



Applying a probability harvest control rule to account for increased uncertainty in setting precautionary harvest limits from past stock assessments

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ABSTRACT

Estimates of current population status, derived from stock assessments and often expressed as spawning output, are uncertain. United States (U.S.) fisheries management has adopted approaches to account for this uncertainty when setting harvest limits with the goal of preventing overfishing, as mandated by the Magnuson Stevens Fishery Conservation and Management Act. For U.S. West Coast groundfish species, the Acceptable Biological Catch (ABC), since 2011, has been set to a fraction of the Overfishing Limit (OFL) based on the uncertainty (σ) surrounding the estimated final year spawning output, as a proxy for the OFL uncertainty, or the uncertainty around the OFL itself. However, uncertainty around population size is expected to increase during the projection period when the true population dynamics may differ from the model projected values. Age-structured, data-rich stock assessments for West Coast groundfish species were evaluated for changes in spawning output uncertainty during a ten-year projection period. The spawning output time series from a more pessimistic population projection, termed the “low state of nature”, was compared to the spawning output time series from the base assessment model adopted for management, for each of 17 West Coast groundfish species (comprising a total of 21 modeled areas). The standardized median estimated σ increased, as measured between the low and base model spawning output, during the projection period for all species evaluated, increasing to a mean of 1.67 times greater than that in the first year after a ten-year projection. Combining the results by species groups used by the Pacific Fishery Management Council (rockfish, roundfish, and flatfish) elucidated that while the σ values increased for all species groups, the rockfish species had the smallest median multiplicative increase in σ (1.65) and the flatfish species had the largest increase in σ (2.66) during the projection period, although only two flatfish assessments were available. Across species, natural mortality was the best life history predictor of the rate of change in σ , however, the correlation between the two was low, though greater within species groups. Applying the estimated increase in σ by year, across species groupings, combined with an adopted risk tolerance probability for exceeding the true OFL (termed P^*) of 0.45 resulted in ABC values that decreased from 93.9 % of the OFL in the first year following the assessment to 90.0 % of the OFL by year ten of the projection period.

1. Introduction

Federal fisheries management in the United States (U.S.) operates under the Magnuson-Stevens Fishery Conservation and Management Act (MSA) and follows ten National Standards (NS) to ensure sustainable and responsible fishery management. The MSA mandates that management should aim to keep populations at or near biomass targets, a predefined level that will produce near optimum socio-economic benefits while ensuring the sustainability of fish populations (or “stocks”).

NS1 dictates that fishery management should take measures to prevent overfishing and achieve optimum yield. Stock assessments form the scientific basis for determining the current size of a fish population relative to the biomass target defined by management. Integrated stock assessment models incorporate a range of data including historical and recent catches, indices of abundance, and length- and age-composition data. These data inform estimates of current and unfished population sizes, often as a measure of spawning, or reproductive, output, along with the relative status of the stock (e.g., often defined as the ratio of the

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size of the current to the unfished spawning output). However, parameter and population estimates produced by stock assessments are uncertain and properly accounting for this uncertainty when determining harvest limits is important to prevent overfishing.

The Pacific Fishery Management Council (PFMC) manages fish species in federal waters along the U.S. West Coast. Harvest limits for federally managed fisheries are determined using three components: (1) estimating and setting an Overfishing Limit (OFL), a level of harvest that if exceeded would constitute overfishing, (2) setting an Acceptable Biological Catch (ABC), a level of harvest less than the OFL, and (3) setting an Annual Catch Limit (ACL) at or below the ABC based on any number of factors including, in some cases, other harvest control rules. The PFMC accounts for both scientific uncertainty (σ) and a pre-specified management risk tolerance (P^*) when setting ABCs to reduce the probability of overfishing as required by NS1. The PFMC specifies σ and P^* separately. Combined, these produce a scientific uncertainty and management risk tolerance reduction, termed the “buffer”, between an OFL and an ABC.

P^* is the probability of exceeding the true but unknown OFL in a single year (Shertzer et al., 2010) where a value of 0.50 represents an equal probability being below or above the true OFL in that year. NS1 requires that federal fisheries incorporate some level of risk aversion by requiring that P^* be set less than 0.50 (Federal Register, 2009). The selection of the P^* is a policy choice, considered a risk tolerance decision, and is selected by Fishery Management Councils considering population and fishery specific factors. The PFMC has adopted a policy of setting the upper limit for P^* equal to 0.45 (PFMC, 2020), and values below that limit have been selected in specific situations where the PFMC deemed that an increased risk aversion was warranted for a particular fish population.

The scientific uncertainty around the estimated OFL in the final year of the assessment model is represented by σ . The Scientific and Statistical Committee (SSC) of the PFMC specifies this value for each endorsed assessment, where σ is defined as the standard deviation of a log-normal distribution, and where the estimated OFL in the final model year is considered the median of that distribution. The magnitude of σ estimated within an assessment is closely linked to the amount of data and/or the modeling assumptions used in the stock assessment, hence, the SSC has specified default σ values that serve as lower limits that should be incorporated into data-rich, data-moderate, and data-limited assessments of West Coast groundfish species. The initial analysis to evaluate the σ of data-rich assessments was conducted in 2011 and examined the uncertainty within and among assessments for West Coast groundfish species around the estimated final year spawning output, as a proxy for the uncertainty in the OFL (Ralston et al., 2011). In 2019, an analysis was conducted to directly measure the uncertainty of the OFL estimates rather than that around spawning output (Privitera-Johnson and Punt, 2020). Each of these analyses estimated a single median value for the uncertainty at the end of the modeled period for data-rich assessments, which would then apply a constant fraction to set the ABCs below the OFLs across the model projection period. While this constant reduction incorporates some level of uncertainty when specifying ABCs, it fails to incorporate any potential increase in uncertainty that would be expected during an extended projection period.

The PFMC Groundfish Fishery Management Plan (FMP) includes nearly 90 fish species, some of which are comprised of multiple stocks across the West Coast. Of those fish species, approximately half have been assessed to estimate population size and status. Generally, assessments for U.S. West Coast groundfish are performed in two-year cycles. However, only a select subset of species in the FMP, typically ranging between six to ten, are selected for assessment within a single cycle due to limited capacity (e.g., otolith age reading, assessors to conduct assessments, review resources). The frequency of assessment for species varies based upon national guidance (Methot, 2015) for prioritizing species for assessment that includes a variety of factors, such as commercial importance (e.g., an indicator of potentially high exploitation),

the time since the last assessment, whether recent data raise concern for relative or absolute biomass (e.g., downward trend in the index of abundance), and whether new information or data sources are available.

Stock assessments of West Coast groundfish species provide ten-year projections of future OFLs and the resulting ABCs where the ABCs for each year were, until recently, set at the same proportion of corresponding annual OFL (i.e., time-invariant reduction). However, a priori, the uncertainty around spawning output projections should be expected to increase further out in time from the year in which the assessment was conducted. This increase in projection uncertainty often arises due to unknown recent or future events just prior to and during the projection period, such as recruitment success. Given the lack of capacity to conduct new assessments every two years for every assessed groundfish species off the U.S. West Coast, management advice for many species is based on older assessments conducted between two and ten years earlier, or even further in the past. Therefore, there is a need to explicitly acknowledge and account for the associated increased scientific uncertainty when determining year-specific ABCs from older assessments. This work provides estimates of the change in uncertainty during the model projection period within and among existing stock assessments of U.S. West Coast groundfish.

