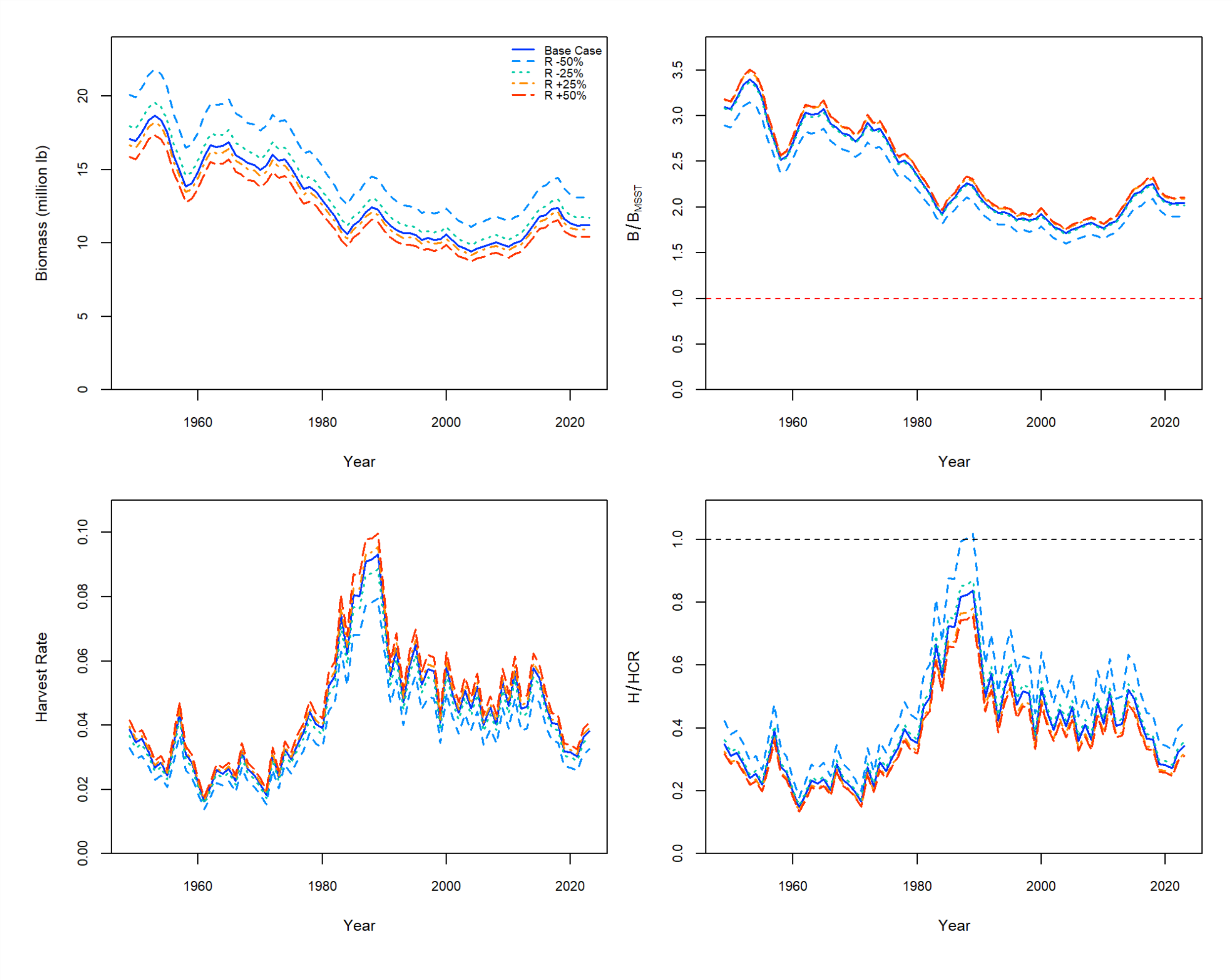


Figure 27. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model (solid blue line) and models with different prior means for intrinsic growth rate (*r*). Values of *r* were calculated as ±25% and ±50% of the mean value used for the base case (0.096, blue line).

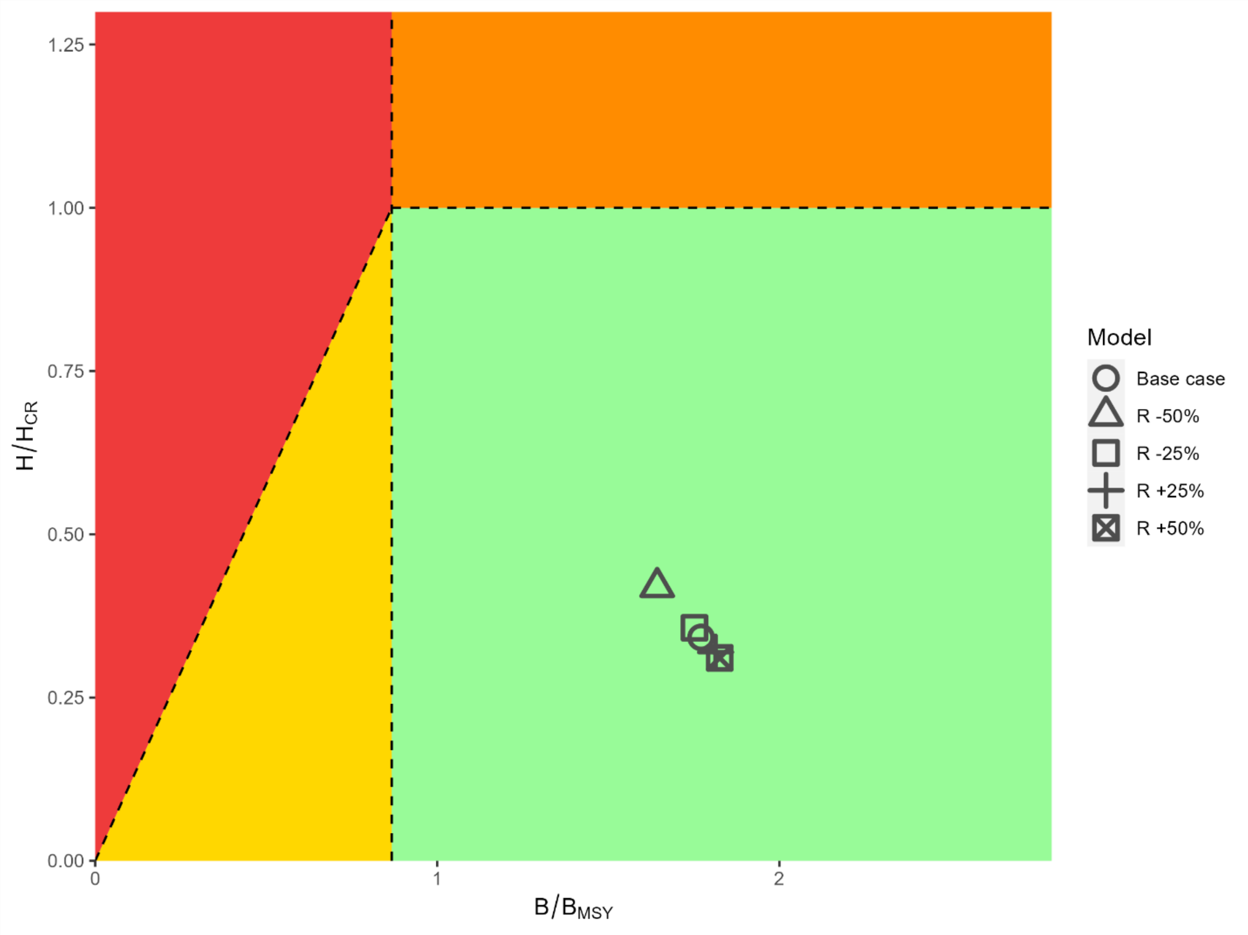


Figure 28. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing different prior means of intrinsic growth rate (*r*). Values of *r* were calculated as ±25% and ±50% of the mean value used for the base case (0.096, circle).

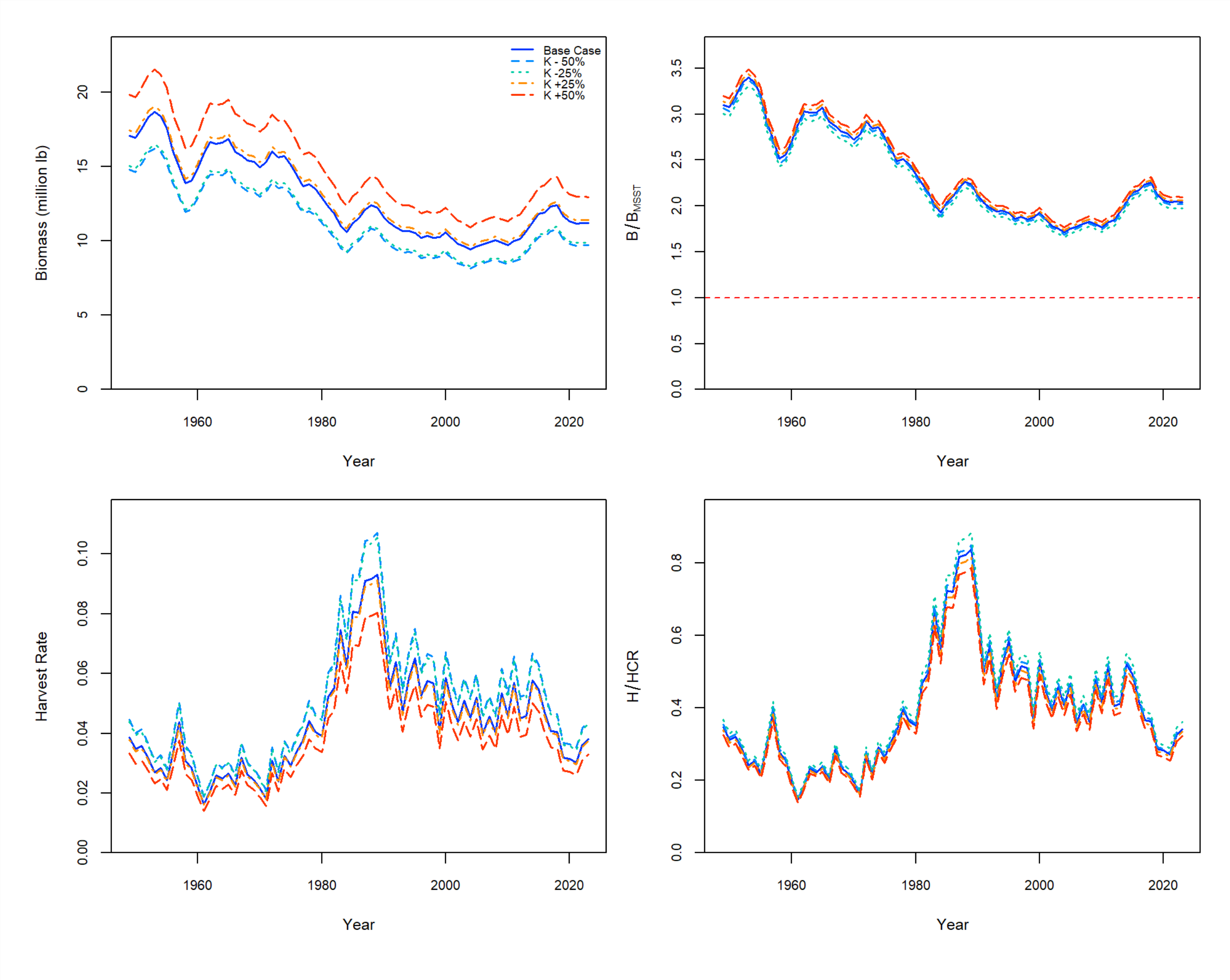


Figure 29. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model (solid blue line) and models with different prior means for carrying capacity (K). Values of K were calculated as ±25% and ±50% of the mean value used for the base case (9.14 million pounds, blue line).

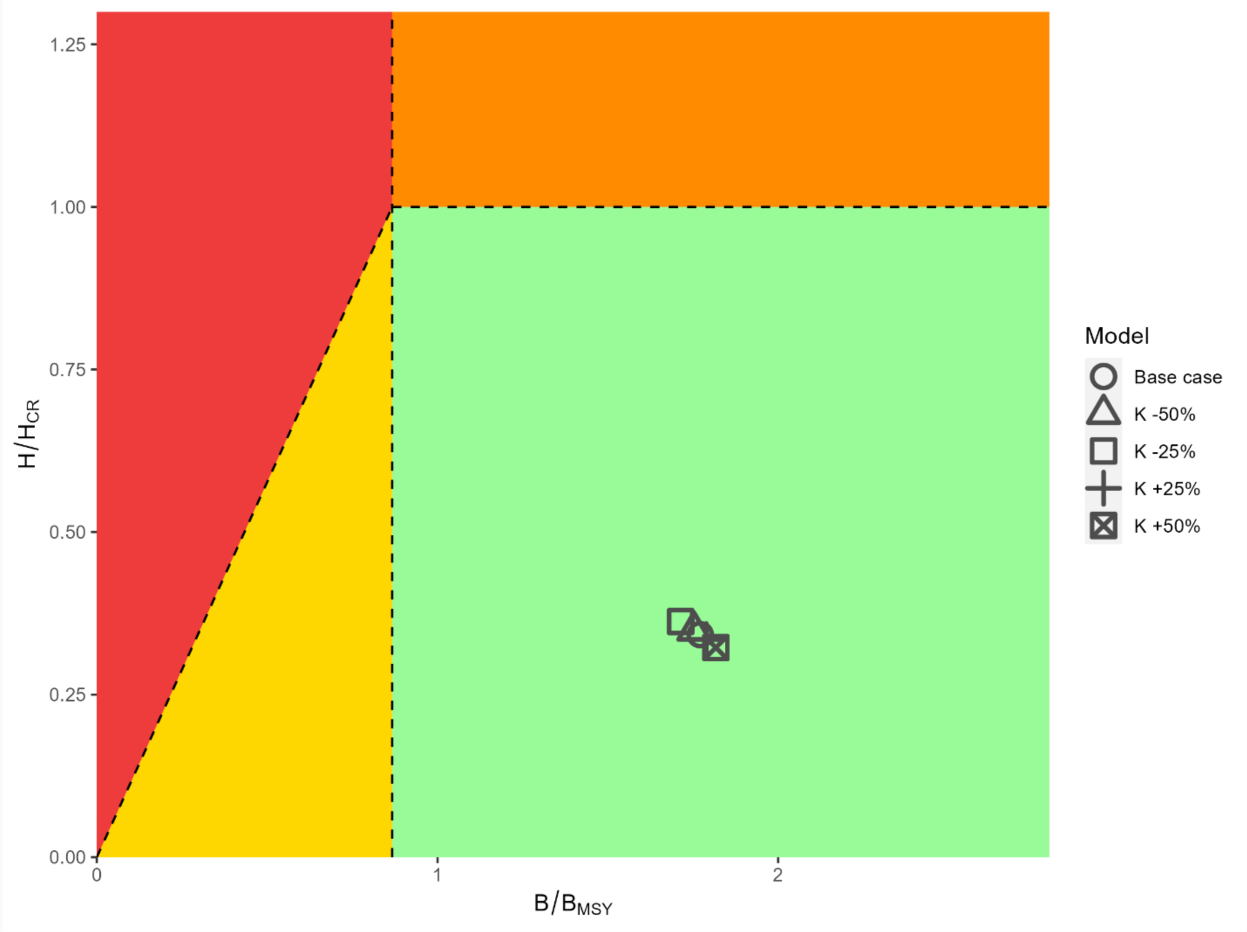


Figure 30. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing different prior means of carrying capacity (*K*). Values of *K* were calculated as ±25% and ±50% of the mean value used for the base case (9.14 million pounds, circle).