2. Materials and methods

U.S. West Coast groundfish stock assessment models for 17 species (a total of 21 modeled areas, including 14 coastwide and 7 regional models) were used to quantify potential changes in uncertainty given the length of time since the last assessment (Table 1). The SSC assigns assessments into one of three categories based on available data, model complexity, and overall model uncertainty, where category 1 reflects the highest category of data-rich stock assessments. A category 1 data-rich assessment generally incorporates robust indices of abundance, and size- and age-composition data that supports estimation of population parameters and dynamics including estimates of annual variation in recruitment. Category 2 assessments often incorporate similar data types to that used in a category 1 assessment, but are considered data-moderate due to fewer data or data-types and/or greater uncertainty around the available data. Finally, category 3, data-limited assessments, are the most data limited and do not estimate abundance but rather provide OFL estimates or a likely sustainable yield based on population parameters and historical exploitation.

Each assessment model examined in this analysis was classified as a category 1, data-rich stock assessment adopted by the PFMC. Models were implemented using Stock Synthesis (Methot and Wetzel, 2013), the default assessment platform for U.S. West Coast groundfish species. Calculations for year-specific ABC removals during the projection period and the resulting spawning output were based on the estimated parameters from the base models with recruitment set equal to that predicted from the stock-recruitment curve (e.g., no recruitment deviations during the projection period) and the corresponding species-specific harvest control rule as defined by the PFMC (PFMC, 2020). ABC removals during the projection period were equal to the PFMC default data-rich, category 1, buffer value determined by a σ of 0.36 (Ralston et al., 2011), coined σ_{default} , and a P^* set to a probability of 0.45. This initial σ default of 0.36 was based on analysis examining the uncertainty around spawning output from West Coast groundfish assessments (Ralston et al., 2011). The adopted σ_{default} for data-rich stock assessments has since been updated to 0.50 based on subsequent analysis conducted by Privitera-Johnson and Punt (2020) examining the estimated uncertainty around OFLs. The change in the adopted σ_{default} is likely to have a limited impact on this analysis and the conclusions drawn since 1) the uncertainty around the spawning output was appropriately applied to the spawning output rather than the OFL in this analysis and 2) all evaluations around the change of uncertainty across the projection period were standardized relative to the uncertainty in the first year, resulting in an estimated proportional rate of change of

Table 1

U.S. West Coast data-rich, category 1, stock assessments included in the analysis, for species assessed either on a coastwide stock basis or as area-based stocks, grouped by species groups. Area based assessments were either based on State (California [CA], Oregon [OR], or Washington [WA]), State-area (OR-South, CA-North, CA-South), or by region across the coast (North or South).

Species Group	Stock	Scientific name	Reference
Rockfish	Aurora rockfish	<i>Sebastodes aurora</i>	Hamel et al. (2013)
	Black rockfish (CA)	<i>Sebastodes melanops</i>	Cope et al. (2016)
	Black rockfish (WA)	<i>Sebastodes melanops</i>	Cope et al. (2016)
	Bocaccio	<i>Sebastodes paucispinis</i>	He et al. (2015)
	California scorpionfish	<i>Scorpaena guttata</i>	Monk et al. (2017)
	Canary rockfish	<i>Sebastodes pinniger</i>	Thorson and Wetzel (2015)
	Chilipepper rockfish	<i>Sebastodes goodei</i>	Field et al. (2015)
	Darkblotched rockfish	<i>Sebastodes crameri</i>	Gertseva et al. (2015)
	Splitnose rockfish	<i>Sebastodes diploproa</i>	Gertseva et al. (2009)
	Widow rockfish	<i>Sebastodes entomelas</i>	Hicks and Wetzel (2015)
	Yelloweye rockfish	<i>Sebastodes ruberrimus</i>	Gertseva and Cope (2017)
	Yellowtail rockfish	<i>Sebastodes flavidus</i>	Stephens and Taylor (2017)
Roundfish	Cabezon (OR-South)	<i>Scorpaenichthys marmoratus</i>	Cope and Key (2009)
	Cabezon (CA-North)	<i>Scorpaenichthys marmoratus</i>	Cope and Key (2009)
	Cabezon (CA-South)	<i>Scorpaenichthys marmoratus</i>	Cope and Key (2009)
	Kelp greenling	<i>Hexagrammos decagrammus</i>	Berger et al. (2015)
	Lingcod (North)	<i>Ophiodon elongatus</i>	Haltuch et al. (2018)
	Lingcod (South)	<i>Ophiodon elongatus</i>	Haltuch et al. (2018)
Flatfish	Sablefish	<i>Anoplopoma fimbria</i>	Stewart et al. (2011)
	Dover sole	<i>Microstomus pacificus</i>	Hicks and Wetzel (2011)
	Petrale sole	<i>Eopsetta jordani</i>	Haltuch et al. (2013)

uncertainty by assessment that can be considered independently from the σ_{default} .

U.S. West Coast groundfish stock assessments express within-model uncertainty through a “Decision Table” approach which creates a range of potential alternative states of nature, termed low and high, for the assessed population relative to the base model. The low and high states of nature typically include modifications to the base model of a single key parameter (e.g., natural mortality (M), steepness (h), catchability of a survey (q), etc.) or a combination of multiple parameters that are considered highly uncertain or influential to the estimated stock status or stock size within that particular assessment model. This approach involves assigning a probability to each state of nature, where the base model is considered the most likely, assigned a 50 % probability of being the true state of the population, and subsequently both the low and high states of nature are given equal probability of 25 %. We initially considered using the low and base states of nature from each assessment for our analysis. However, the alternative states of nature in Decision Tables were not developed with this sort of analysis in mind. The variability in approaches used across West Coast groundfish Decision Tables results in wide variability in the biomass level, age distribution, and relative dynamics of the alternative states of nature relative to the base model.

In this analysis, a standardized low state of nature for each assess-

ment was created through modifications to the initial unfished equilibrium recruitment (R_0) such that the resultant model had a spawning output in the terminal year (prior to the projection period) that was reduced by the scientific uncertainty buffer from that of the base model in each case (Fig. 1). The spawning output for the low state of nature (SB_{low}) was calculated as:

$$SB_{\text{low},s,y=p} = \frac{SB_{\text{base},s,y=p}}{e^{\sigma_{\text{default}} * z_{\text{value}}}} \quad (1)$$

where $SB_{\text{base},s,y=p}$ is the spawning output of the base model for stock s in first year of the projection period (p) of the model ($y = p$), σ_{default} of 0.36, and a z_{value} of 1.15 based on the 75 % confidence interval of the standard normal distribution (i.e., mean of 0 and standard deviation of 1), which would reflect the mid-quantile (12.5 %) of the low state of nature assuming a 25 % probability. $SB_{\text{low},s,y=p}$ was created in three steps. First, find and fix R_0 to the value that results in the desired $SB_{\text{low},s,y=p}$ with a 1 % margin of error in the final model year as defined in Eq. (1) while allowing for all other parameter values (e.g., recruitment deviations, selectivity, growth) to be estimated in the same manner as in the base model. Second, fix the future removals during the projection period in both the low state of nature and base models equal to the annual ABC estimated from the base model. Finally, project both the base and low state of nature populations forward in time for ten years. During the projection period, recruitment was predicted from the stock-recruitment curve without variability in the same manner as future recruitments in the respective models. However, the annual realized recruits during the model period differed for each model based on the calculated spawning output levels in each year.