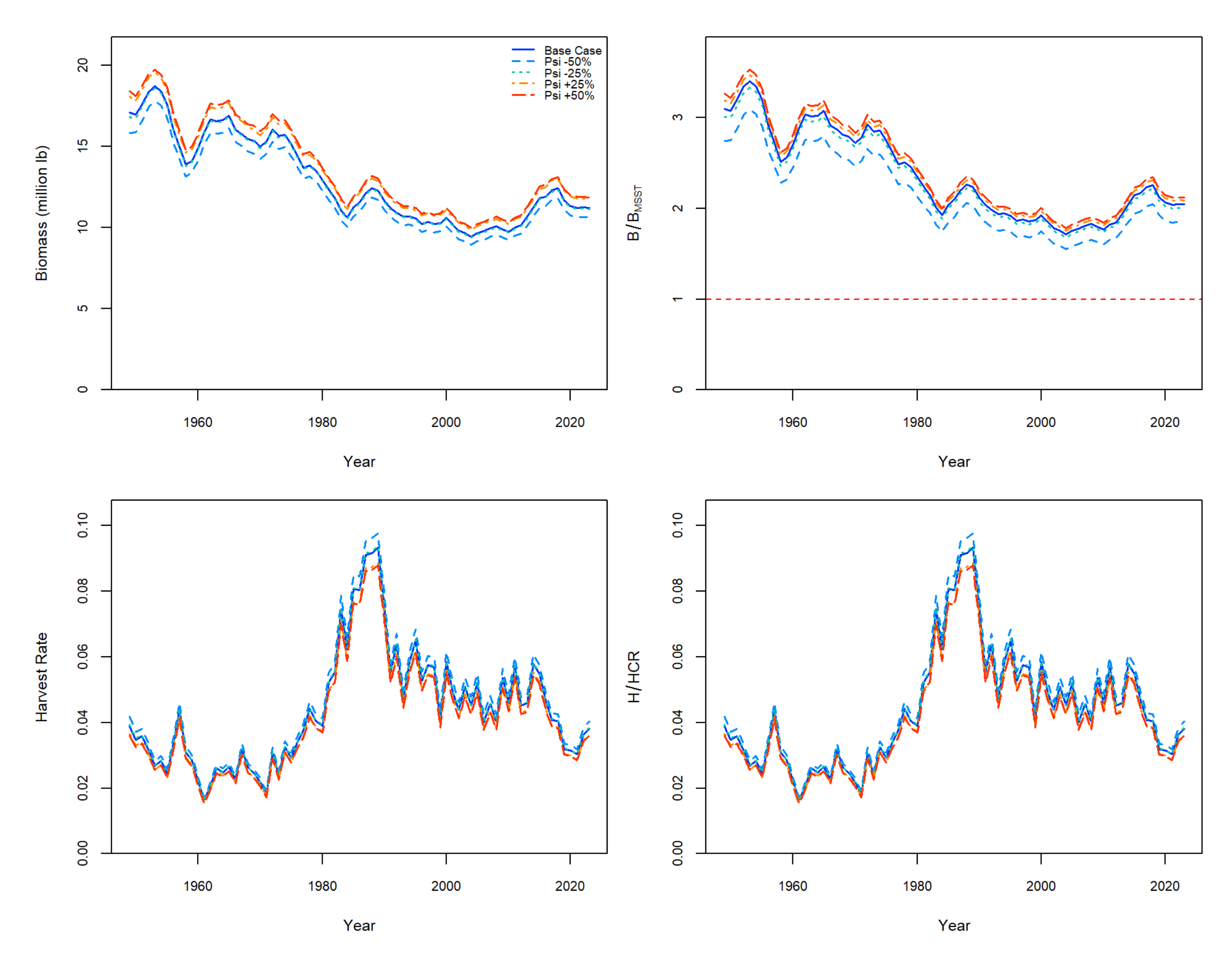


Figure 31. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model (solid blue line) and models with different prior means for initial proportion of carrying capacity (ψ). Values of ψ were calculated as ±25% and ±50% of the mean value used for the base case (0.75, blue line).

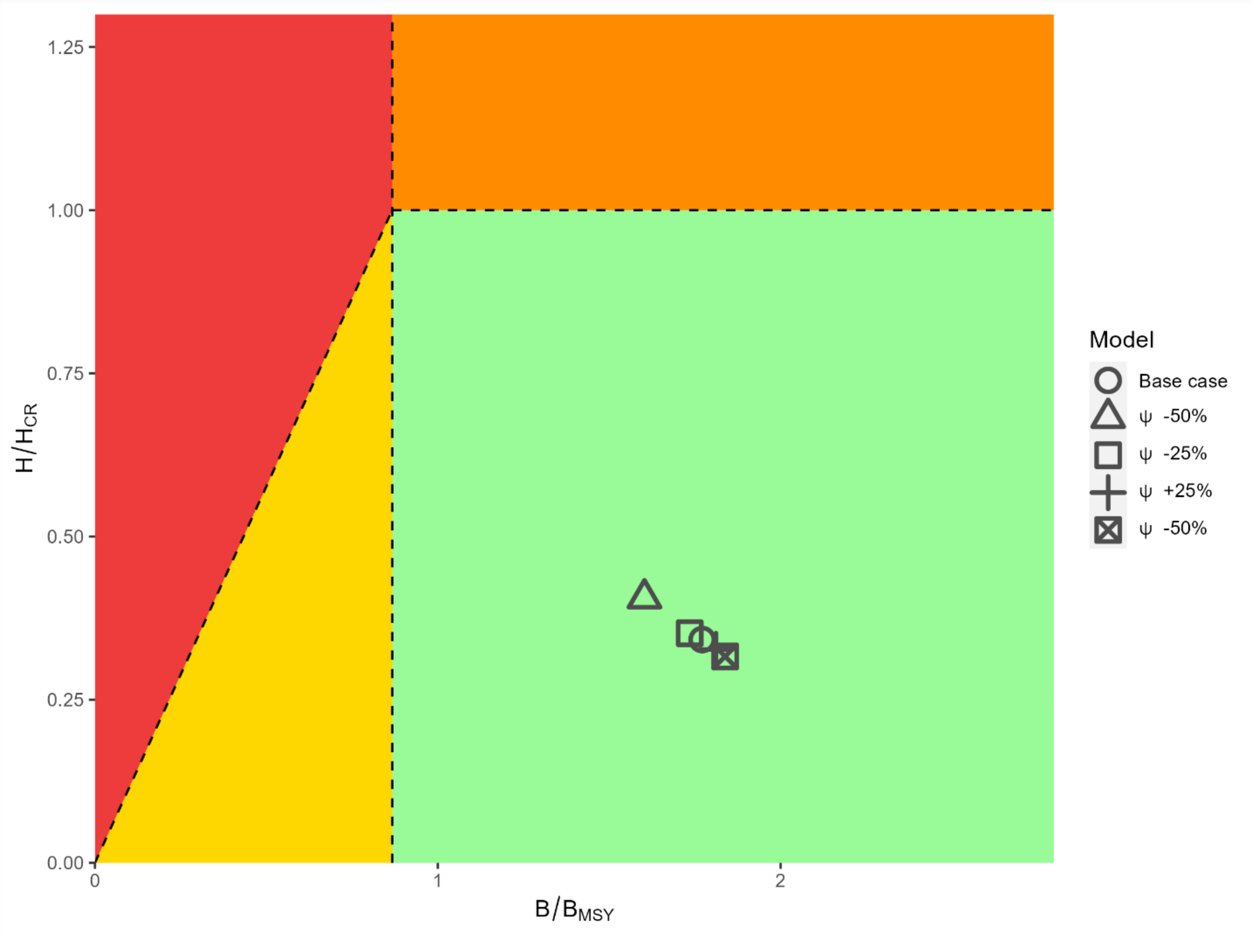


Figure 32. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing different prior means of initial proportion of carrying capacity (ψ). Values of ψ were calculated as ±25% and ±50% of the mean value used for the base case (0.75, circle).

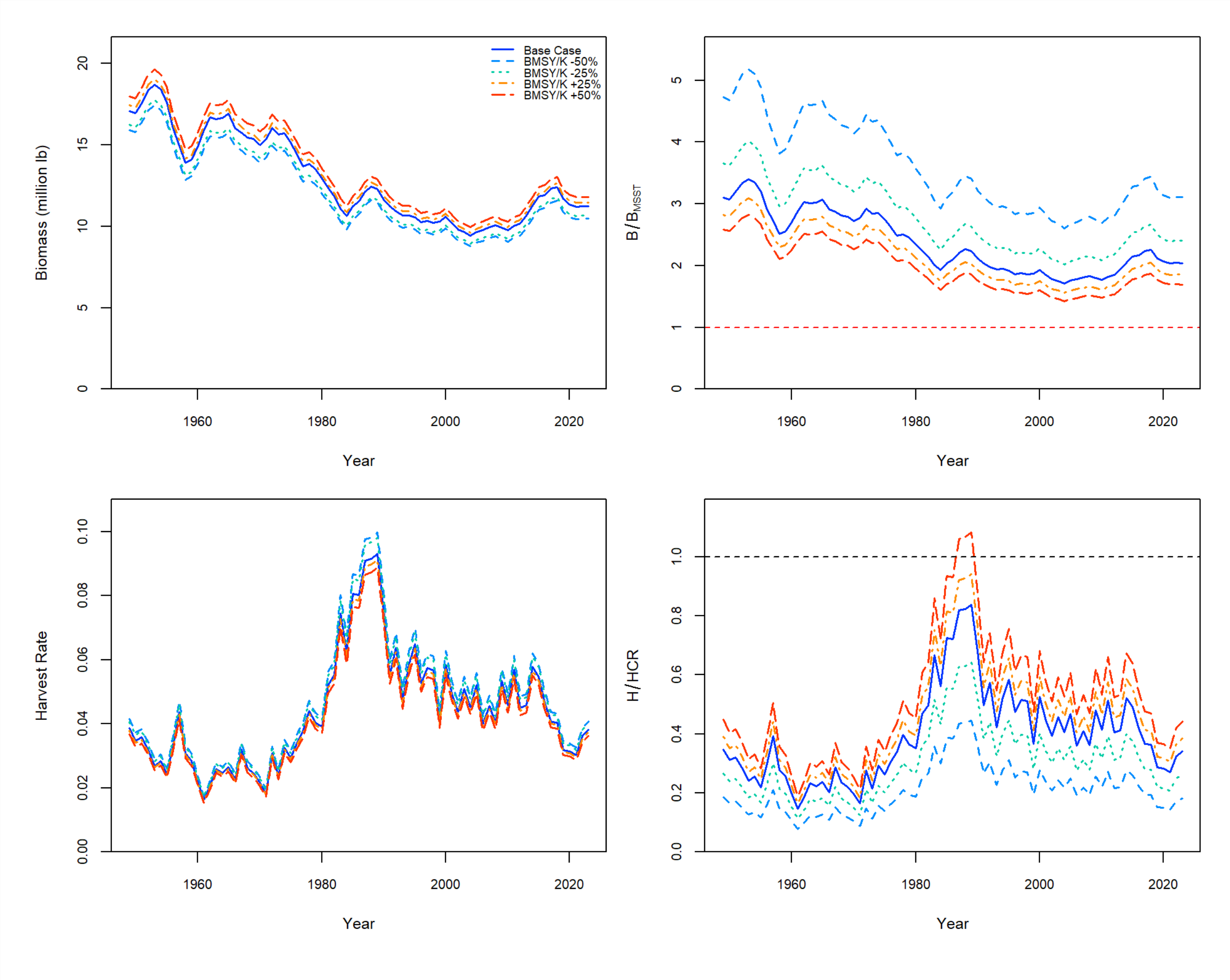


Figure 33. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model (solid blue line) and models with different prior means for B_{MSY}/K. Values of B_{MSY}/K were calculated as ±25% and ±50% of the mean value used for the base case (0.329, blue line).

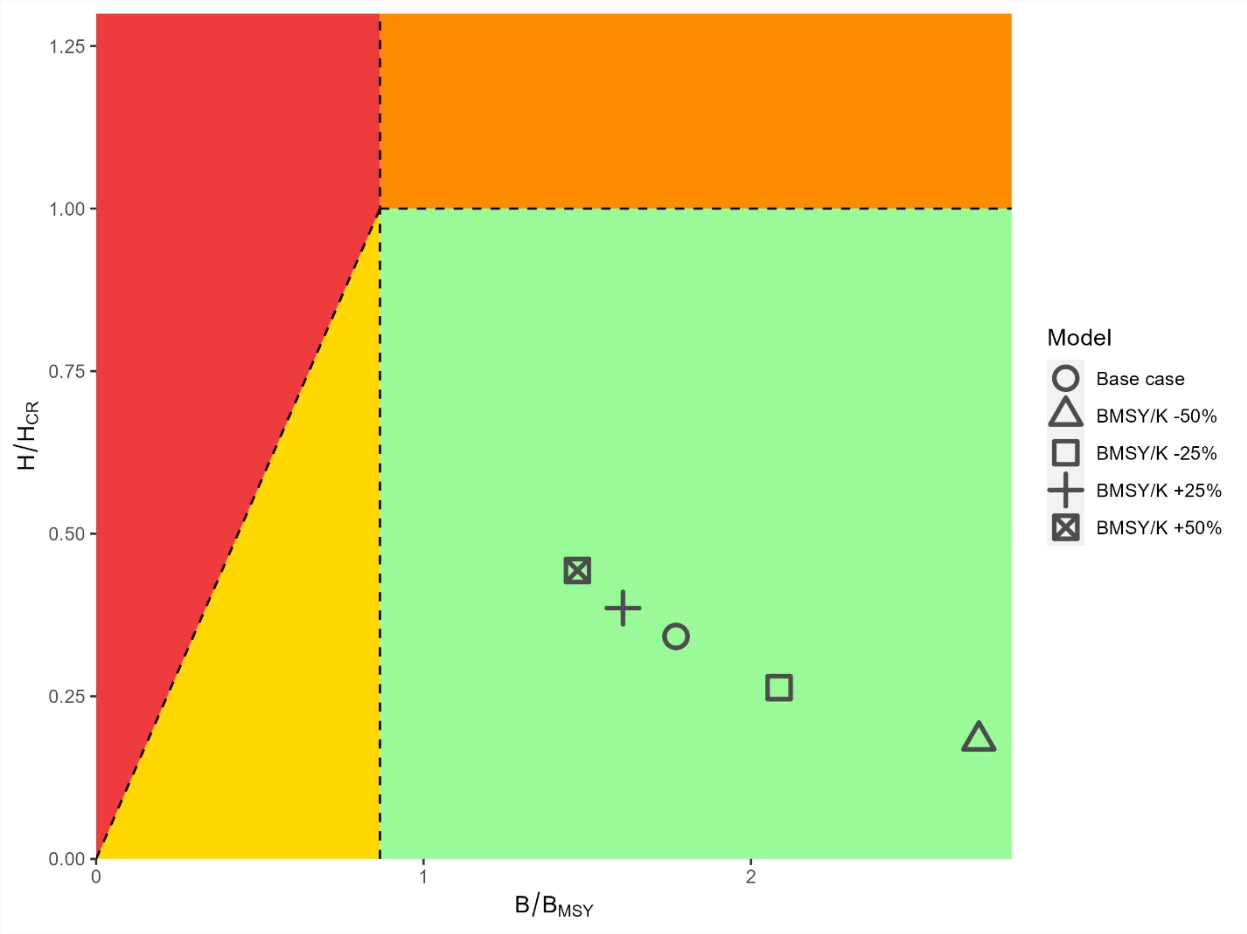


Figure 34. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing different prior means of B_{MSY}/K. Values of B_{MSY}/K were calculated as ±25% and ±50% of the mean value used for the base case (0.329, blue line).

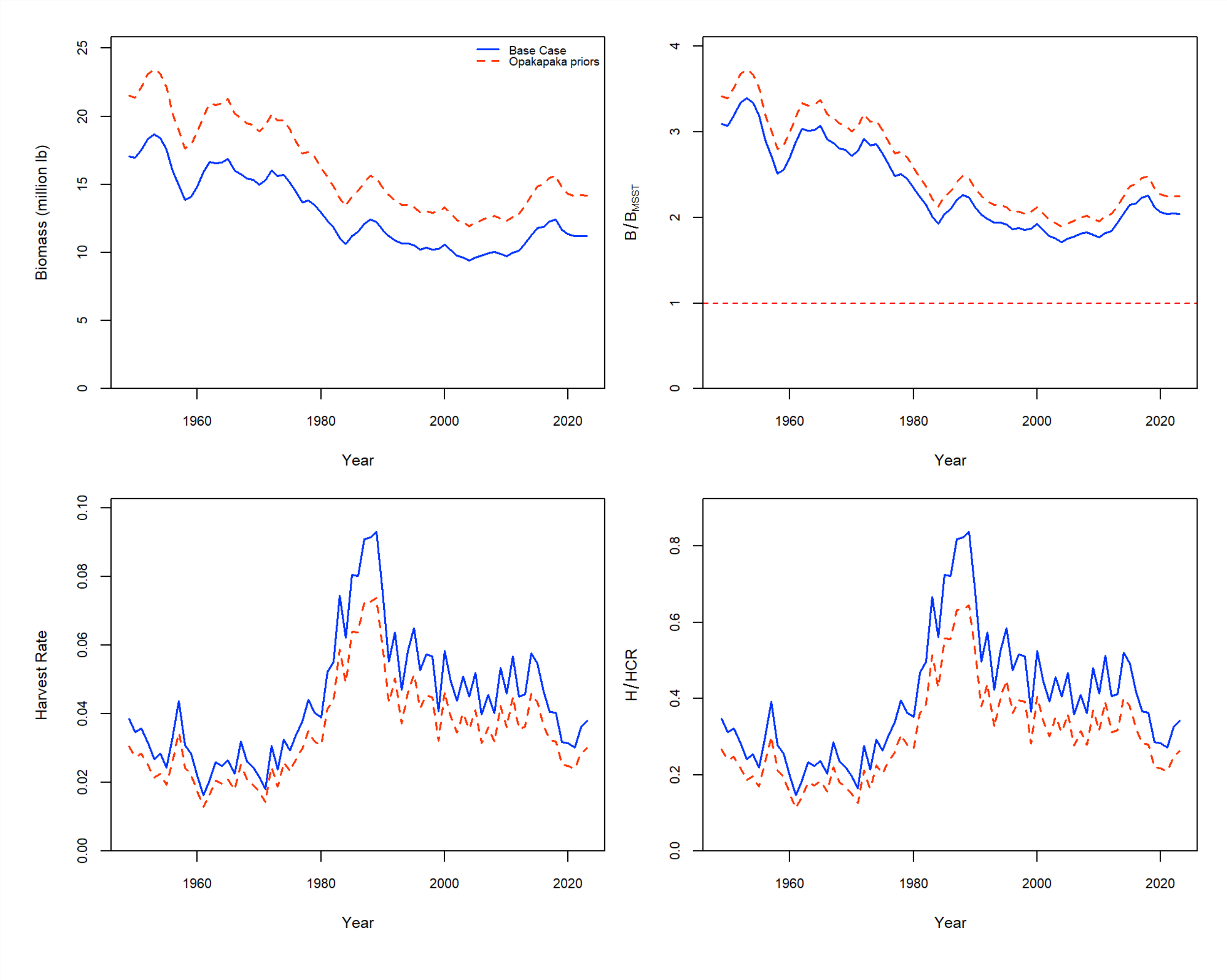


Figure 35. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model and alternative scenarios using ‘ōpakapaka-specific priors for r and B_{MSY}/K.

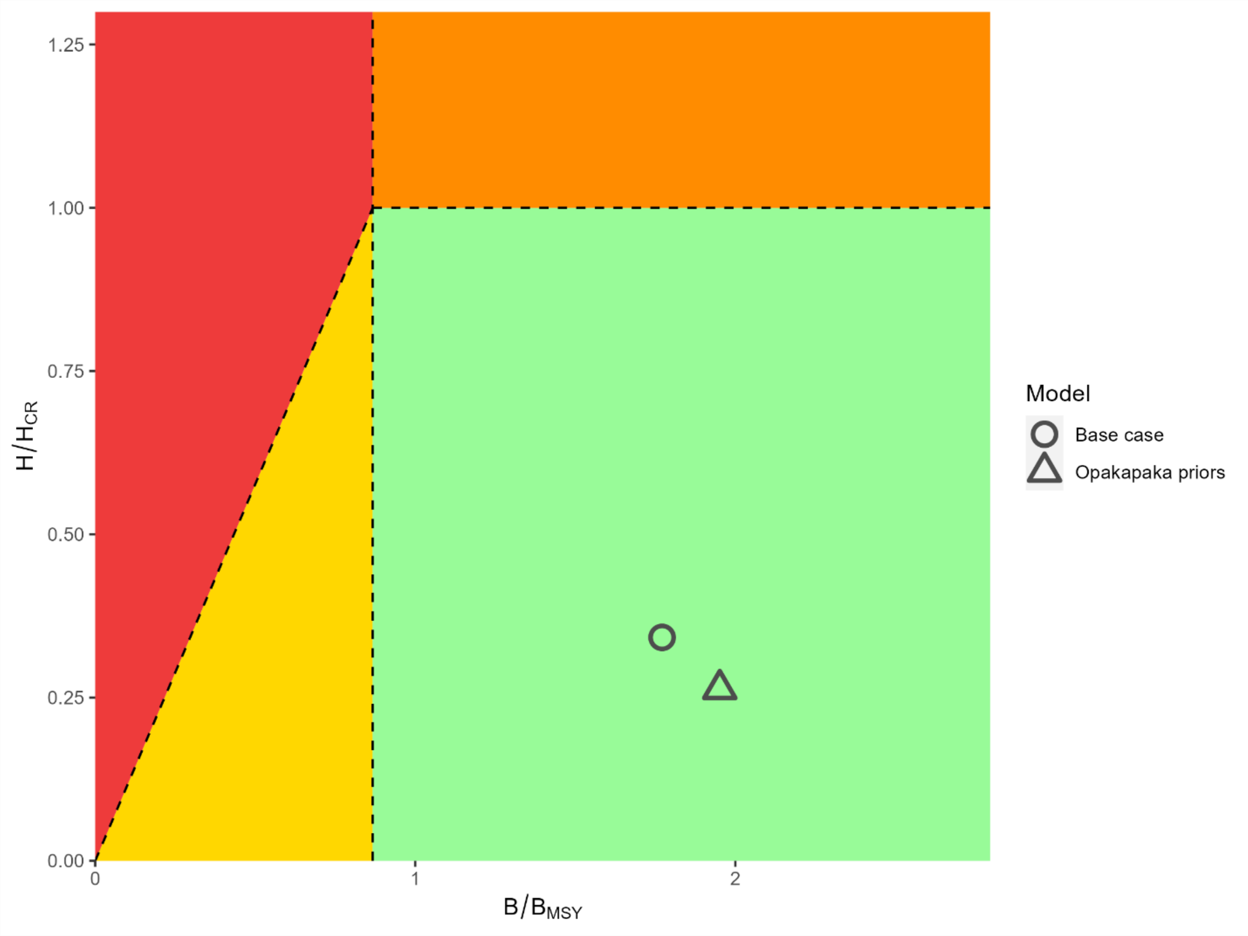


Figure 36. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing ‘ōpakapaka priors. The circle shows the stock status when complex-level priors were used (base model) and the triangle indicates the stock status when ‘ōpakapaka-specific priors were used.

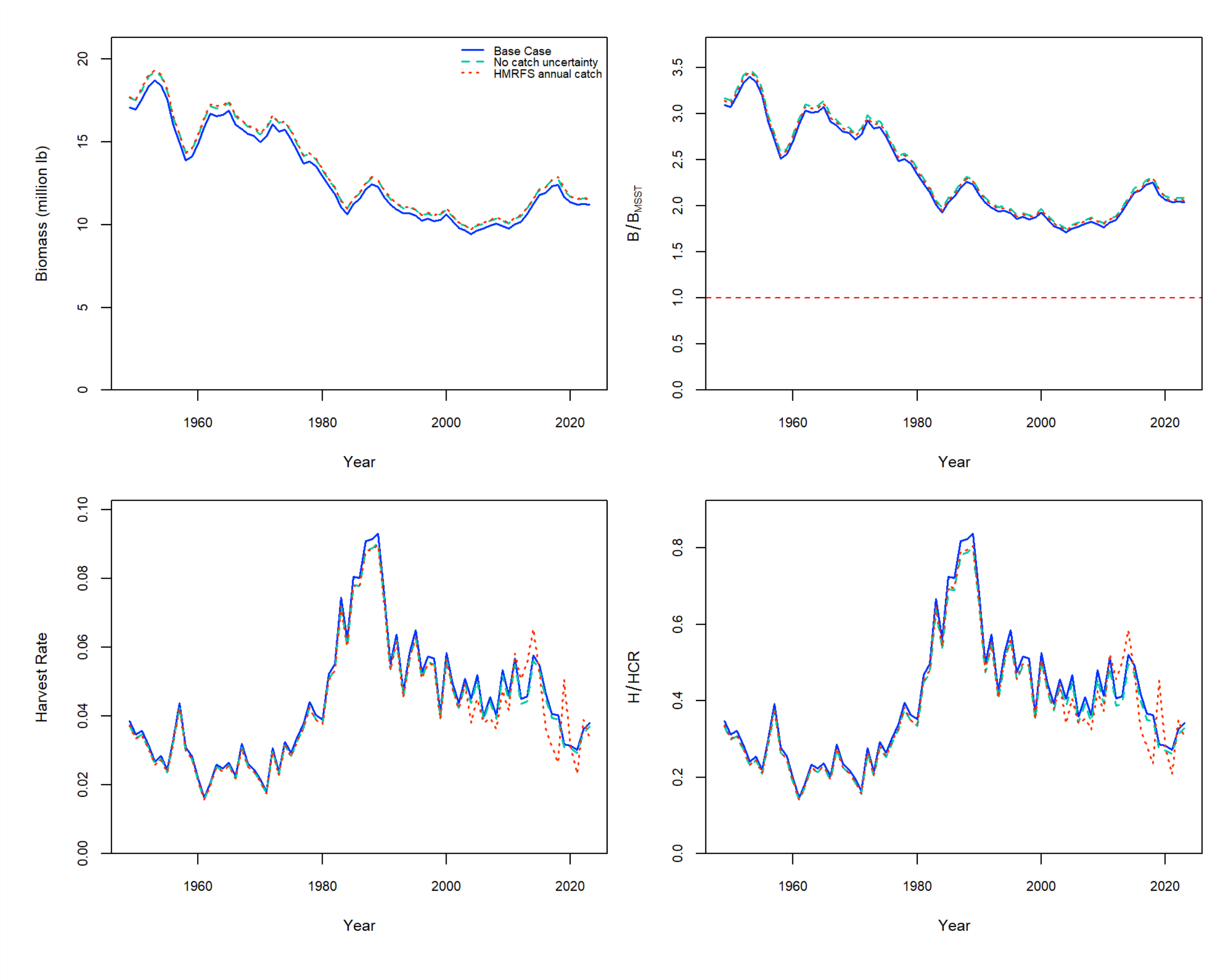


Figure 37. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model (solid blue line) and models with different treatments of catch. The alternative models were using annual estimates of HMRFS (red dashed line) and not including catch uncertainty (green long dash line).

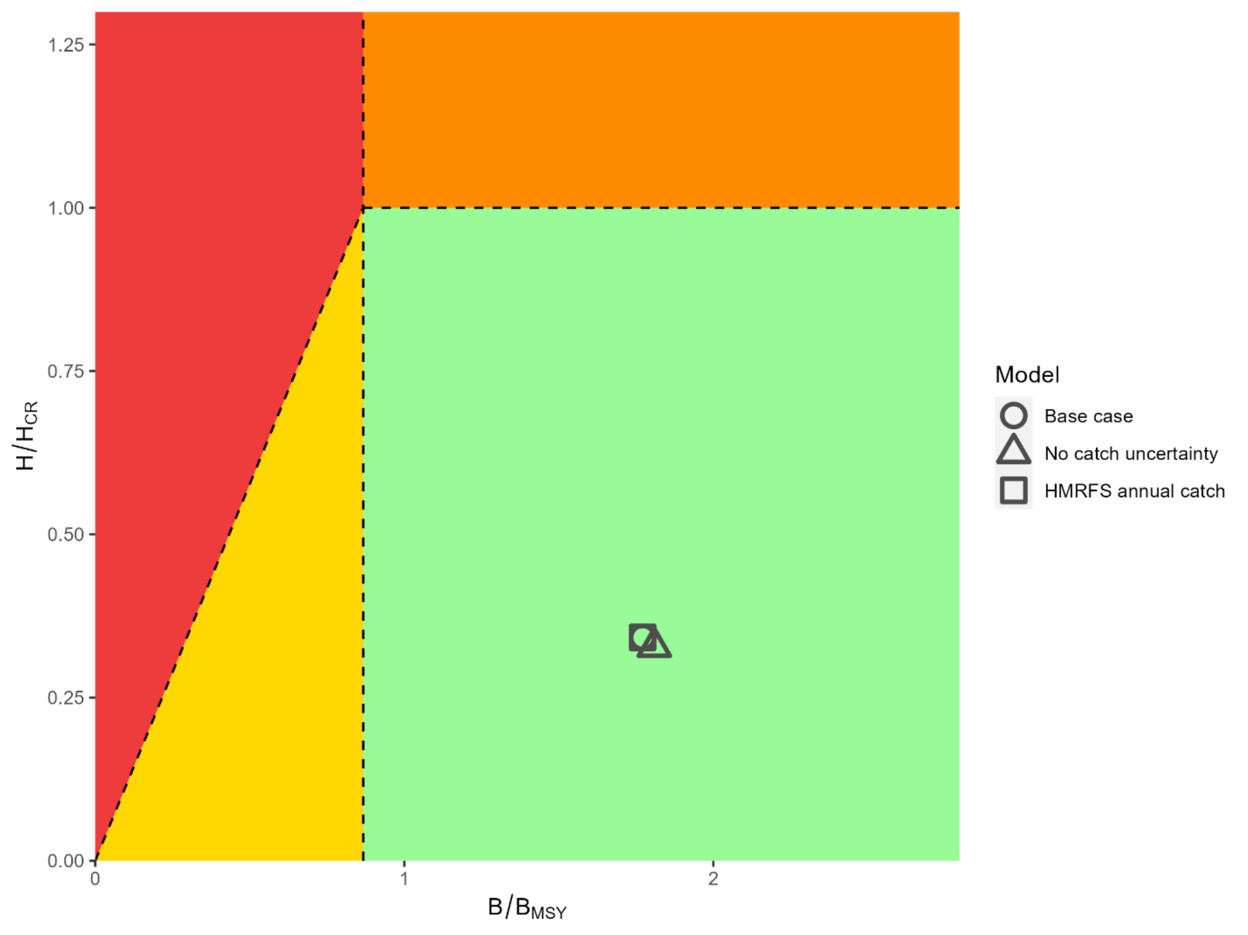


Figure 38. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing treatments of catch. The circle represents stock status under the base case, the triangle is the stock status when not including uncertainty in catch, and the square is the stock status when using HMRFS annual estimates of catch.

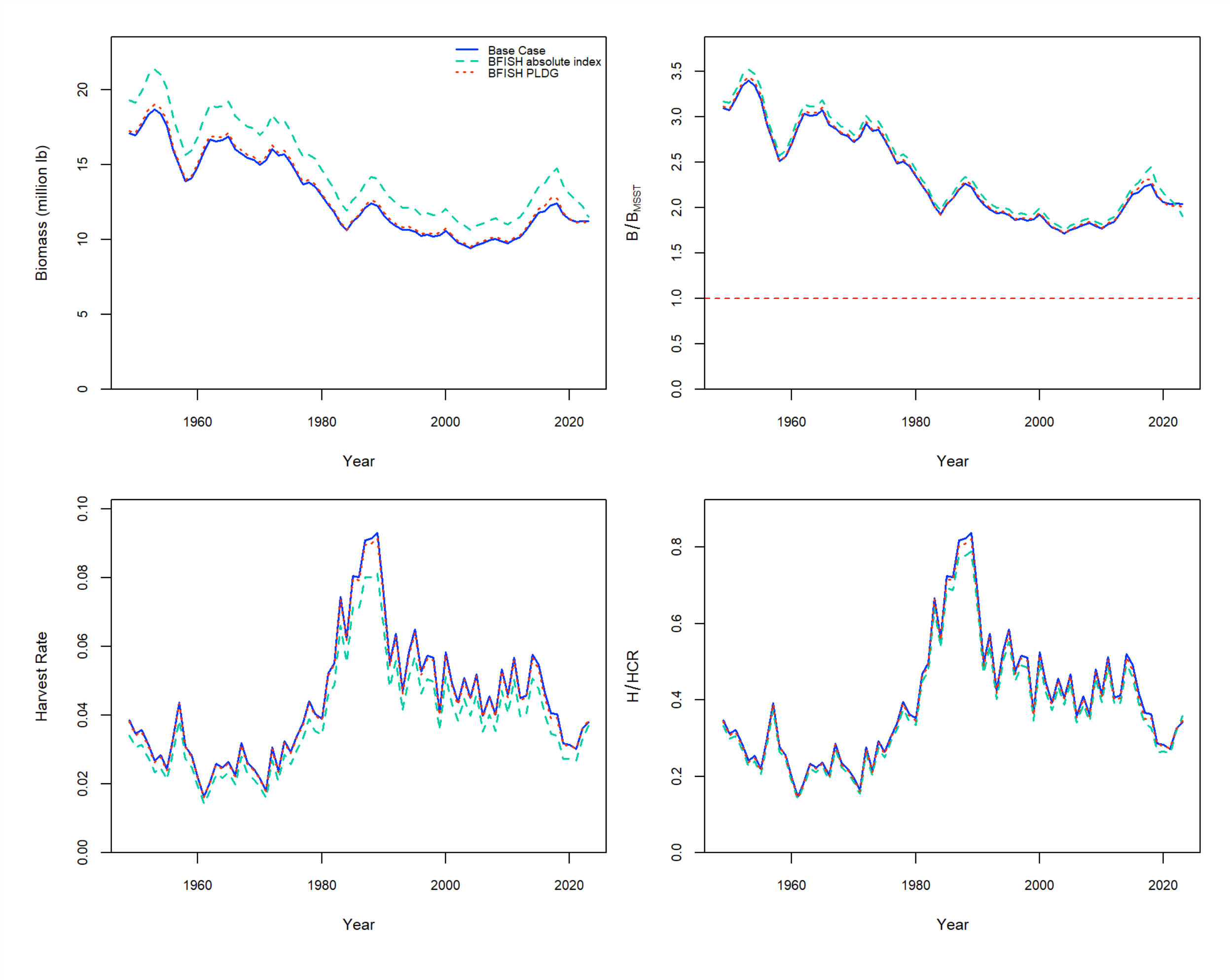


Figure 39. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model (solid blue line) and models with different treatments of the BFISH index of abundance. The alternative models were using a different standardization model (red dashed line) and treating it as an absolute abundance (green long dash line).