The spawning outputs by stock and year during the projection period between the base and low states of nature models were compared and the uncertainty in the spawning output was calculated as:

$$\sigma_{s,y} = \frac{\log(SB_{\text{base},s,y} / SB_{\text{low},s,y})}{z_{\text{value}}} \quad (2)$$

where y ranged from the first projection year (p) to the tenth projection year and the z_{value} was 1.15 as above. The linear rate of change, r , in the σ by stock and year was calculated as:

$$r_{s,y} = \frac{(\sigma_{s,y} / \sigma_{\text{default}}) - 1}{y - 1} \quad (3)$$

For ease of comparison and visualization the $\sigma_{s,y}$ values across the projection period were standardized relative to the first projection year ($y = p$). Finally, the estimated rate of change r in σ , combined with the default initial uncertainty (σ_{default}) and management risk tolerance, P^* , values were used to calculate new annual buffer values as:

$$\text{buffer} = e^{(r_{s,y} + 1)\sigma_{\text{default}}\Phi^{-1}(P^*)} \quad (4)$$

where Φ^{-1} is the inverse cumulative normal distribution function and P^* was equal to either 0.45 or 0.40 (e.g., commonly selected P^* values for category 1 groundfish stock assessments).

Life history parameters from each base model were recorded to investigate linkages between biological traits and uncertainty estimated in assessment model projection period. For each species the value of M , slope of maturity curves (s), length at 50 % maturity ($L_{50\%}$), maximum length (L_{MAX}), Brody growth coefficient (k), annual recruitment variation (σ_R), and h were examined (Table 2). Additionally, the ratio of L_{MAX} to $L_{50\%}$ was derived as a measure of the rate of growth that would also incorporate the trait of some fish (e.g., particularly rockfish) maturing at sizes near the L_{MAX} (Table 2). Many U.S. West Coast groundfish assessments assume sex-specific biology, and for simplicity, and, because spawning output is based on females only, the female biological parameters were used for σ comparisons. The majority of West Coast groundfish stock assessments model M as a single parameter across all ages. However, the stock assessment of canary rockfish (*Sebastodes*

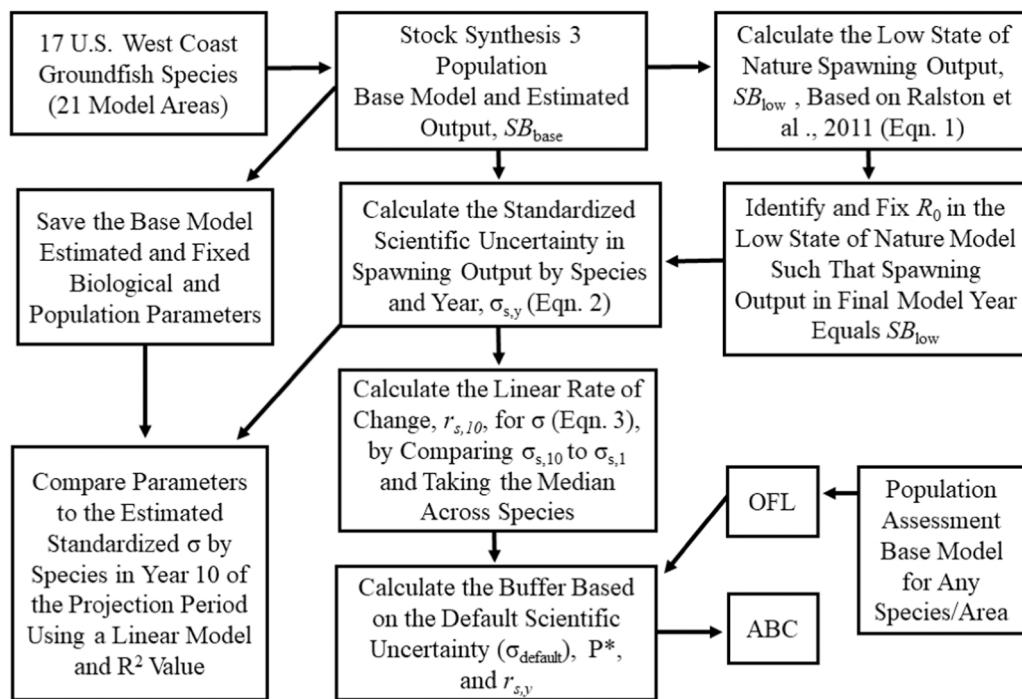


Fig. 1. A conceptual overview of the steps taken to estimate the change in uncertainty ($\sigma_{s,y}$) in spawning output between the base model (SB_{base}) and a low state of nature (SB_{low}) during the projection period, estimation of the linear rate of change ($r_{s,y}$), and the determination of the time-varying buffer based reduction between the Overfishing Limit (OFL) and Acceptable Biological Catch (ABC).

Table 2

Biological parameters for each West Coast stock assessment. Sex-specific parameters reflect the value for females for each stock. Three stocks, black rockfish, cabezon, and lingcod, had multiple models and parameters are the average value across the models. The natural mortality for canary rockfish is the average value across all ages.

Stock	Biological Parameters							
	Natural Mortality ($M \text{ yr}^{-1}$)	Steepness (h)	Variation in Recruitment (σ_R)	Slope of Maturity (s)	Length at 50 % Maturity (cm, $L_{50\%}$)	Brody Growth Coefficient (k)	Maximum Length (cm, L_{MAX})	Ratio of $L_{MAX} / L_{50\%}$
Aurora rockfish	0.04	0.78	0.50	-0.62	25.5	0.09	30.8	1.21
Black rockfish	0.17	0.77	0.50	-0.66	43.7	0.14	54.2	1.24
California	0.18	0.77	0.50	-0.66	43.7	0.15	54.5	1.25
Washington	0.16	0.77	0.50	-0.66	43.7	0.14	53.9	1.23
Bocaccio	0.18	0.77	1.00	-0.33	37.7	0.23	67.3	1.79
Cabezon	0.25	0.70	0.57	-0.59	37.6	0.16	63.4	1.69
South California	0.26	0.70	0.70	-0.70	34.6	0.21	55.8	1.61
North California	0.24	0.70	0.50	-0.70	34.6	0.21	55.8	1.61
Oregon	0.24	0.70	0.50	-0.34	43.7	0.33	64.4	1.47
California scorpionfish	0.24	0.72	0.60	-1.20	18.0	0.29	31.9	1.77
Canary rockfish	0.09	0.77	0.50	-0.25	40.5	0.13	60.0	1.48
Chilipepper rockfish	0.16	0.57	1.00	-0.27	24.4	0.19	47.3	1.94
Darkblotched rockfish	0.05	0.77	0.75	-0.64	34.6	0.20	42.7	1.23
Dover sole	0.12	0.80	0.35	-0.78	35.0	0.15	47.8	1.37
Kelp greenling	0.36	0.70	0.65	-1.00	29.3	0.26	36.4	1.24
Lingcod	0.26	0.70	0.65	-0.24	54.5	0.13	101.8	1.87
South	0.26	0.70	0.75	-0.22	52.3	0.13	93.4	1.79
North	0.26	0.70	0.55	-0.27	56.7	0.13	110	1.94
Petrale sole	0.16	0.84	0.40	-0.74	33.1	0.13	54.3	1.64
Sablefish	0.08	0.60	1.10	-0.13	58.0	0.33	64.0	1.10
Splitnose rockfish	0.05	0.58	1.00	-0.57	21.8	0.16	29.6	1.36
Widow rockfish	0.16	0.80	0.60	-0.77	29.7	0.20	50.4	1.70
Yellowtail rockfish	0.17	0.72	0.50	-0.40	42.5	0.14	53.6	1.26
Yelloweye rockfish	0.04	0.72	0.50	-0.40	42.1	0.06	63.5	1.51