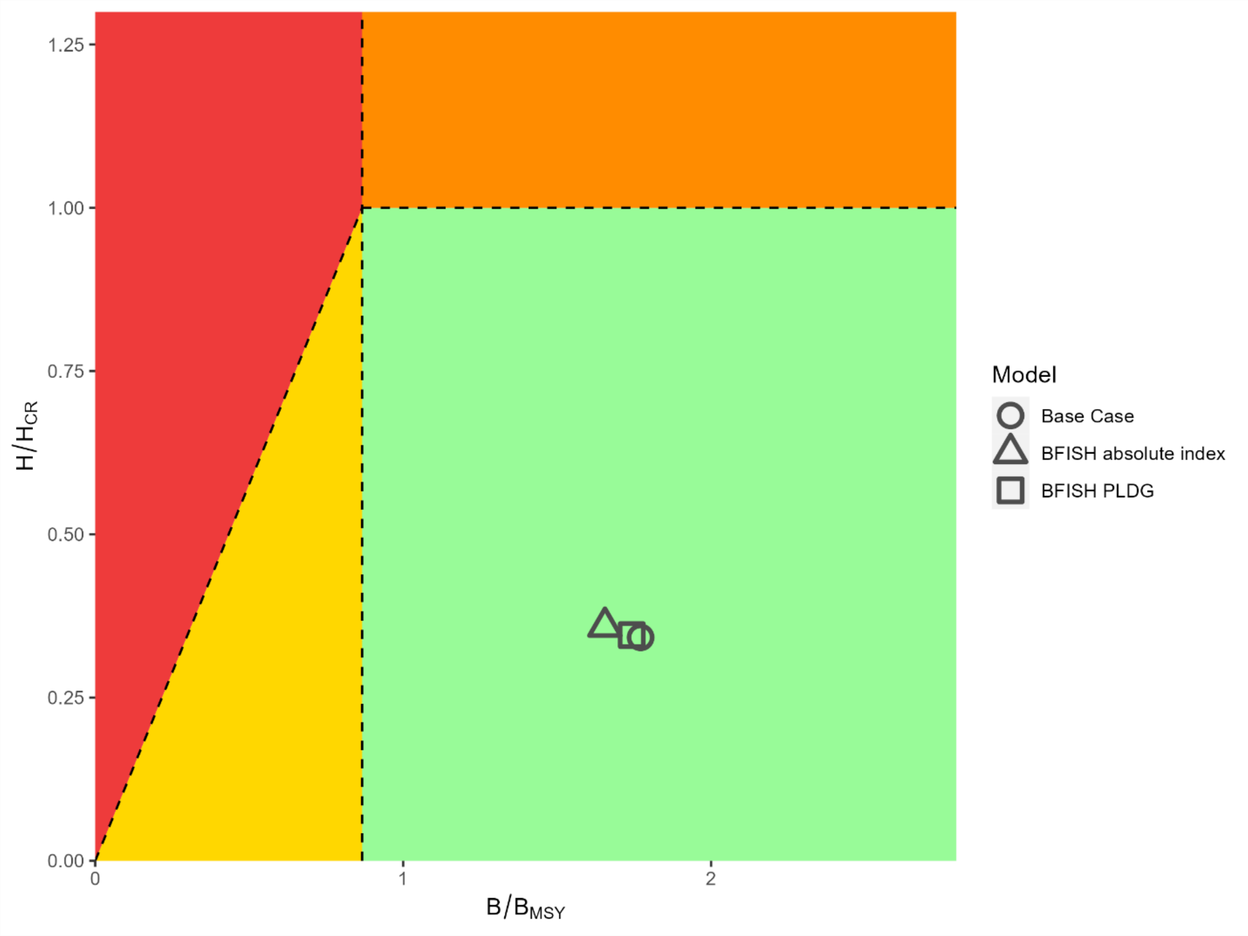


Figure 40. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing treatments of BFISH index of abundance. The circle represents stock status under the base case, the square is the stock status when using an alternative standardization model, and the triangle is the stock status when treating it as an index of absolute abundance.

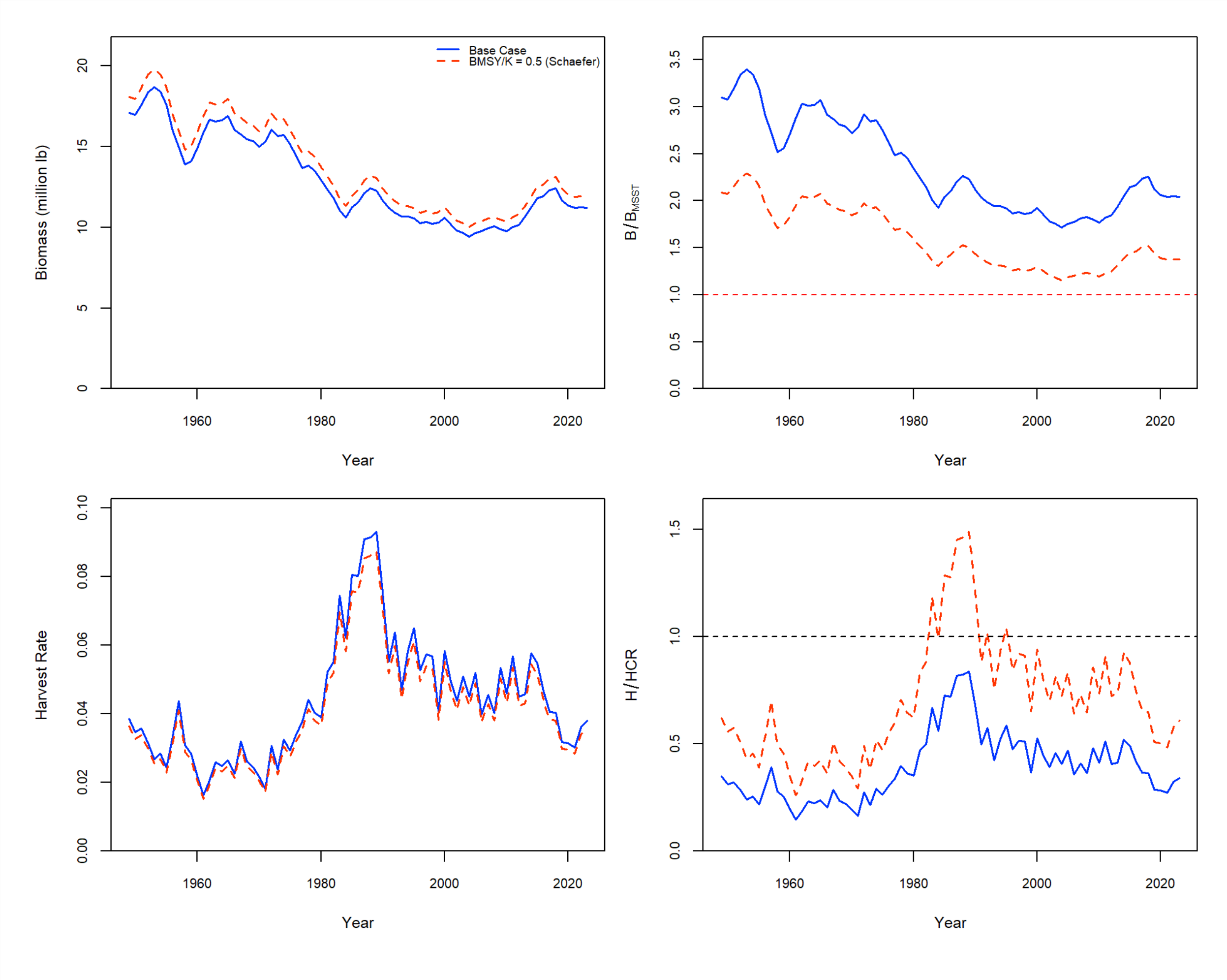


Figure 41. Estimated median exploitable biomass (million pounds), B/B_{MSST}, harvest rate, and H/H_{CR} for the base model (B_{MSY}/K = 0.329, solid blue line) and a Schaefer model (B_{MSY}/K = 0.5, red dashed line).

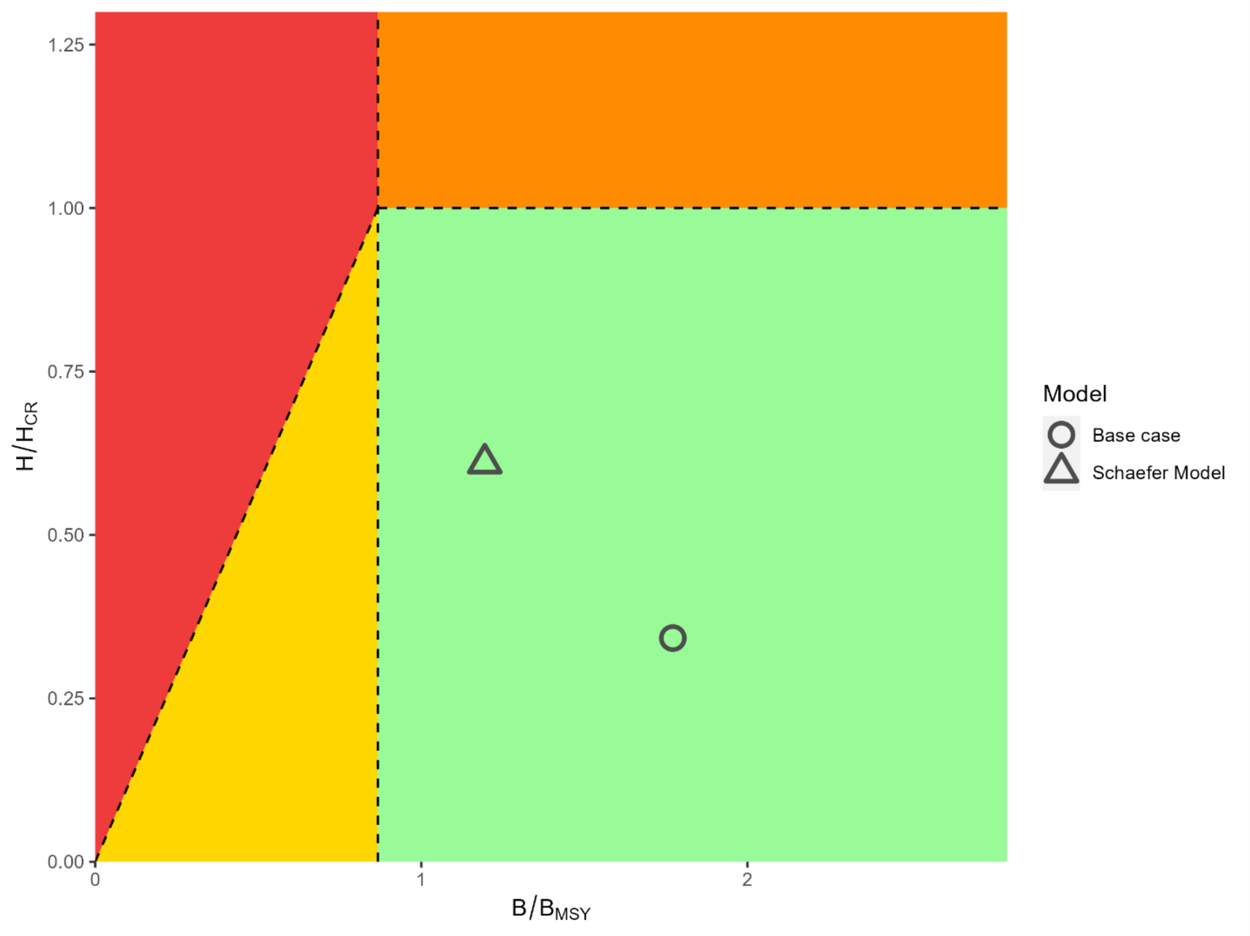


Figure 42. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for the sensitivity run testing a Schaefer model (B_{MSY}/K=0.5, triangle) and the base case (B_{MSY}/K=0.329, circle).

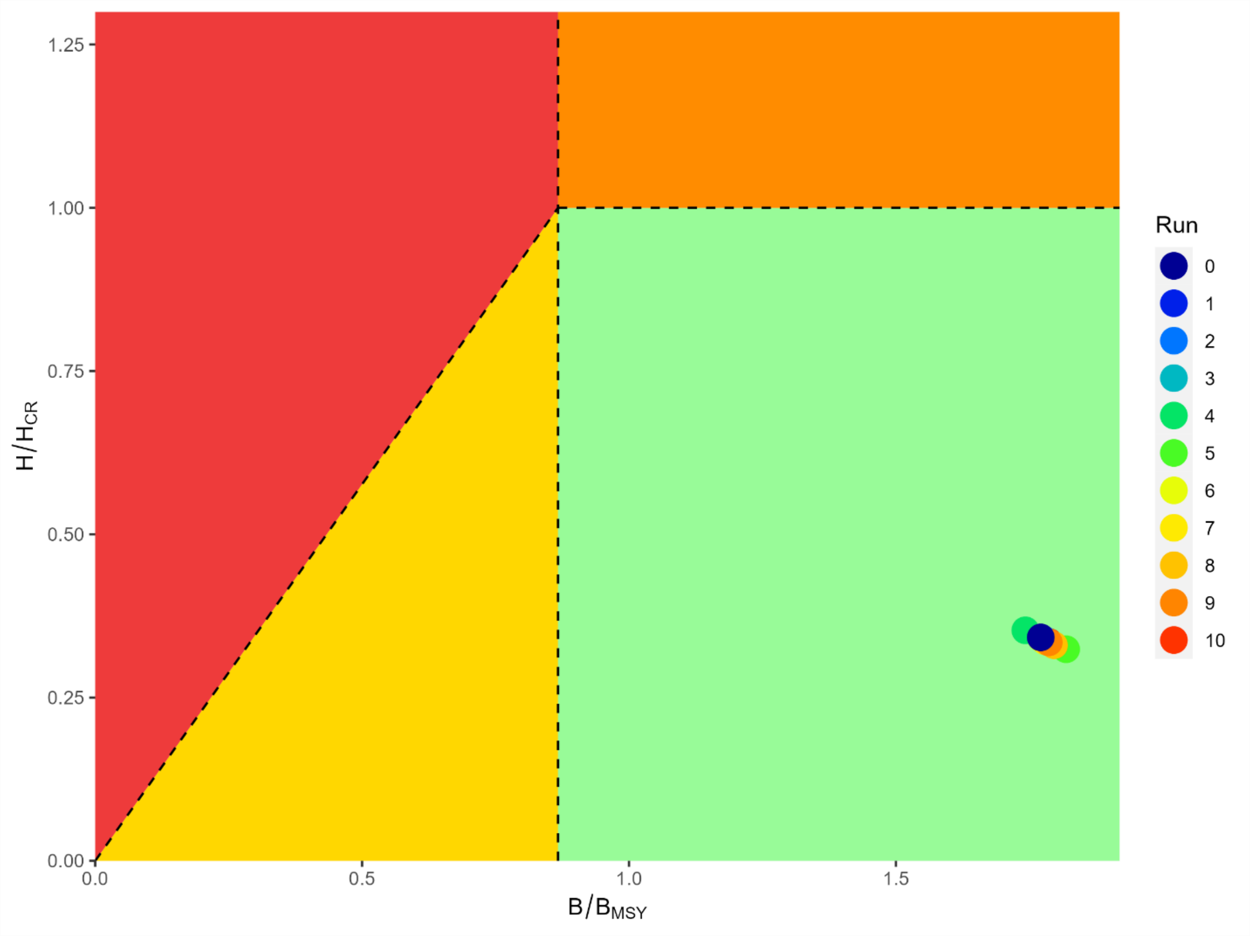


Figure 43. Kobe plot for terminal year stock status for the Deep 7 bottomfish complex in the main Hawaiian Islands for sensitivity runs testing initial conditions. The colors represent the base model (Run 0) and ten alternate initial conditions runs (1–10).

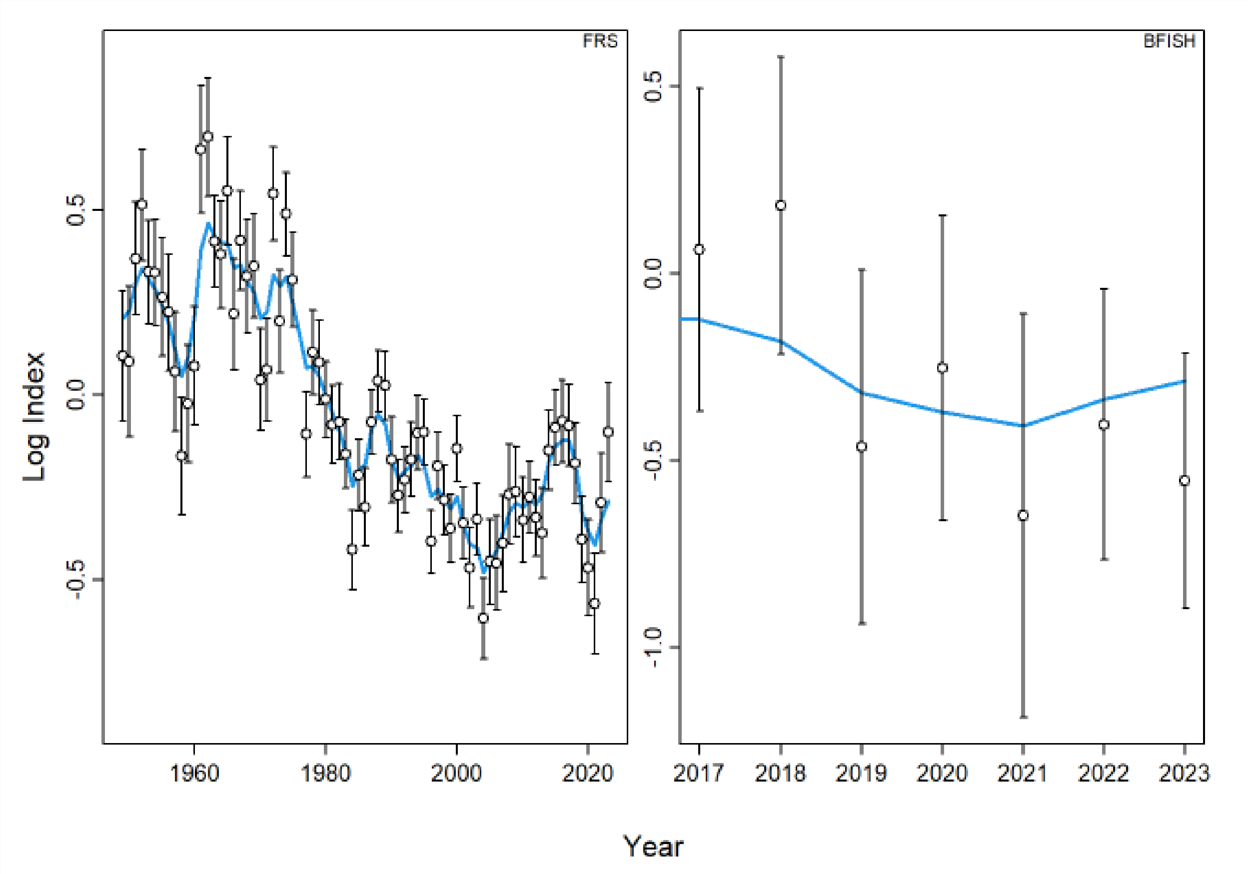


Figure 44. Fits of the FRS CPUE and BFISH indices of abundance for the ‘ōpakapaka production model on the log scale. Open circles are observations, vertical lines are the associated log standard error, and the blue line is the expected values from the model fit.

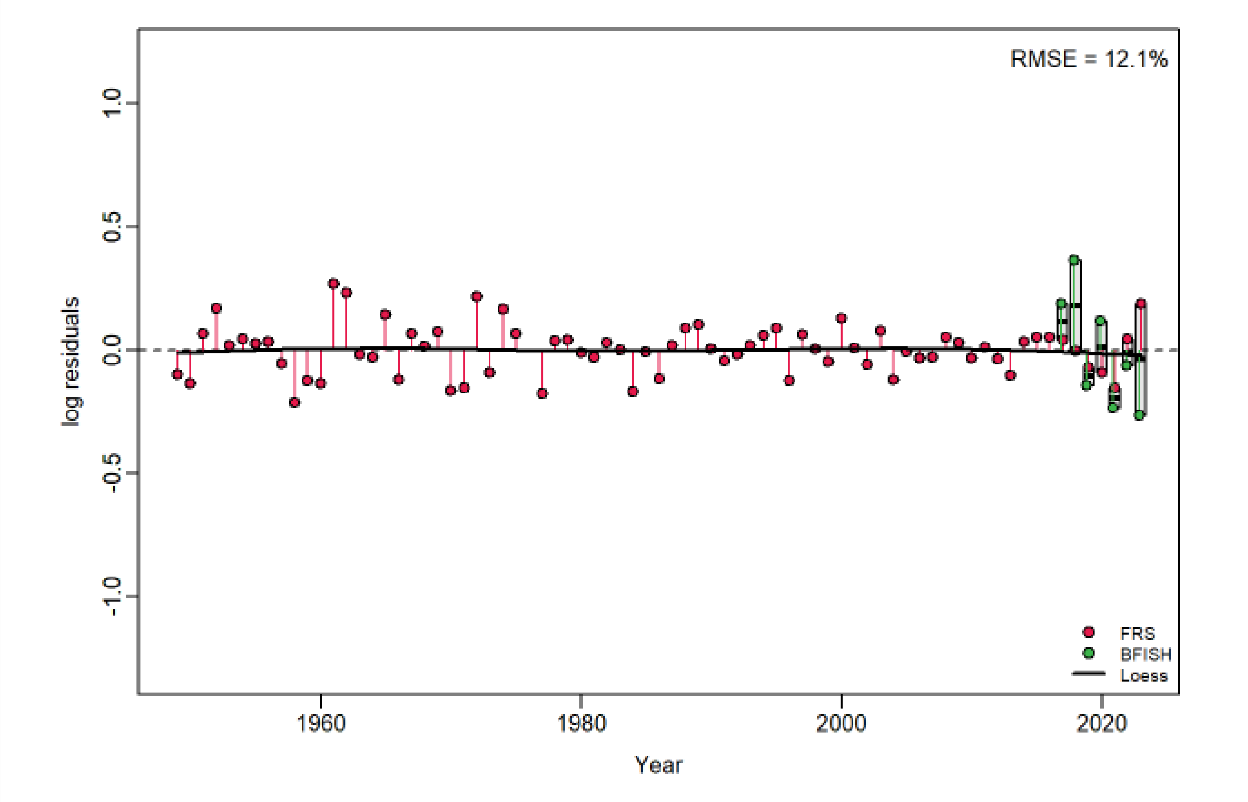


Figure 45. Residuals for FRS and BFISH indices of abundance and combined RMSE (12.1%) for the ‘ōpakapaka production model.

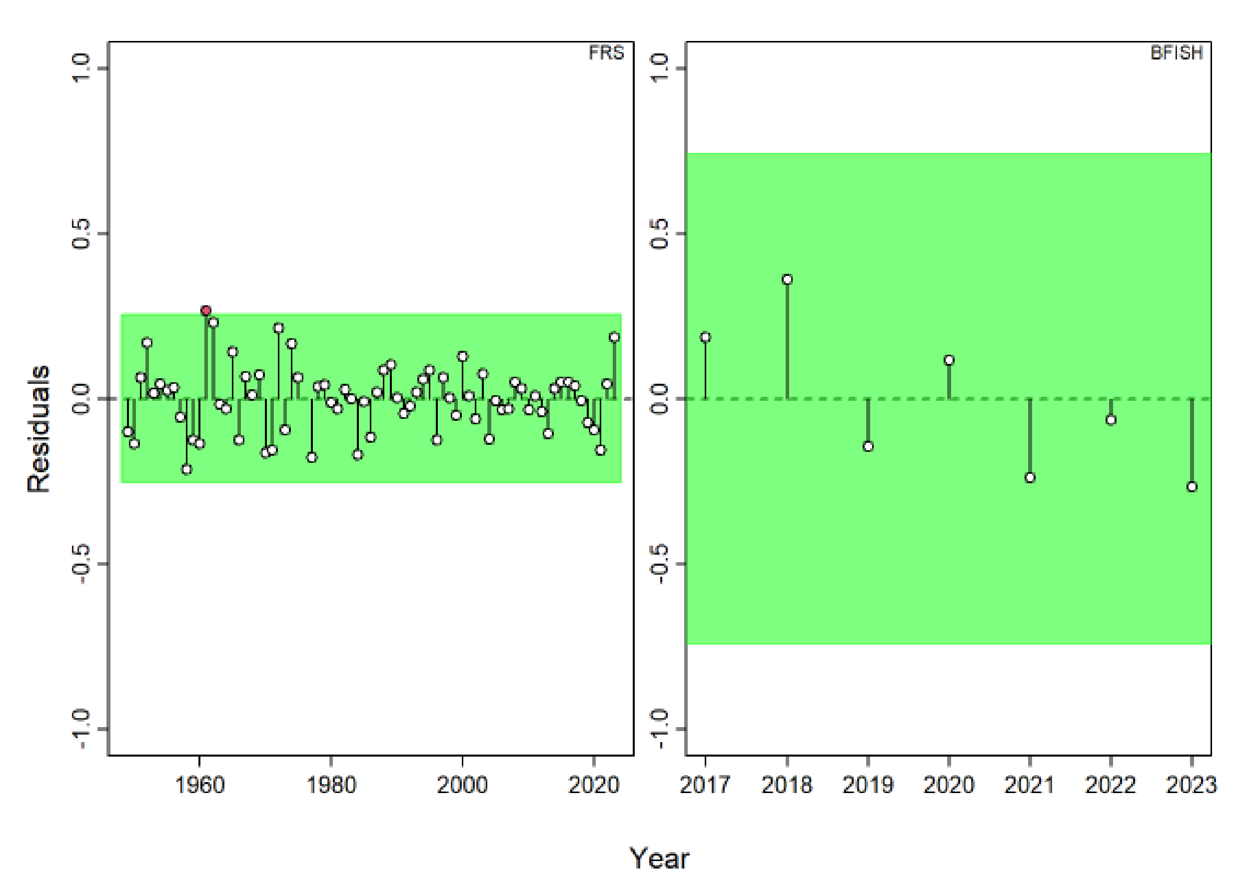


Figure 46. Runs test for FRS CPUE (left) and BFISH index of abundance (right) for the ʻōpakapaka production model. One residual from FRS CPUE in 1962 was greater than three standard deviations away from the mean as indicated by the red circle.

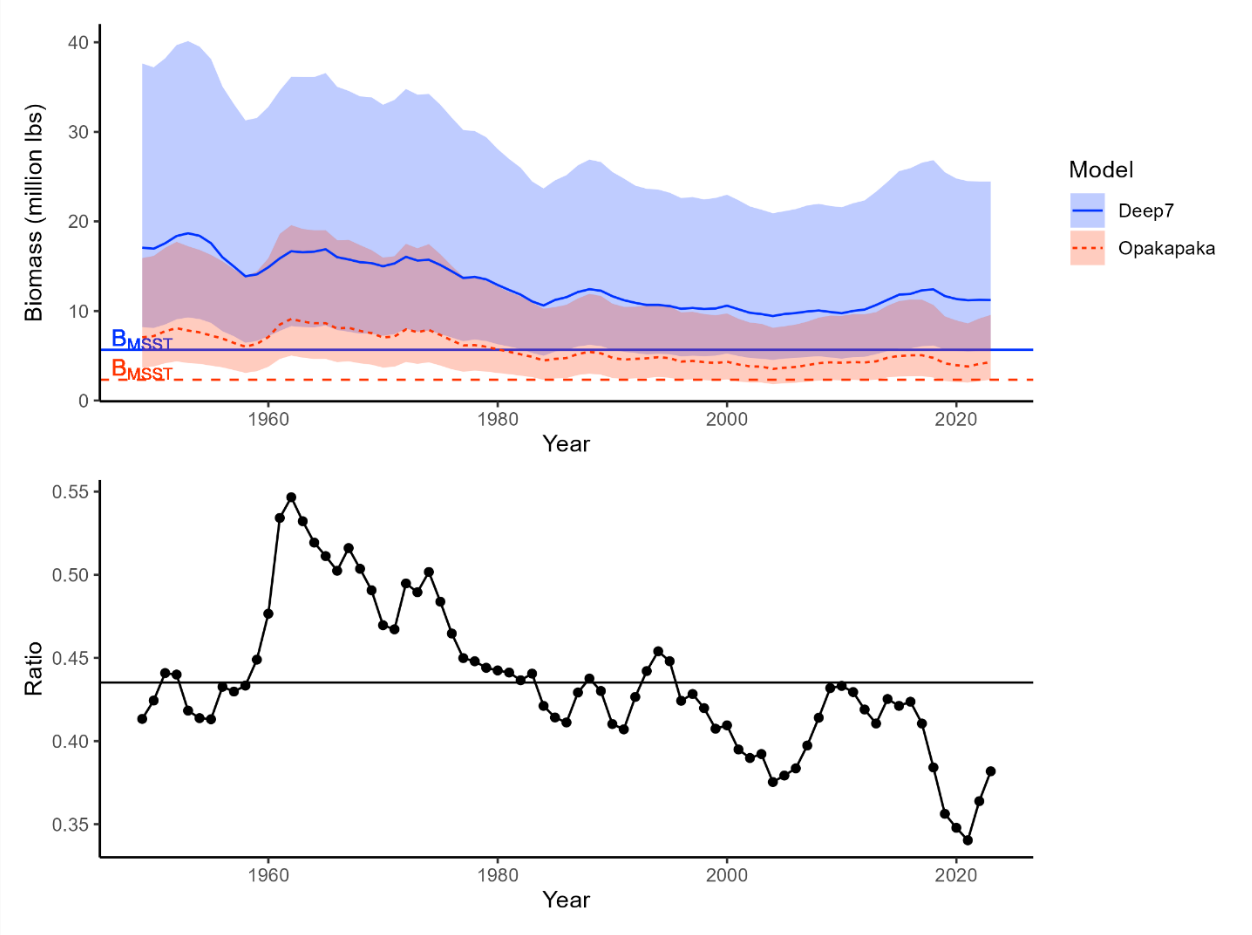


Figure 47. Biomass comparison between the Deep 7 complex production model and the ʻōpakapaka production model. Top panel: Posterior median exploitable biomass estimates and 95% credible intervals for Deep 7 (blue solid line) and ʻōpakapaka (red dashed line). Bottom panel: Ratio (black line with points) and average ratio (0.44, horizontal line) of the posterior median exploitable biomass from the ʻōpakapaka production model to the posterior median biomass from the Deep 7 production model.

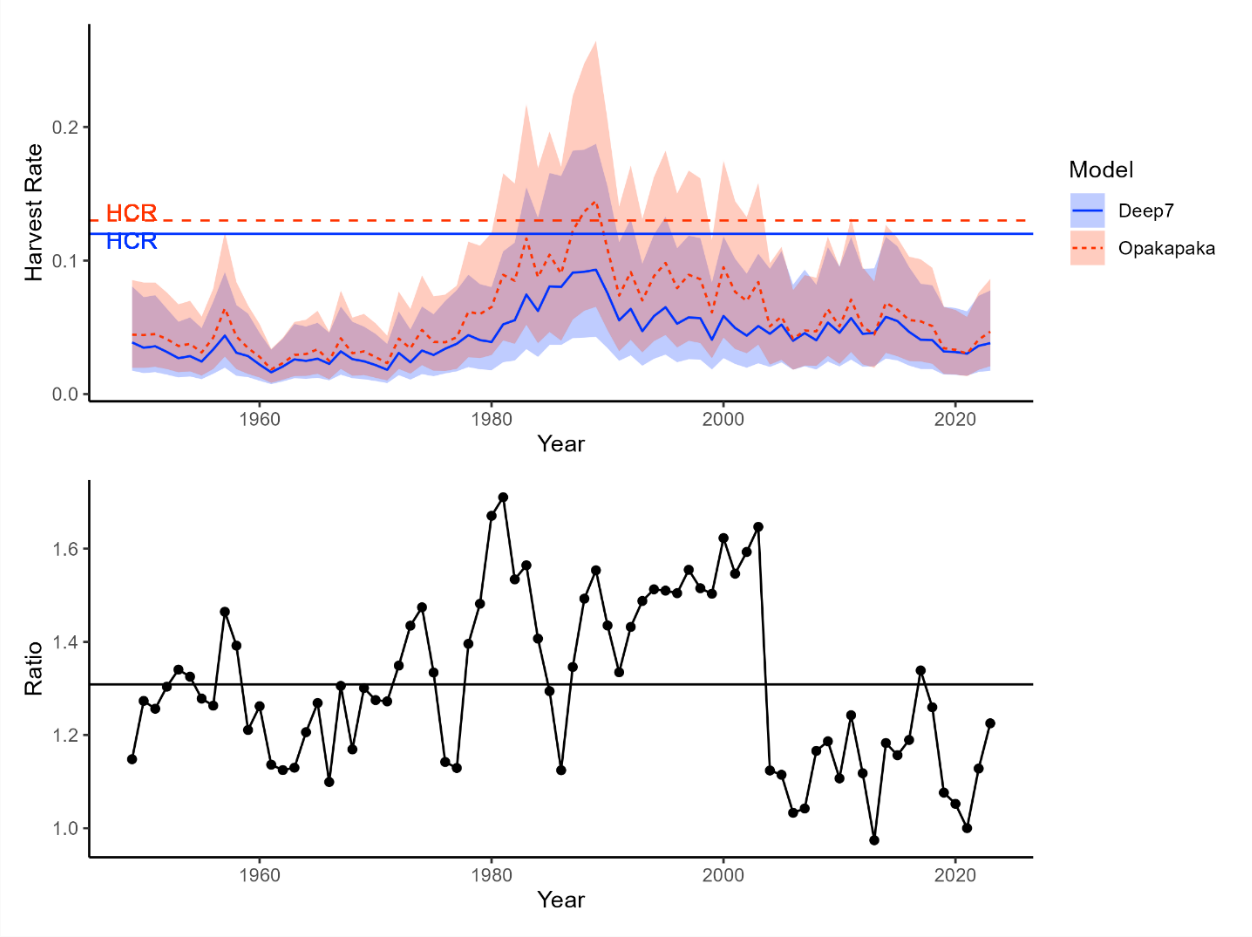


Figure 48. Harvest rate comparison between the Deep 7 complex production model and the ‘ōpakapaka production model. Top panel: Posterior median harvest rate estimates and 95% credible intervals for Deep 7 (blue solid line) and ‘ōpakapaka (red dashed line). Bottom panel: Ratio (black line with points) and average ratio (1.3, horizontal line) of the posterior median harvest rate from the ‘ōpakapaka production model to the posterior median harvest rate from the Deep 7 production model.