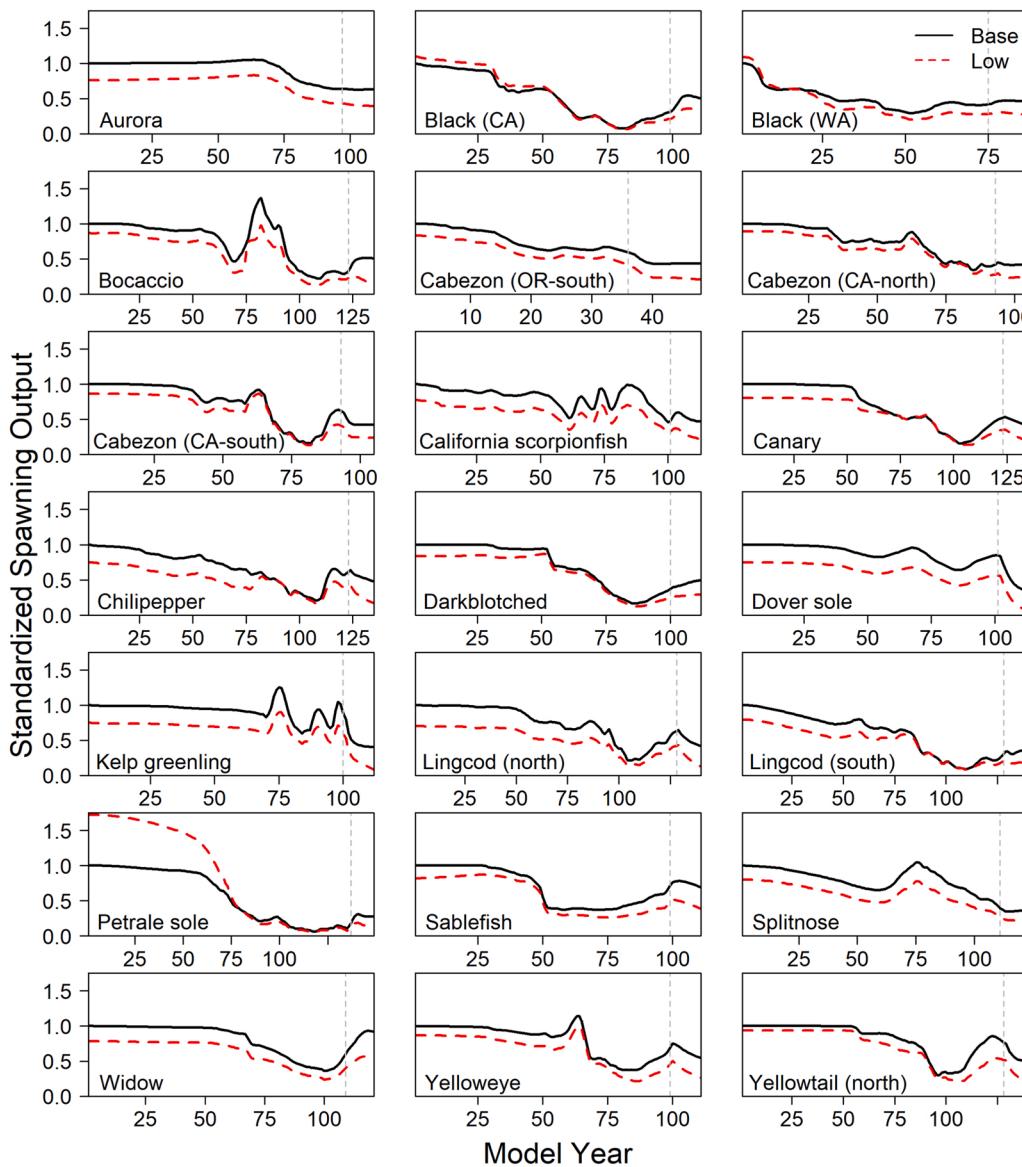


Fig. 2. The normalized spawning output trajectory from the base model (solid black line) and the low state of nature (dashed red line) for each stock by model year. Values to the right of the vertical, dashed grey line are for the ten year projection period. Spawning output was standardized to the unfished spawning output in the first year of the base model for each stock to visualize differences since the initial spawning output was highly variable across stocks. The historical model years were standardized to start at 1 for clarity across assessments which had highly variable model years.

pinniger) has a unique specification of M where it is lower for female fish younger than 6 years of age fixed at a value of 0.052 yr^{-1} and increases linearly from that value at age 6 to a peak fixed value of 0.10 yr^{-1} for fish age 14 and above. For simplicity, the life history analysis defined a single M value for canary rockfish by using a weighted across-age M value of 0.088 yr^{-1} (Table 2). Each of the life history parameters was compared to the estimated change in σ at the end of the projection period. A linear model was fit to the data for each biological parameter and an R-squared (R^2) value calculated.

The most recent benchmark assessment model as of 2019 for each stock (Table 1), where PFMC defines a benchmark assessment model as a new evaluation of a stock where all previous data and modeling assumptions may be re-evaluated, was used for this analysis. One exception was made in the case of chilipepper rockfish (*Sebastodes goodei*), where an update assessment was used instead of the benchmark assessment because the most recent benchmark assessment, performed in 2007, was conducted using an out-of-date modeling platform which prohibited the creation of a new low state of nature. In contrast to a benchmark assessment, an update assessment is defined as a re-evaluation of a stock where all previous data and modeling assumptions are retained and the only change is that the most recent data are added to the assessment model.

The species included in this analysis included three that have multiple area-based stock assessments, black rockfish (*Sebastodes melanops*), cabezon (*Scorpaenichthys marmoratus*), and lingcod (*Ophiodon elongatus*). Black rockfish has three models used for management; however, only the California and Washington models are classified as category 1 assessments. The results from each model were weighted such that each species received the same weight in the final analysis when summarizing results across all species and species groupings.

Species were grouped into categories based on management practices of the PFMC for U.S. West Coast groundfish species: rockfish, roundfish, and flatfish. The three groundfish categories have category-specific proxy F_{MSY} harvest rates which have been defined based upon life history traits (Dorn, 2002; Ralston, 2002). The change in σ during the projection period was evaluated by individual species, species grouping, and across all species combined. The grouped results, either by species groupings or for all species, were conducted using the weightings described above. California scorpionfish (*Scorpaena guttata*) are members of the *Scorpaenidae* family within the *Scorpaeniformes* order, the same order that includes the rockfish genus (*Sebastodes*). Hence, California scorpionfish were included in the rockfish species group.