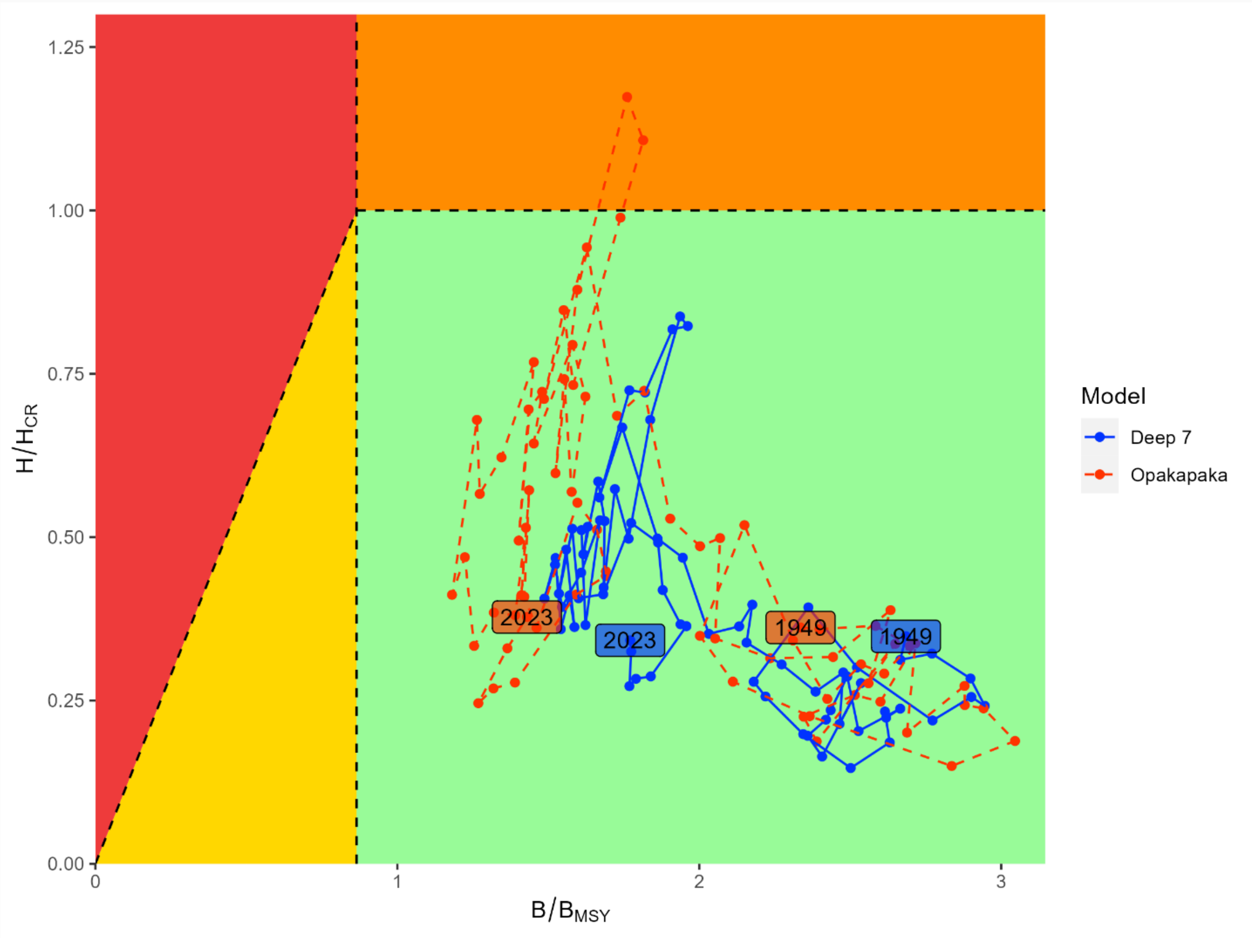


Figure 49. Status of ʻōpakapaka, as based on the ʻōpakapaka only model (red line), compared to the status estimated from the model of the Deep 7 bottomfish complex (blue line) for the main Hawaiian Islands.

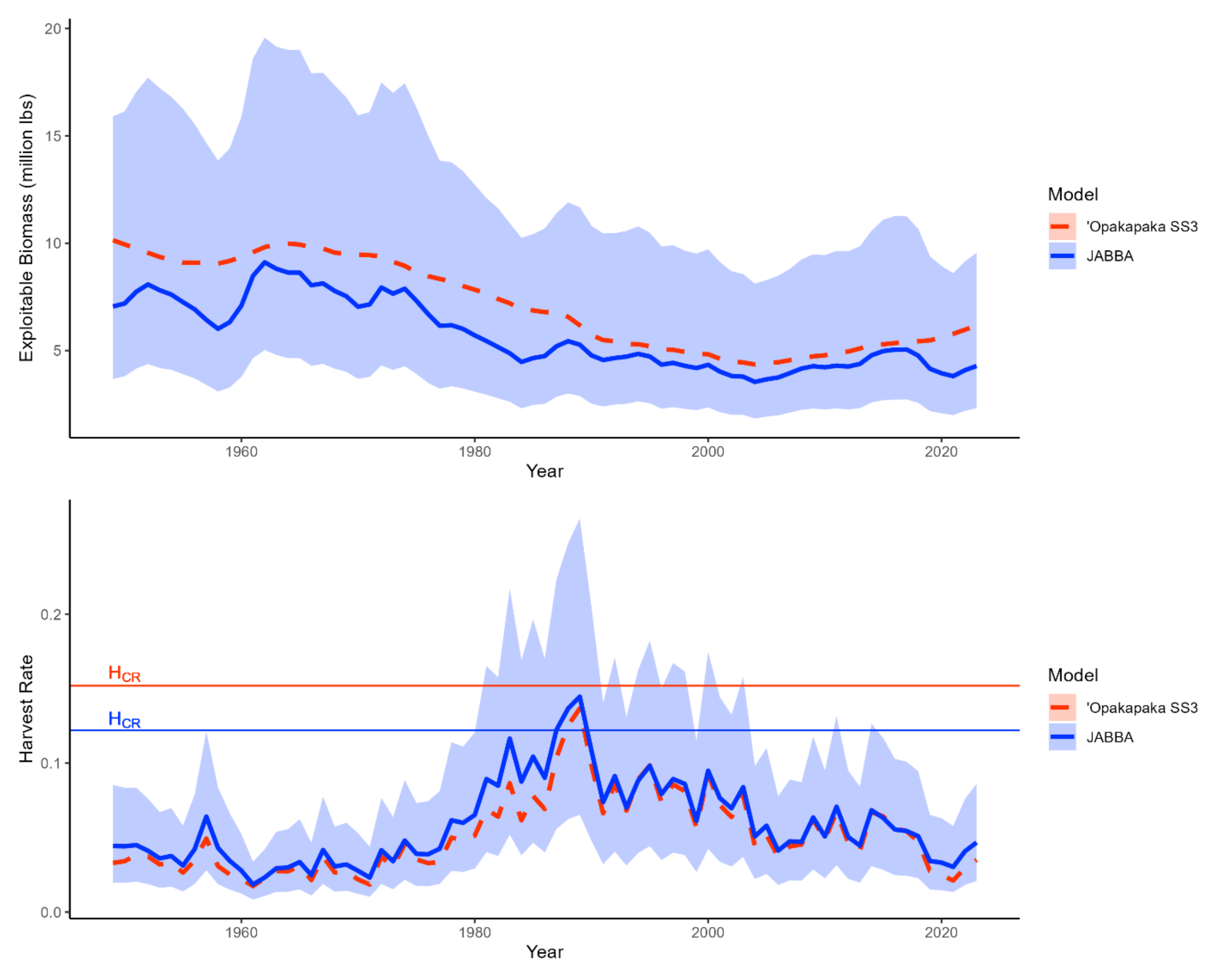


Figure 50. Exploitable biomass (top panel) and harvest rate (bottom panel) comparisons between the 'ōpakapaka single-species JABBA model (blue, solid line) and the stock synthesis model (red, dashed line). The solid horizontal lines in the bottom panel represent H_{MSY} for the stock synthesis model (top, red line) and the JABBA model (bottom, blue line).

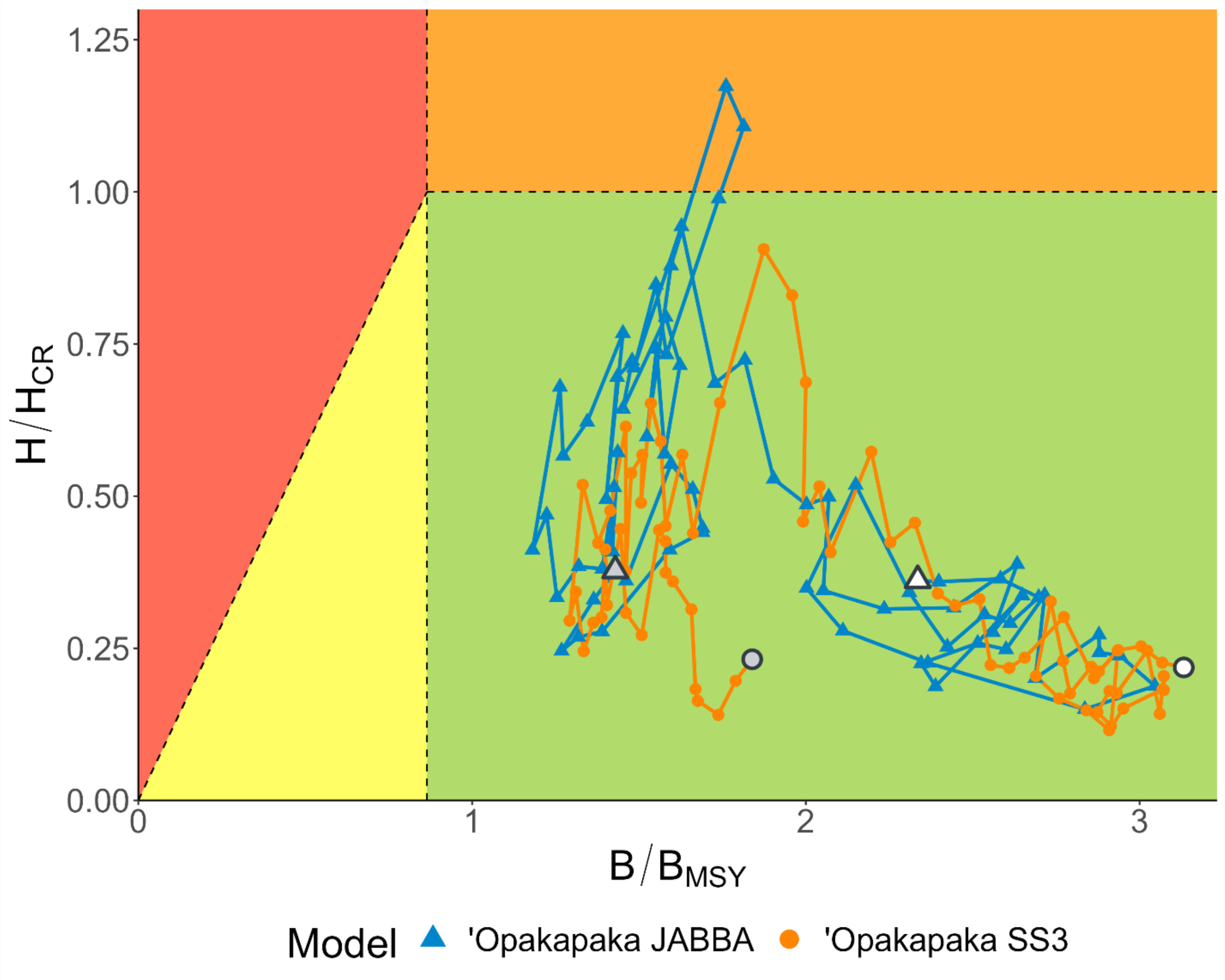


Figure 51. Time series of stock status of ʻōpakapaka, as estimated by the JABBA production model (blue triangles) and the age-structured Stock Synthesis (SS) model (orange circles). Initial and terminal year statuses are represented by the white and grey points respectively, where triangles represent the JABBA model and circles represent the SS model.

Appendix A. WPSAR Panel recommendations for 2018 Benchmark and 2021 Update stock assessments from panel summary reports

Review of 2018 Benchmark Stock Assessment

The review of the 2018 Benchmark Stock Assessment (Martell et al. 2017) produced several “medium-term” recommendations to be completed over a 1–5+ year time period. However, WPSAR framework specifies that changes to methodology are to only be incorporated in benchmark stock assessments, the next of which will be reviewed through WPSAR in 2023. Their recommendations are listed below.

- Continue to implement the fisheries-independent survey on an annual basis, ensuring there are paired sampling blocks where both survey gears (i.e., research fishing and camera) are deployed to monitor changes in gear conversion factor used to scale the two survey gears.
- Continue with field work and research and development for the bottom camera work. This work will be instrumental for monitoring bottomfish restricted fishing areas (BRFAs).
- Join vessel size database with FRS data to assign each record to a vessel class (length on water or horsepower), leading to a better basis for e.g., filtering single-day vs. multi-day trips.
- Explore alternative classification models (e.g., regression trees, PCA, GAMs) for identifying trips targeting ‘ōpakapaka.
- Increase effort to better estimate unreported catches in future years. Uncertainty in the unreported catch is probably the single most important missing piece of data that informs overall population scale.
- Examine the residual patterns in the commercial CPUE. Determine whether they are sensible, or if there is still concern that multi day trips are not being filtered out correctly further biasing the CPUE index.
- Continue with the life history work (growth, maturity) for the remaining species in the Deep 7 complex.
- More fully examine year-area interactions in the CPUE data (e.g., contraction and expansion of the fishery, displacement of effort due to BRFAs) or other possible covariates (e.g., decadal scale environmental variables, currents).
- Continue with the CPUE standardization efforts with consideration for any new information.
- The panel discussed creating another data set where the CPUE series is continuous from 1949–2016. In this case, use the catch per day (by ignoring the hours fished) for the post-2002 data such that there is only one overall q for fisheries-dependent CPUE data.

- Explore alternative parameterization of the surplus production model that allows the B_{MSY}/K ratio to fall below 0.378.
- Explore alternative models that make use of mean weight data (e.g., delay-difference model), especially for the ‘ōpakapaka assessment given the recent life history information.
- Explore a history of gear changes over time and how the advent of modern reels, increased vessel speed, and loss of institutional memory has affected fishing power over time. For example, what proportion of the fishers land 90% of the total reported catch, and has this proportion decreased over time?
- Continue to address the life history differences in the Deep 7 that could result in changes in the prior distribution for *r* in this stock complex.
- Explore alternative software tools for conducting this stock assessment. Our recommendation is not restricted to Bayesian methods only.

Review of 2021 Stock Assessment Update

High Priority

Data workshops and stakeholder connections. Maintain direct communications with fishers about stock assessment activities. Conduct data workshops with the fishing community to develop collaborative contributions to the data and methods included in the next benchmark stock assessment.

Unreported catch. Unreported catch is a significant source of uncertainty. Continued collaboration between NOAA Fisheries, the Council, and the fishing community should be pursued to improve the collection of data describing non-commercial catch. These activities could include improvements to the Marine Recreational Information Program (MRIP), the federal non-commercial license program, and pilot programs to directly collect catch and effort data from non-commercial fishermen.

Medium Priority

Complex and single-species assessments. Continue to present both the Deep 7 complex and single-species assessments for important species with sufficient information (e.g., ‘ōpakapaka) in the next benchmark assessment. We recommend further data collection and life history studies for other species in the complex to facilitate stock assessments.

Fishery-independent survey methods. Perform research activities to provide improved empirical estimates of the survey area for the stereo-video method used in the fishery-independent survey. Species-specific issues should be investigated regarding diurnal schooling characteristics and vertical behavior in relation to the orientation and field of view of the camera system. The collection of life history and behavior data from Deep 7 species useful for improving fishery-independent survey data should be strongly promoted.

CPUE standardization. Explore the inclusion of additional factors that may impact Deep 7 CPUE identified at previous and future workshops on data standardization in future benchmark stock assessments. Interact with fishers and the scientific community for additional ideas to improve the standardization process. Where data on potentially important factors are lacking, make recommendations to appropriate agencies to conduct research and collect these data.

Low Priority

Software. There is a minor limitation in propagating process errors in the stock projections. There may be other modelling platforms (transition from JAGS, Stan, etc.) that more suitably capture the process error component. However, relative to the magnitude in the errors associated with the reported catch, this additional error may be infinitesimal.

Appendix B. Ratios of unreported catch to reported commercial catch for the main Hawaiian Islands Deep 7 bottomfish during 1948-2003

A note on the terminology used for non-commercial catch: the term "unreported catch" used in Courtney and Brodziak (2011) would be equivalent to "recreational catch" in Martell et al. (2006). In this report, we did not change the terms used in other reports, such as "recreational catch" in Martell et al. (2006), "unreported catch" in Courtney and Brodziak (2011), and "non-commercial catch" in Ma et al. (2023). The "non-commercial catch" in Ma et al. (2023) was intended to represent the catch not captured in the commercial fishing reports.

Total catch for the Deep 7 bottomfish fishery is composed of both commercial (reported) catch and non-commercial catch. In Hawaiʻi, fishers selling fish are required to have a commercial marine license and report the catch in commercial fishing reports. Catch estimates from the Hawaiʻi Marine Recreational Fishing Survey (HMRFS) are available for years 2003–2023. For non-commercial (i.e., non-sold) catch estimation with HMRFS survey data, the catch claimed to be sold in the survey data was excluded by Ma et al. (2023). The sum of the reported commercial catch and the non-commercial catch estimates from HMRFS can be used to define total fishing removal during 2003–2023. For the period 1948–2003, the non-commercial catch has been estimated using ratios of non-commercial catch to commercial catch based on available studies (Courtney and Brodziak 2011).

In the recent Deep 7 bottomfish stock assessments, the baseline non-commercial catch scenario for years prior to implementation of HMRFS (i.e., 2002 and earlier) was based on the species-specific ratios of recreational catches to commercial catches in 1990 derived by Martell et al. (2006). Courtney and Brodziak (2011) used the 1990 ratios (U_{1990}) derived by Martell et al. (2006) as ratios of *total unreported catch to reported commercial catch* in 1990.

The U_{1990} ratios for individual species were carried backward to 1948 and forward to 1997. The ratio of noncommercial to commercial catch from 2002 onward was informed by a subsequent study (Lamson et al. 2007) averaged with HMRFS estimates in 2004 and 2005 (Courtney and Brodziak 2011). The ratios for years between 1997 and 2002 were linearly interpolated. Martell et al. (2006) derived the species-specific ratios of recreational catches to commercial catches in 1990 utilizing the survey results in Hamm and Lum (1992). The small-boat fisheries survey in Hamm and Lum (1992) was conducted on Oahu from March 1990 to May 1991.

During the catch interview, surveyors recorded catch disposition (% keep, % sell, and % other) of aggregated catch from a fishing method, such as bottomfishing or trolling.

For the estimated catch by bottomfishing, 34% of the catch was “sold”. Thus, 66% (= 100%–34%) of the bottomfish catch was regarded as “non-sold”. Martell et al. (2006) used 1.94 (66% for “non-sold” divided by 34% for “sold”) as the ratio of recreational catch to commercial catch. The sold proportion estimated from the 1990 survey was believed to be conservative by Hamm and Lum (1992), i.e., the non-sold proportion may be overestimated. Courtney and Brodziak (2011) interpreted the ratio 1.94 as the ratio of total unreported catch to *reported commercial catch* (see the second paragraph on page 6 of Courtney and Brodziak 2011).

Unreported ratios for 1990

Taking 1.94 as the ratio of unreported bottomfish catch to reported commercial bottomfish catch as a whole, we followed the approach by Martell et al. (2006) to derive unreported ratios for individual Bottomfish management unit species (BMUS). The total unreported bottomfish for the main Hawaiian Islands was scaled up by the reported commercial catch in 1990 (271.45 metric tons for BMUS in the main Hawaiian Islands, Table B1) times 1.94. T

he proportion of expanded catch by taxa for bottomfish surveyed by Hamm and Lum (1992) was used to allocate the total unreported bottomfish catch to individual BMUS species (Table B1).The unreported ratio for a species is the ratio of the unreported catch to the reported commercial catch for the species (Figure B1 and Table B1). A simplification to the calculation of the unreported ratio for a species can be found in Equation B1 in Appendix B.

Table B1. The species-specific unreported catch estimates (metric tons) and the ratios of unreported catch to reported commercial catch in 1990. Following Martell et al. (2006), proportions of 0.3, 0.01, and 0.18 were used to allocate the reported miscellaneous jacks *Carangidae* to the reported commercial catch for white ulua, black ulua, and butaguchi.

Species	Proportion (H&L 1992)	Reported commercial catch	Unreported catch	Unreported ratio
Ehu	0.0200	16.12	10.53	0.65
Gindai	0.0003	1.53	0.14	0.09
Hapu’upu’u	0.0080	7.62	4.21	0.55
Kahala	0.0570	9.25	30.02	3.25
Kalekale	0.0003	8.98	0.16	0.02
Lehi	0.0004	9.72	0.21	0.02
Onaga	0.0430	51.54	22.64	0.44
‘Ōpakapaka	0.2290	68.69	120.59	1.76
Ta’ape	0.4250	23.20	223.81	9.65
Uku	0.1170	48.64	61.61	1.27
White ulua	0.0760	16.02	40.02	2.50
Black ulua	0.0010	0.36	0.53	1.46
Butaguchi	0.0220	9.78	11.59	1.18
Total BMUS	0.9990	271.45	526.06	1.94
(Sources)	Table 16 in Martell et al. 2006	Reported data for MHI in 1990	-	-

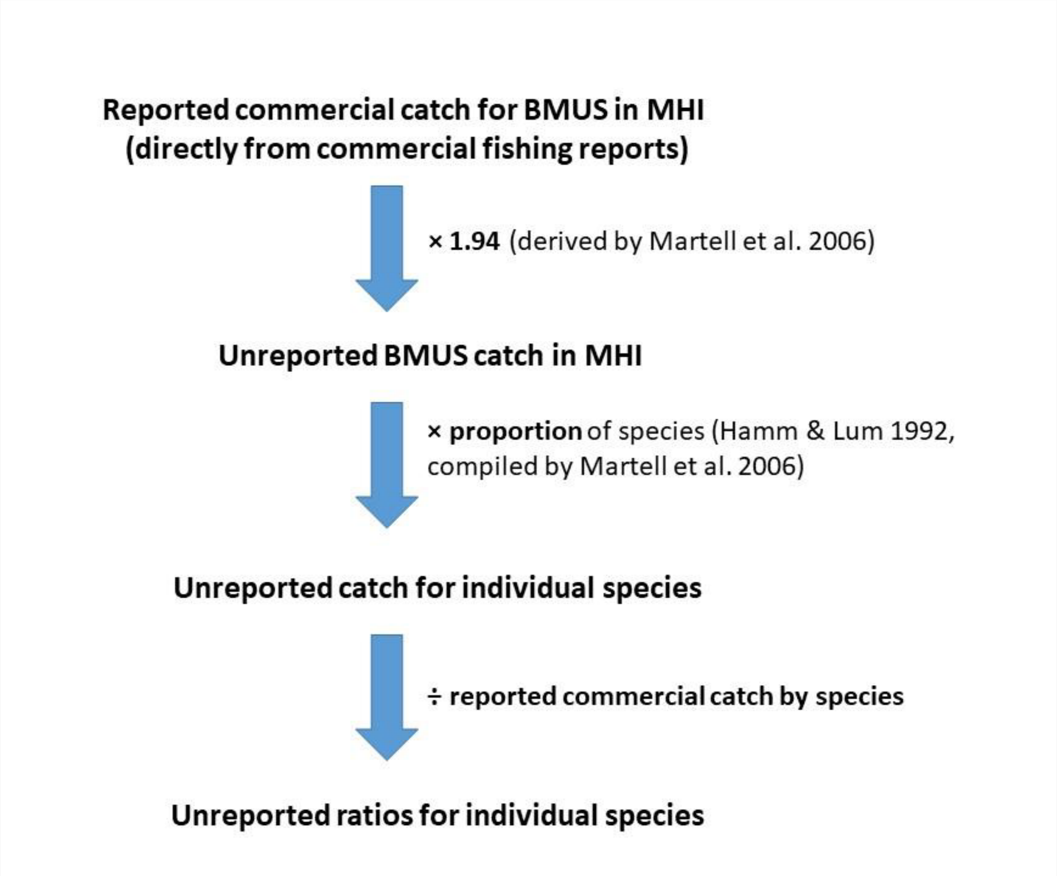


Figure B1. Flow chart for estimating the unreported ratios by species

Comparisons with ratios of recreational catch to commercial catch in 1990 by Martell et al.

Our estimated unreported ratios for individual BMUS species are smaller than the ratios derived by Martell et al. (2006) (Table B2). The total recreational BMUS catch of 853.66 metric tons in Martell et al. (2006) would suggest that a commercial catch of 440 tons (= 853.66/1.94) was used to scale up the total recreational catch.

The reported commercial catch for BMUS in 1990 was 436 tons in the entire Hawaiian Archipelago including the main Hawaiian Islands and the Northwestern Hawaiian Islands (Table 10 in Martell et al. 2006). However, because there is no recreational bottomfishing in the remote Northwestern Hawaiian Islands (Zeller et al. 2008) we believe the commercial catch from the Northwestern Hawaiian Islands may not need to be included for scaling up recreational catch in the main Hawaiian Islands. Based on the estimated recreational catch and the ratios of recreational catch to commercial catch for individual BMUS species from Martell et al. (2006), the implied commercial catch by species was back calculated (the second last column in Table B2).

The back-calculated commercial catches (with a total of 279.5 tons) are similar to the reported commercial catch in the main Hawaiian Islands (cf. the third column in Table B1). It appears that a commercial catch close to the reported commercial catch in the Hawaiian Archipelago (440 tons) was used to scale up the total recreational catch in Martell et al. (2006). However, the reported commercial catch in the main Hawaiian Islands was used to estimate the ratio of recreational catch to commercial catch for individual species (Equation B2).

The ratio of the sum of species-specific recreational catch (third column in Table B2) to the sum of implied commercial catch (fourth column in Table B2) is 3.05 and is larger than the scaling factor 1.94, indicating that the species-specific ratios of recreational catch to commercial catch in Martell et al. (2006) are likely overestimated if applied to the main Hawaiian Islands.

Table B2. Recreational catch (tons), ratio of recreational catch to commercial catch, and implied commercial catch by species (= recreational catch divided by the ratio of recreational to commercial catch) in Martell et al. (2006). The unreported ratios from Table 1 are included for comparison.

Species	Recreational catch (tons)	Ratio (Rec/comm)	Implied commercial catch (tons)	Unreported ratio
Ehu	17.27	1.11	15.56	0.65
Gindai	0.23	0.15	1.53	0.09
Hapu'upu'u	6.92	1.02	6.78	0.55
Kahala	48.31	5.22	9.25	3.25
Kalekale	0.28	0.03	9.33	0.02
Lehi	0.35	0.04	8.75	0.02
Onaga	36.54	0.73	50.05	0.44
'Ōpakapaka	195.86	2.87	68.24	1.76
Ta'ape	362.41	15.63	23.19	9.65
Uku	100.26	2.27	44.17	1.27
White ulua	65.19	2.41	27.05	2.50
Black ulua	1.05	1.24	0.85	1.46
Butaguchi	18.98	1.29	14.71	1.18
Total BMUS	853.65	(3.05)	279.48	1.94
(Sources)	Table 17 in Martell et al. 2006	Table 18 in Martell et al. 2006		Table 1 in this report

Unreported ratios for 1948–2003

Instead of using a single species-specific unreported ratio from 1990 for all years in 1948–2003, the species proportions in the reported commercial catch were calculated for each year in 1948–2003. Ta’ape was introduced to Hawai’i in the 1960s and the commercial catch of ta‘ape was absent or minimal prior to 1975. During 1975–2003, ta’ape comprised a substantial proportion of the commercial BMUS catch. Thus, the average species proportions in the commercial catch were calculated among years within two periods: during 1948–1974 (when ta’ape catch was absent or minimal) and 1975–2003 (when kahala catch was lower but ta’ape catch was large).

The unreported ratios were estimated as 1.94 times the ratio of species proportion in recreational catch to species proportion in commercial catch for different periods as done in the previous calculations (see Equation B1). The resulting unreported catch for the Deep-7 species from the updated approach compared to the previous approach (i.e., Martell et al. 2006) are depicted in Figure B2. The ratio of unreported to reported Deep 7 catch exhibits a step change when transitioning from the 1990 ratios (U_{1990}) derived by Martell et al. (2006) to the ratios informed by Lamson et al. 2007 and the HMRFS estimates in 2004 and 2005 (Figure B3). The ratios from this study are more similar to those informed by Lamson et al. (2007) and HMRFS estimates and provide noncommercial catches that are more similar to the HMRFS estimates from 2003 to 2022 (Ma et al. 2023).

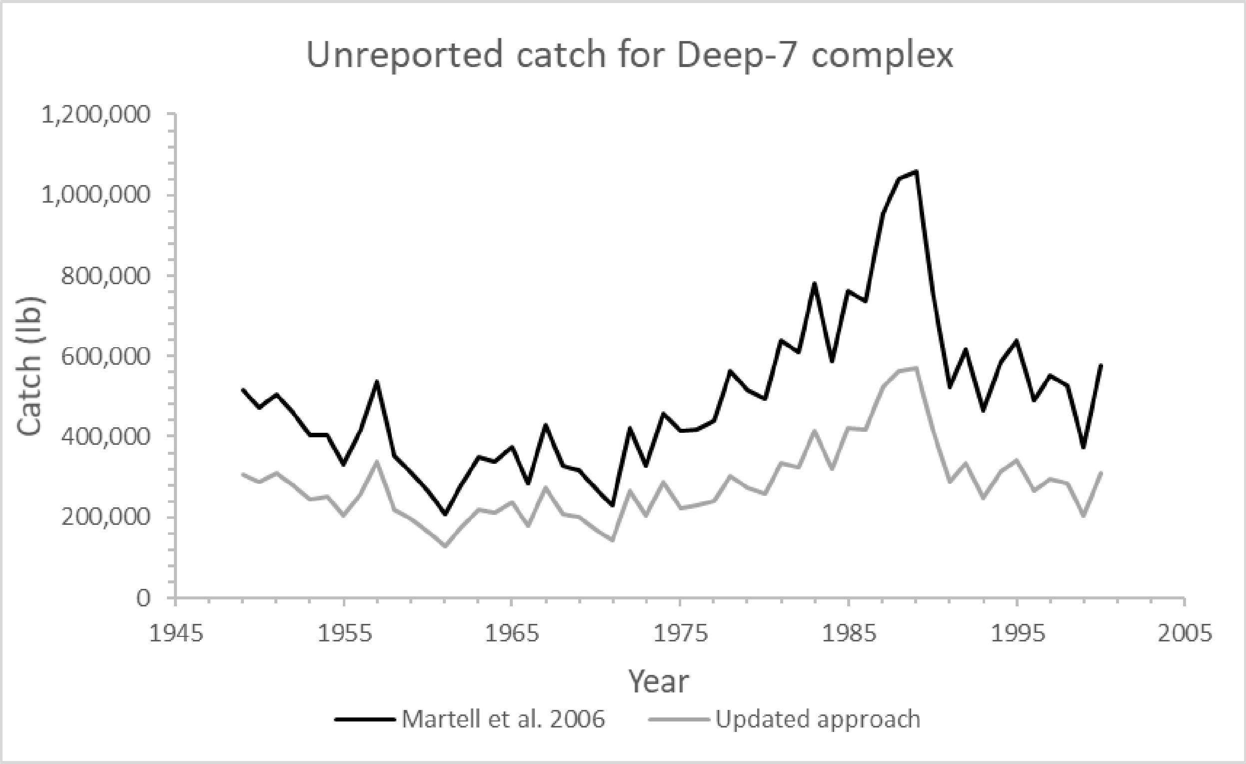


Figure B2. Total unreported catch from 1949 to 2000 summed among Deep 7 species using the updated approach (described in the current report) and the previous approach by Martell et al. (2006).