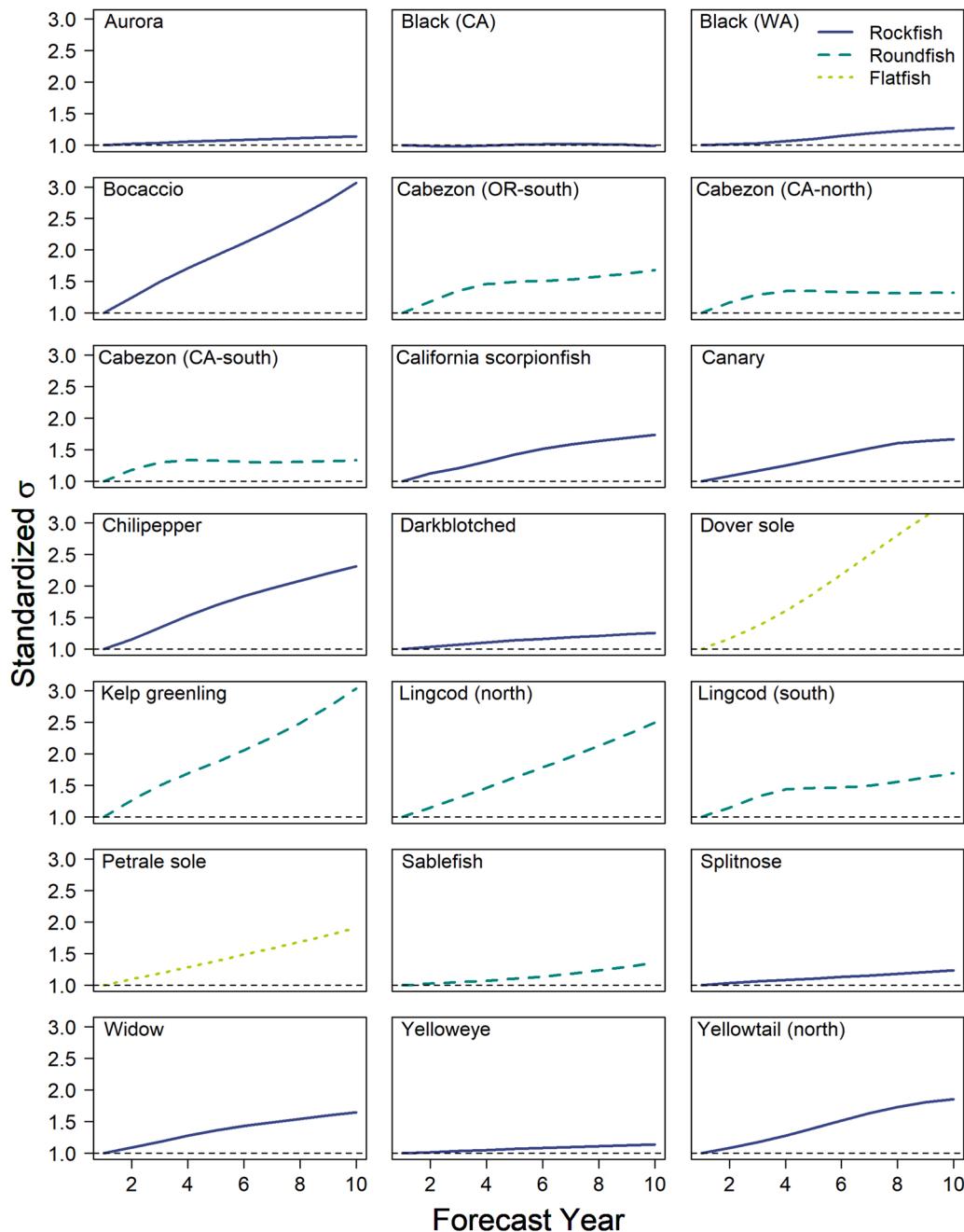


Fig. 3. The standardized change in scientific uncertainty (σ) during the projection period between the base and low states of nature for each stock. The final year value for Dover sole of 3.4 was not shown due to scale. The life-history categories are indicated by line color where rockfish are shown in solid blue, roundfish in dashed teal, and flatfish in dotted yellow.

3. Results

3.1. Stock-specific changes in σ

Generally, the unfished spawning outputs for the low state of nature were less than the base model (Fig. 2), as would be expected given that the low state of nature should reflect a lower initial population size or a more-depleted state (or both) relative to the base model. However, there were some exceptions where the spawning output at unfished equilibrium for the low state of nature model was larger than the base model (i.e., black rockfish off California and Washington and petrale sole) and then decreased to a lower state than the base model as the time series reached the terminal model year y (thus reflecting a more depleted state). This occurred due to other estimated parameters changing

between the base and low state of nature models. The change in spawning output between the base and low states of nature during the projection period, shown to the right of the vertical dashed grey line (Fig. 2), varied by population, driven by model-specific parameter assumptions (e.g., fixed versus estimated parameters including recent annual variations in recruitment, M , or h) and life history (e.g., longevity).

The annual rate of change in the stock-specific σ was variable across species and within species stocks (Fig. 3). The assessments for kelp greenling (*Hexagrammos decagrammus*), bocaccio (*Sebastodes paucispinis*), and Dover sole (*Microstomus pacificus*) each had large increases in σ during the ten-year projection period with increasing σ to 3.0, 3.1, and 3.4 times the base value, respectively. In contrast, some populations had very little change in σ during the projection period. Assessments for

aurora rockfish (*Sebastodes aurora*), yelloweye rockfish (*Sebastodes ruberrimus*), splitnose rockfish (*Sebastodes diploproa*), and black rockfish off California provided the smallest increase between the base and low states of nature spawning output ranging between factors of 1.0–1.2.

Black rockfish, cabezon, and lingcod had variable results by modeled area. These results were generally driven by area-specific modeling assumptions and whether the population would reach the management target during the projection period. The California-north and California-south models for cabezon had similar trends in the changes between the base and low state of nature spawning output values, with the trajectories stabilizing at or near the management targets for the base models during the projection period resulting in limited change in the σ . In contrast, the Oregon-south model for cabezon had an increase in σ during projections (Fig. 3). California and Washington black rockfish assessment models demonstrated two marginally different patterns in σ during the projections (Fig. 3). The Washington black rockfish assessment model had a small increase in σ while the California model had little to no change, with even a small decrease in σ at the end of the projection period between the base and low state of nature spawning outputs. Both lingcod models had increasing uncertainty by the end of the projection period, but the North model had a sharper increase in uncertainty. The contrast in results by area for black rockfish, cabezon, and lingcod suggests that both model structure and life history based population dynamics, and potentially the exploitation history, impact the resulting change in σ since the observed inter-species differences in σ did not appear to be associated with life history characteristics in a predictable fashion. This indicates that the result of this analysis for individual stock assessments may be less reliable than results combined across assessments.

3.2. Pooled-stock changes in σ

Combining the results across all species, with species-specific, area-based models weighted accordingly, there was an increasing trend in σ , resulting in a median increase by a factor of 1.67 in σ across all populations by year ten of the projection period (Table 3 and Fig. 4a). The change in the standardized σ over the projection period varied based on the species group where rockfish had the smallest change in the median σ compared to roundfish and flatfish (Fig. 4b-d). The median increase in σ by year ten of the projection period for the rockfish, roundfish, and flatfish groups were 1.65, 1.77, and 2.66 times the base value, respectively (Table 3). However, the result for the flatfish species group is based on only two species, petrale sole and Dover sole, and may not be representative of the uncertainty for future assessments of flatfish species off the U.S. West Coast. The individual results across species were highly variable even among rockfishes.

To explore the basis for the increase in σ further, the rockfish species were subdivided into two groups based on somewhat arbitrarily selected “low” (i.e., $M < 0.10 \text{ yr}^{-1}$) and “high” (i.e., $M \geq 0.10 \text{ yr}^{-1}$) M where the estimated or fixed M values from the base models ranged from 0.035 to 0.235 yr^{-1} . The rockfish species included in the low- M group, listed in

Table 3

The median change in the standardized scientific uncertainty (σ) for each projection year for all species combined and grouped by species group.

Projection Year	All species	Rockfish	Roundfish	Flatfish
1	1.00	1.00	1.00	1.00
2	1.09	1.08	1.16	1.13
3	1.18	1.16	1.31	1.28
4	1.28	1.25	1.42	1.44
5	1.38	1.34	1.47	1.63
6	1.43	1.43	1.51	1.83
7	1.52	1.49	1.55	2.04
8	1.60	1.54	1.62	2.24
9	1.64	1.60	1.69	2.45
10	1.67	1.65	1.77	2.66

ascending order of M , were aurora rockfish, yelloweye rockfish, splitnose rockfish (*Sebastodes diploproa*), darkblotched rockfish (*Sebastodes crameri*), and canary rockfish (Table 2). Widow rockfish (*Sebastodes entomelas*), chilipepper rockfish, black rockfish (i.e., two area models for California and Washington), yellowtail rockfish (*Sebastodes flavidus*), and bocaccio, listed in ascending order of M values, comprised the high- M group (Table 2). The change in σ during the projection period when rockfish species were sub-divided showed a larger increase in σ for species with high M (Fig. 5). Canary rockfish was an outlier in the low- M group, having a trend in σ that was more similar to the high- M group. This may be related to the unique specification of an age-based M value for canary rockfish, such that the averaged value of M used does not fully represent the impact of age-based M dynamics and potential change within the projection period between the base and the low state of nature models.