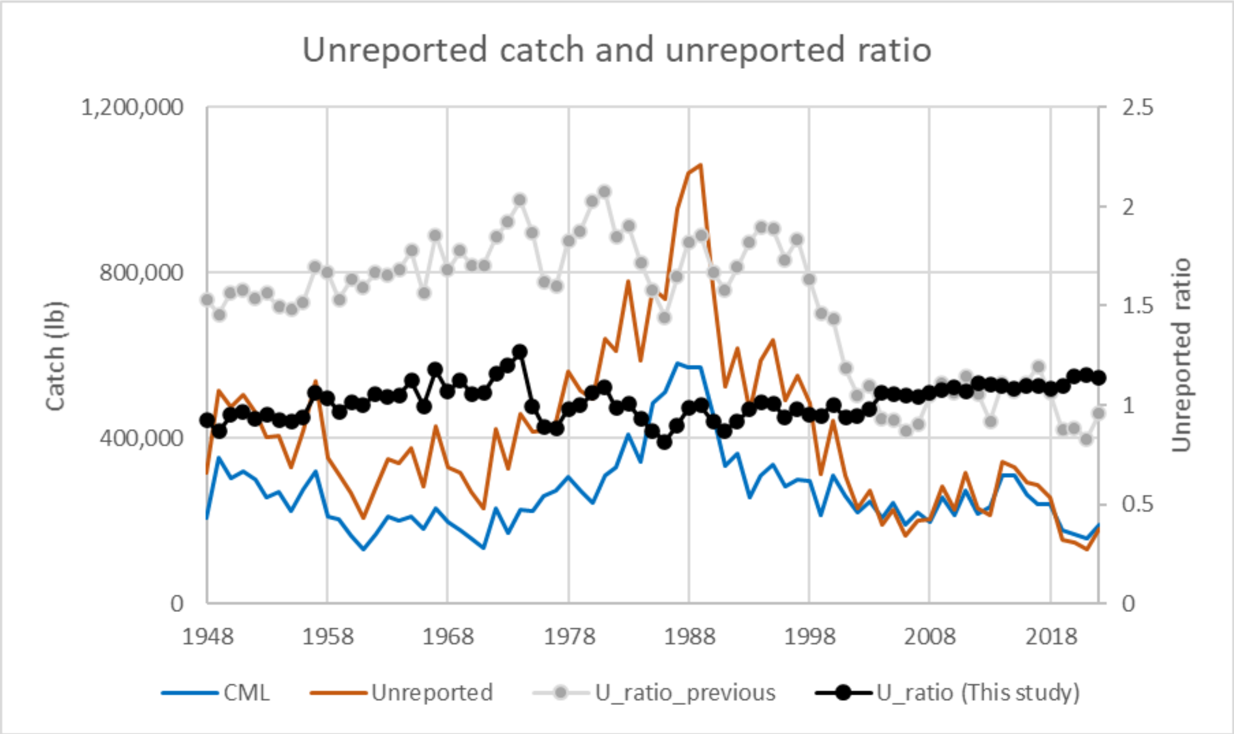


Figure B3. Deep 7 catch from 1948 to 2022 reported by commercial marine license holders (CML; blue), unreported Deep 7 catch estimated from species-specific ratios used in previous assessments (orange; Courtney and Brodziak 2011), the overall ratio of unreported to reported catch for Deep 7 species combined using the previous ratios (gray), and the overall ratio of unreported to reported catch for Deep 7 species combined using ratios determined in this study (black). The ratios in Courtney and Brodziak (2011) depicted above (gray) used the Martell et al. (2006) estimate for 1948–1997 and the average of Lamson et al. (2007) and HMRFS estimates (from 2004 and 2005) for years after 2001. The black points from 2004–2022 are based on the mean ratios for individual species from Ma et al. (2023). Note that catch from 1948 was not reported for the full year.

Equations

Equation B1: a simplified method to calculate the unreported ratio using species proportion information and the scaling factor 1.94.

$$R_S = \frac{C_{n,S}}{C_{c,S}} = \frac{P_{survey,S} \cdot C_{n,BMUS}}{P_{c,S} \cdot C_{c,BMUS}} = \frac{P_{survey,S} \cdot 1.94 \cdot C_{c,BMUS}}{P_{c,S} \cdot C_{c,BMUS}} = \frac{1.94 \cdot P_{survey,S}}{P_{c,S}}$$

Equation B2: a possible method used by Martell et al. 2006.

$$R_S = \frac{P_{survey,S} \cdot C_{n,Archipelagic\ BMUS}}{P_{c,S} \cdot C_{c,BMUS}} = \frac{P_{survey,S} \cdot 1.94 \cdot C_{c,Archipelagic\ BMUS}}{P_{c,S} \cdot C_{c,BMUS}}$$
$$> \frac{1.94 \cdot P_{survey,S}}{P_{c,S}} \quad because\ of \quad \frac{C_{c,Archipelagic\ BMUS}}{C_{c,BMUS}} > 1$$

***Rs*:** Ratio of noncommercial (= unreported) to commercial catch of BMUS species *S* in the main Hawaiian Islands

$C_{c,\,S}$: Commercial (= reported) catch of BMUS species *S* in the main Hawaiian Islands

$C_{n,\,S}$: Noncommercial (unreported) catch of BMUS species *S* in the main Hawaiian Islands

$P_{survey,\,S}$: Proportion of BMUS species *S* in the main Hawaiian Islands survey (i.e., Hamm and Lum 1992)

$P_{c,\,S}$**:** Proportion of BMUS species *S* in commercial BMUS catch in the main Hawaiian Islands

$C_{c,\,BMUS}$: Total commercial catch of all BMUS species in the main Hawaiian Islands

$C_{n,\,BMUS}$: Total noncommercial catch of all BMUS species in the main Hawaiian Islands

$C_{c,\,Archipelagic\ BMUS}$: Total commercial catch of all BMUS species in the entire Hawaiian Archipelago including the Northwestern Hawaiian Islands

$C_{n,\,Archipelagic\ BMUS}$: Total noncommercial catch of all BMUS species in the entire Hawaiian Archipelago including the Northwestern Hawaiian Islands

Appendix C. Supplementary methods and results for ‘ōpakapaka production model

Table C1. Summary of log-likelihood results and reduction in Akaike Information Criterion (AIC) during model selection for the best-fit ‘ōpakapaka. Only Bernoulli and lognormal processes.

Process	Selected predictor	Delta	Log-likelihood	Number of parameters
Bernoulli				
	null	0	-39932.7	1
	+fisher	13923.2	-32970.1	2
	+year	593.4	-32600.4	76
	+area	1144.2	-31941.3	163
	+area:year	1248.9	-31315.8	164
	+quarter	411.8	-31106.9	167
Lognormal				
	null	0	-105551.1	1
	+fisher	21057.3	-95021.5	3
	+year	1333.6	-94281.7	76
	+area	2025.2	-93182.1	164
	+area:year	1374.5	-92493.80	165
	+quarter	913.7	-92033.9	168

Table C2. Annual index of standardized commercial CPUE and coefficient of variation (CV) for ʻōpakapaka.

Year	Estimated CPUE	CV	Year	Estimated CPUE	CV
1949	68.37	0.09	**1971**	65.89	0.07
1950	67.30	0.10	**1972**	106.03	0.06
1951	88.95	0.08	**1973**	75.08	0.07
1952	102.78	0.08	**1974**	100.28	0.06
1953	85.77	0.07	**1975**	84.04	0.06
1954	85.65	0.07	**1976**	N/A	N/A
1955	80.17	0.08	**1977**	55.34	0.06
1956	76.92	0.08	**1978**	68.98	0.06
1957	65.59	0.08	**1979**	67.18	0.06
1958	52.16	0.08	**1980**	60.81	0.05
1959	60.07	0.08	**1981**	56.80	0.05
1960	66.57	0.08	**1982**	57.24	0.05
1961	119.50	0.09	**1983**	52.49	0.05
1962	123.49	0.08	**1984**	40.47	0.06
1963	93.07	0.06	**1985**	49.57	0.05
1964	89.91	0.07	**1986**	45.46	0.05
1965	106.81	0.07	**1987**	57.15	0.04
1966	76.56	0.08	**1988**	63.86	0.04
1967	93.47	0.07	**1989**	63.08	0.05
1968	84.74	0.08	**1990**	51.68	0.06
1969	87.24	0.07	**1991**	46.85	0.05
1970	64.14	0.07	**1992**	48.95	0.05
1993	51.71	0.05	**2015**	56.35	0.05
1994	55.53	0.05	**2016**	57.34	0.06
1995	55.64	0.04	**2017**	56.66	0.06
1996	41.39	0.04	**2018**	51.13	0.06
1997	50.81	0.05	**2019**	41.63	0.06
1998	46.31	0.05	**2020**	38.60	0.07
1999	42.92	0.05	**2021**	34.99	0.07
2000	53.26	0.05	**2022**	46.00	0.07

Year	Estimated CPUE	CV	Year	Estimated CPUE	CV
2001	43.55	0.05	**2023**	55.65	0.07
2002	38.60	0.05	**-**	-	-
2003	44.01	0.05	**-**	-	-
2004	33.66	0.06	**-**	-	-
2005	39.20	0.06	**-**	-	-
2006	39.08	0.07	**-**	-	-
2007	41.19	0.07	**-**	-	-
2008	47.06	0.07	**-**	-	-
2009	47.36	0.06	**-**	-	-
2010	43.92	0.06	**-**	-	-
2011	46.69	0.05	**-**	-	-
2012	44.15	0.05	**-**	-	-
2013	42.36	0.06	**-**	-	-
2014	53.01	0.06	**-**	-	-

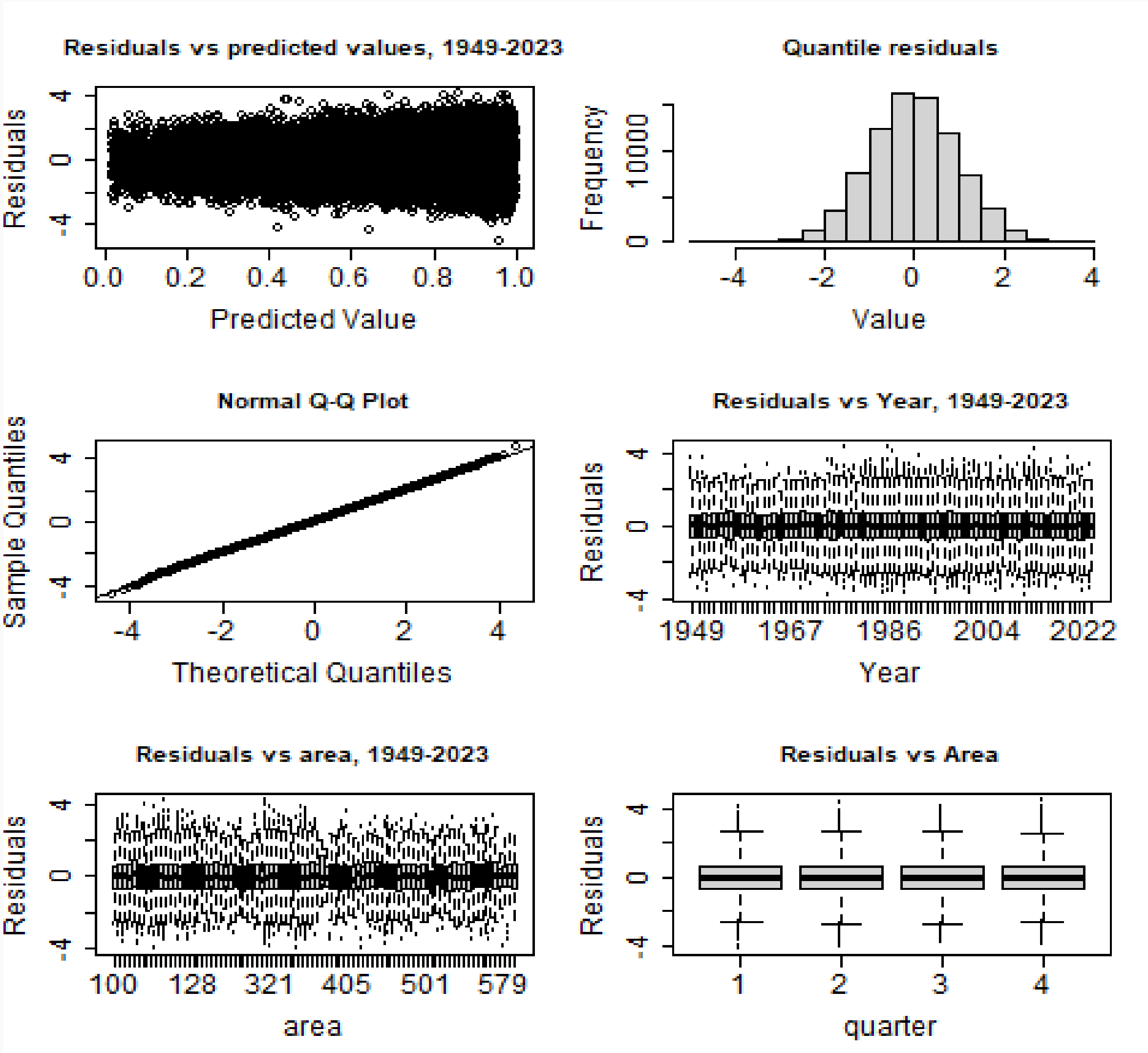


Figure C1. Diagnostic plots for the best-fit ‘ōpakapaka-only Bernoulli model used to standardize commercial CPUE.

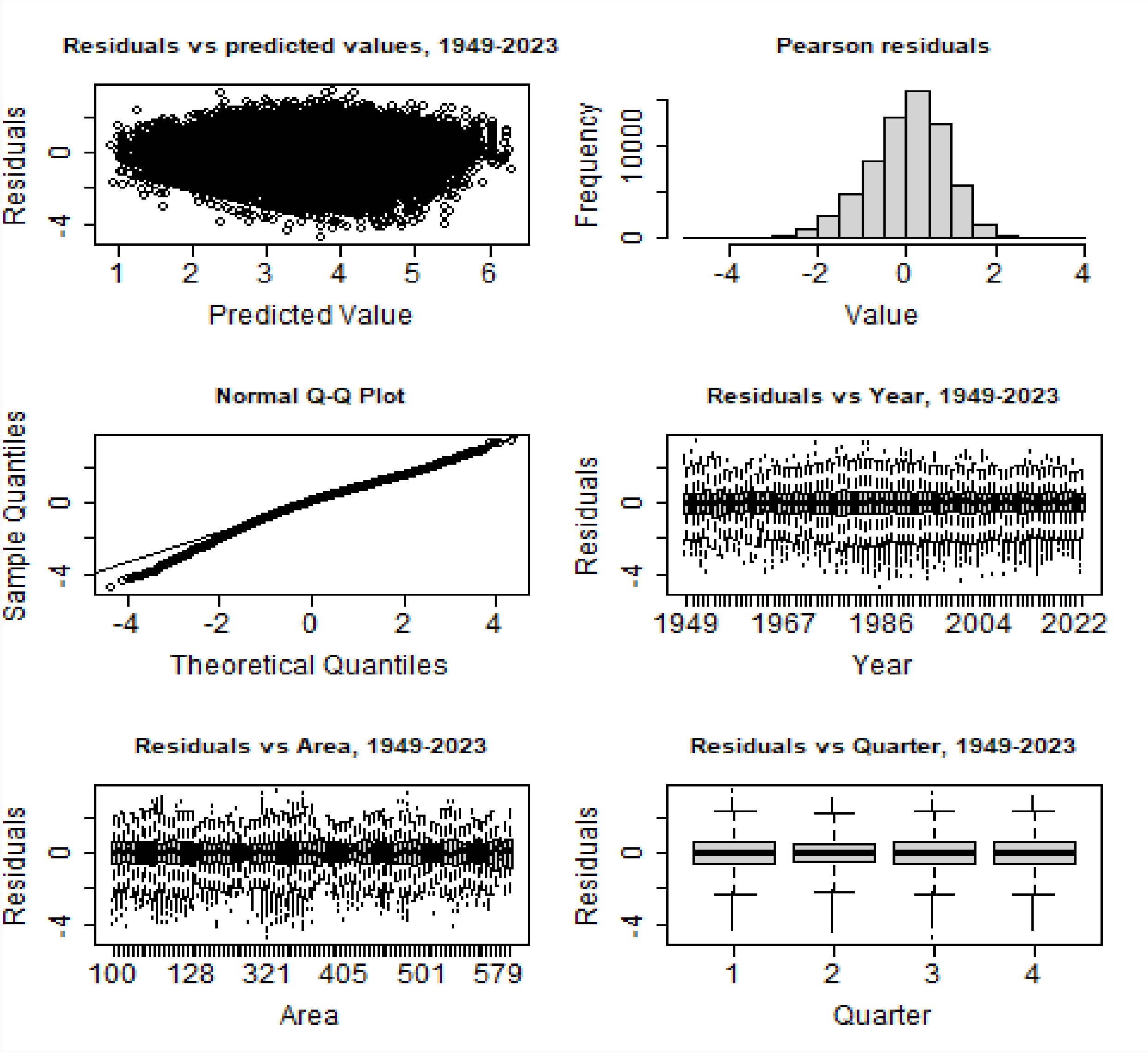


Figure C2. Diagnostic plots for the best-fit ‘ōpakapaka-only lognormal model used to standardize commercial CPUE.