3.3. Life history parameters and changes in σ

Various life history parameters were examined in an attempt to determine their predictive power for the change in σ (Fig. 6). The subset of biological parameters examined were selected because they were thought to possibly be indicative of the population turn-over rate (e.g., M), the pace of life history dynamics (e.g., maturity, growth), or the potential annual variability of recruitment (e.g., σ_R) which could be influential in the rate of change in σ during the projection period. However, across all species, none of the biological parameters examined had substantial explanatory power related to the change in σ (Fig. 6), although M , which is the most obvious candidate, did explain the highest proportion of the variance ($R^2 = 0.17$).

While no strong relationship between the biological parameters examined and the final projection year σ was identified across all species, two potential relationships, primarily for rockfish species, were detected when results were combined by species group (Fig. 7). Rockfish species with lower M have reduced monotonically non-decreasing changes in σ compared to rockfish species with higher M (Figs. 5 and 7a). However, the relationship between M and σ for higher M was highly variable. The ratio between L_{MAX} and $L_{50\%}$ explained 50 % of the total variation in the rate of change in σ for rockfish (Fig. 7c). Generally, species with low M had lower ratio between L_{MAX} and $L_{50\%}$ relative to species with higher M values. The only exception was black rockfish which has a higher M (0.17 yr^{-1}) but a lower ratio between L_{MAX} and $L_{50\%}$ (1.24). The relationship between M and σ explained a large percentage of the total variation for the roundfish species group (Fig. 7b), but was based upon only four observations. Flatfish data were not explored for this analysis because only two species were available.

The small sample sizes for roundfish and flatfish preclude using results from either of those taxa alone. For West Coast groundfish, we would recommend using information from the “all species” column in Table 3. The average annual fractional increase (r) in σ , as a proportion of σ_{default} , between the first and tenth year of the projection period across all species was estimated to be 0.075 (Table 3). For West Coast groundfish the PFMC calculates a percent reduction between the OFL and ABC according to Eq. 4 using both a σ_{default} , the linear rate of change (r) by year in the σ , and a P^* value. Given a P^* value of 0.45, a σ_{default} value of 0.50 (e.g., the current PFMC adopted σ_{default} based on Privitera-Johnson and Punt, 2020), and linear annual rate of change of 0.075, resulted in increasing reductions between the OFL and ABC by year where the ABC in the first projection year was 93.9 % of the OFL, decreasing to 90 % of the OFL by the tenth year of the projection period (Table 4).

Without evidence that life-history parameters are strong predictors of the increase in σ , a reasonable approach for species like West Coast groundfish may be to group species across M values observed (e.g., based on either fixed or estimated values from assessments) to calculate a common rate of increase in σ . Exploring the implications of the relationship between M and σ is still valuable, for sensitivities, and

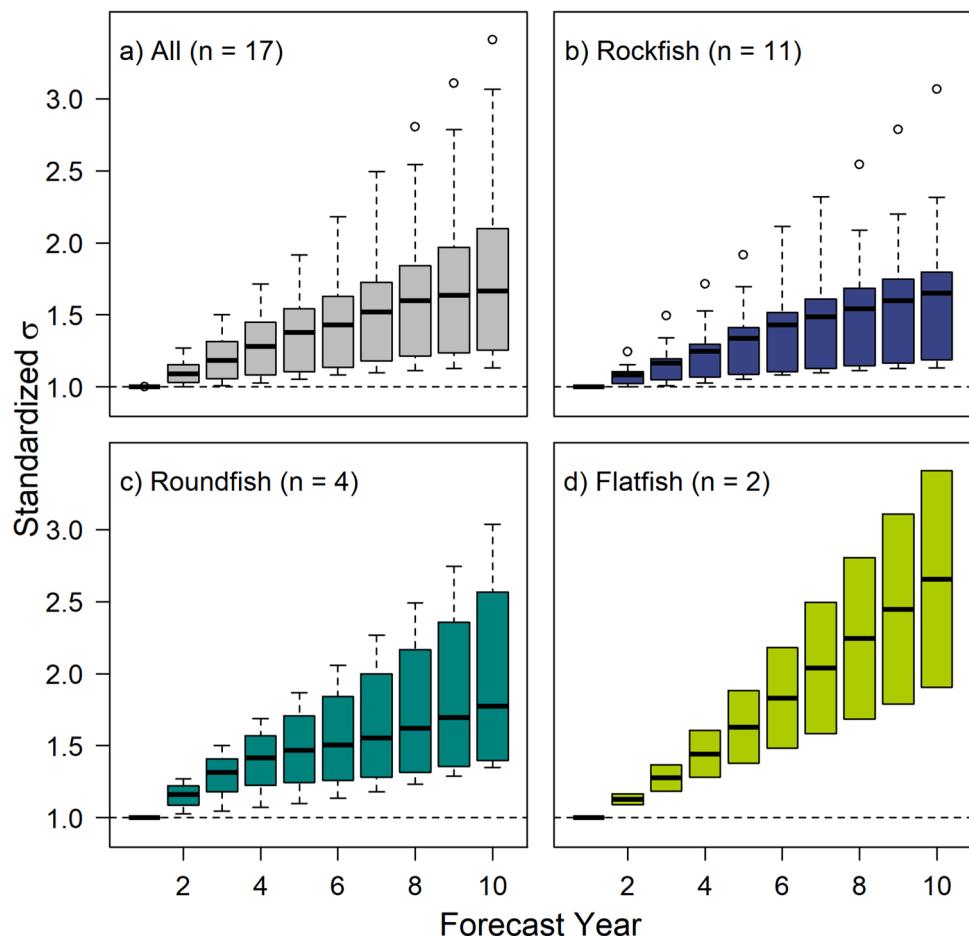


Fig. 4. The change in the standardized scientific uncertainty (σ) during the projection period between the base and low state of nature grouped by species group. The solid black line indicates the median value with the filled area corresponding to the 25th and 75th percentiles. The number (n) of species in each species group is shown in parentheses for each panel.

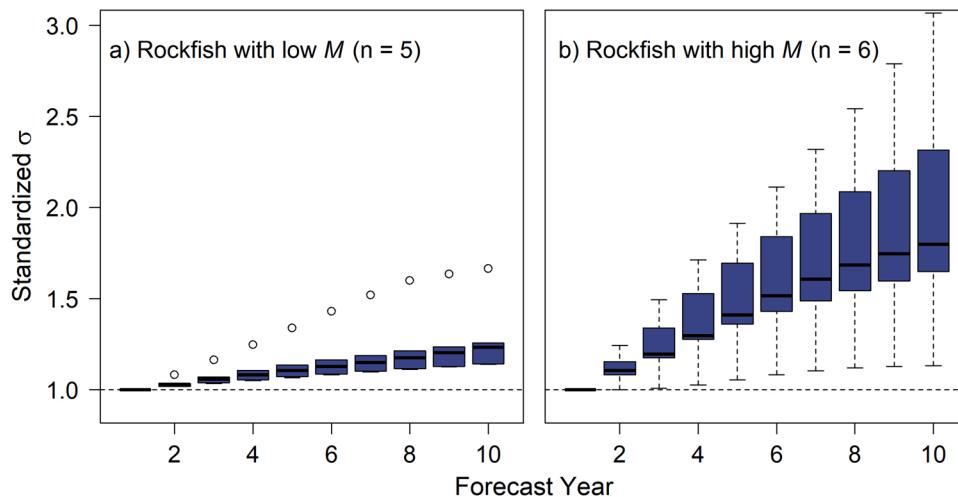


Fig. 5. The change in standardized scientific uncertainty (σ) during the projection period between the base and low state of nature for rockfish species with the results grouped by low natural mortality ($M < 0.10 \text{ yr}^{-1}$) and higher natural mortality ($M \geq 0.10 \text{ yr}^{-1}$) values where the number (n) of species in each M group is show in the parentheses. The low M group (a) was composed of aurora rockfish, canary rockfish, darkblotched rockfish, splitnose rockfish, and yelloweye rockfish and the high M group (b) was comprised of black rockfish, bocaccio rockfish, California scorpionfish, chilipepper rockfish, widow rockfish, and yellowtail rockfish.

especially at the extremes of the range of M observed. To estimate the relationship between M and the rate of increase in σ , a linear model was applied to the M and the estimated change in standardized σ in year ten of the projection period where both quantities were transformed into natural log-space. The observed median relationship between M and the proportional rate of increase in σ from the linear model is $r = 0.52 M$. The estimated proportional rate of change in σ based on M is

approximately equal to the average rate of change across all species when calculated based on the average M of 0.15 yr^{-1} (0.078 yr^{-1} based on M compared to the rate of change of 0.075 yr^{-1} found above).

4. Discussion

The importance of accounting for increased uncertainty during the

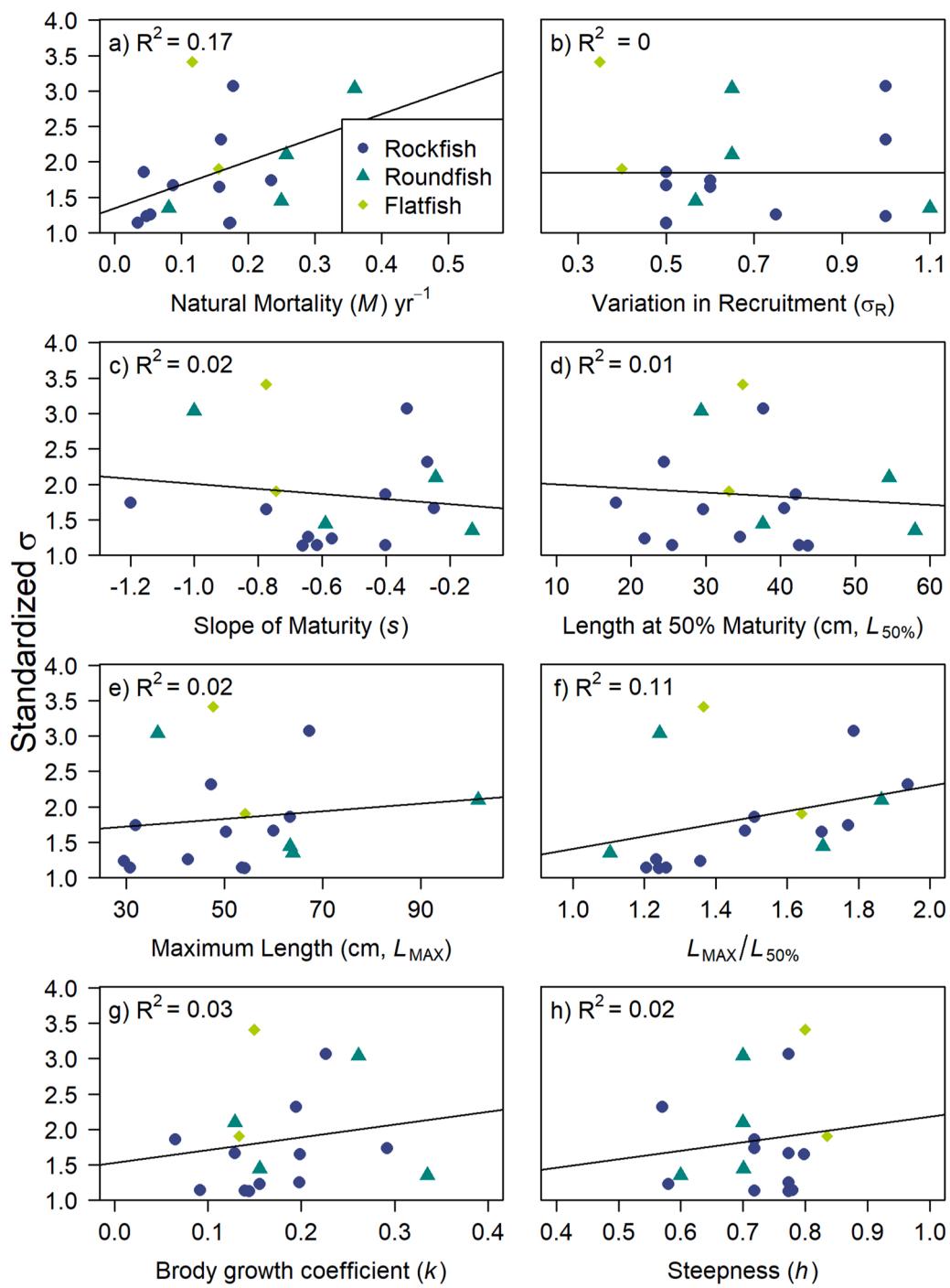


Fig. 6. The standardized scientific uncertainty (σ) at the end of the ten year projection compared to select life history parameters from each base model with rockfish species shown in blue circles, roundfish in teal green triangles, and flatfish in yellow diamonds. A linear model was fit relative to each life history parameter separately and the R^2 is reported in the upper left of each panel.

projection period when determining an ABC depends upon the expected assessment frequency. The large number of species managed by the PFMC and limited assessment and review capacity often results in periods of time between assessments that can range between two and ten years, or even longer periods, depending upon the importance of the species to commercial and recreational fishing, the magnitude of annual removals relative to existing ABCs (or ACLs if set lower than the ABC), and the level of concern around stock status. This work showed that even across the short-term (two years), and especially across the long-term (ten years), an increase in σ can occur, as measured by differences between the low and base model spawning outputs that, if accounted for

when setting ABCs, would result in additional levels of precaution.

The observed variation in the magnitude and trend in the spawning output during the projection period between the base and low states of nature varied greatly by assessment. The variation observed between the base and low states of nature models during the projection period appeared to be driven by multiple factors rather than only the underlying assessment uncertainty around the estimated spawning output. One factor identified was whether the population would reach the management target biomass during the projection period resulting in a stabilization of ABC removals and the projected spawning output. This stabilization of the population seemed to result in a plateauing of the

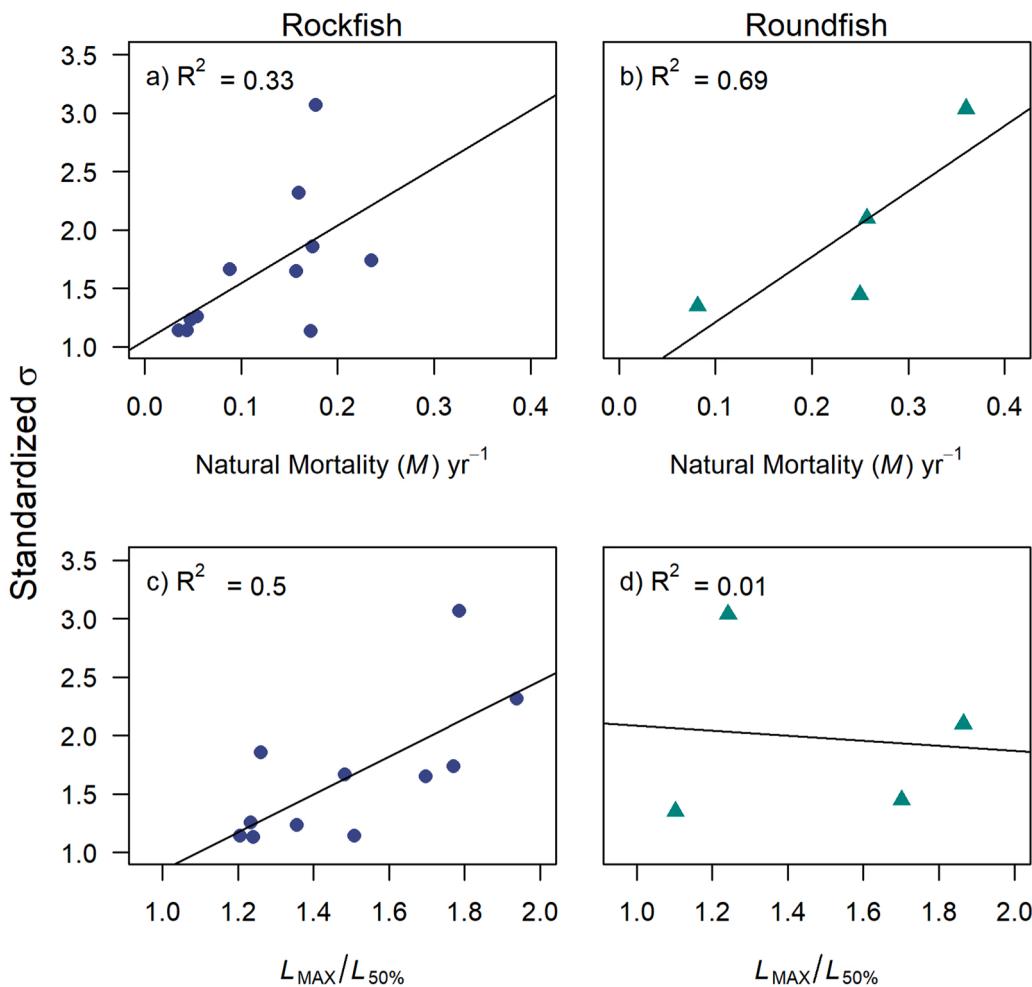


Fig. 7. The standardized scientific uncertainty (σ) at the end of the ten year projection period compared to the natural mortality ($M \text{ yr}^{-1}$) and the ratio of maximum length (cm, L_{MAX}) to the length at 50 % maturity (cm, $L_{50\%}$) for rockfish (panels a and c) shown in blue circles and roundfish (panels b and d) shown in teal green diamonds. A linear model was fit relative to each life history parameter separately and R^2 reported in the upper left on each panel.

Table 4

Year-specific buffers, percent reductions applied to the overfishing limit (OFL) to determine the acceptable biological catch (ABC), for each projection year. The buffer values were calculated using the time-varying linear rate of change (r_y) of 0.075 estimated using all species combined, an initial scientific uncertainty (σ_{default}) of 0.50, and two alternative pre-defined management risk tolerances (P^*) of 0.45 and 0.40. For example, using Eq. 4 to calculate the buffer with a P^* of 0.45 in the first year of the projection period the ABC would be equal to 93.9 % of the OFL and the ABC in the final year of the projection period would be reduced to 90 % of the OFL.

Projection Year	$r_y\sigma$	Buffer $P^* = 0.45$	Buffer $P^* = 0.40$
1	0.5000	93.9 %	88.1 %
2	0.5375	93.5 %	87.3 %
3	0.5750	93.0 %	86.4 %
4	0.6125	92.6 %	85.6 %
5	0.6500	92.2 %	84.8 %
6	0.6875	91.7 %	84.0 %
7	0.7250	91.3 %	83.2 %
8	0.7625	90.9 %	82.4 %
9	0.8000	90.4 %	81.7 %
10	0.8375	90.0 %	80.9 %

calculated σ by year (Eq. 2) between the base and low states of nature. The first example of this behavior is the two stock assessments for California cabezon. Both stocks had plateauing differences in σ during the projection period due to both the base and low spawning output levels

reaching a steady state at the management target (or below for the low state of nature). In contrast, Dover sole was estimated to be well above the biomass target by the base model. This coupled with a relatively large estimated stock size resulted in the base model estimating large ABC removals during the projection period to drive the stock down towards the target biomass. These ABC removals, when applied to the low state of nature, resulted in large differences during the projection period from the base model spawning output estimates and hence a large change in σ during that period.

A second factor impacting the uncertainty during the projection period was related to what parameters were fixed or estimated in the assessment model. Many U.S. West Coast rockfish assessments are unable to internally estimate M and/or h and often these parameters are fixed at externally-determined point estimates from meta-analyses (Hamel, 2015; Hamel and Cope, 2022; Thorson et al., 2018). M and h are closely linked to reference points (i.e., F_{target} , SB_{target}), stock size, and the rate of change that the population can undergo on short time scales, such as the ten-year projection period examined here. The rate of change in σ was lowest in species where not only these parameters were fixed but, in particular, M was fixed at a low value as observed for some rockfish species: aurora, yelloweye, splitnose, and darkblotched rockfishes. The exceptions to this were the projected spawning outputs by the assessment models for both of the black rockfish stocks in which h was fixed but M values were estimated within the range observed or estimated for the high M rockfish stocks (between 0.16 and 0.18 yr^{-1} , Cope et al., 2016), but the estimated change in σ across the projection

period was nonetheless very low. Each of the black rockfish stocks also had low state of nature models displaying the non-typical behavior of having a larger initial spawning output compared to the base model, indicating potential unusual model rigidity and unexpected parameter compensation (e.g., changes in parameters that would not be anticipated to be linked to a low state of nature of the stock such as annual recruitment deviations).

Another population that had a higher unfished spawning output in the low state of nature relative to the base model was petrale sole. Petrale sole is one of the most data-rich groundfish species off the U.S. West Coast with large quantities of length- and age-composition data, as well as having highly informative fishery-independent trawl survey data (e.g., indices of abundance, lengths, and ages, [Haltuch et al., 2013](#)). The abundance of data supports the estimation of a large number of parameters (i.e., 298 parameters were estimated) including h and sex-specific M . Exploitation of petrale sole, across time, has been high, with the stock assessment estimating an all-time low population size in 1993 at 6 % of unfished spawning output. Combining the high model flexibility via estimated parameters and the rigidity of the population being highly depleted during the time series creates a situation where the initial spawning output of the low state of the population needed to be at a level considerably higher than that of the base model (i.e., primarily created by estimating h at the upper bounds combined with low sex-specific M). This divergence in estimated unfished spawning output between the two states of nature combined with the life history resulted in a moderate rate of change in the difference in spawning output between the two models during the projection period ([Fig. 3](#)).

There was limited evidence for the ability of specific life history parameters to predict the amount of variation between the low and base states of nature during the projection period. Rockfish comprised the largest percent of species examined and are known to have low M and late maturity ([Love et al., 2002](#)). The ratio of L_{MAX} to the $L_{50\%}$ and the value of M for rockfish species had the highest correlations with the estimated change in σ . However, these parameters explained a limited amount of variance. The change in σ relative to the value of M for rockfish species was variable with only a limited amount of the variance being explained by M ($R^2 = 0.33$) particularly for rockfish species with higher fixed or estimated M values where there was large variation in the standardized σ . The amount of variance explained by M for observed roundfish species (e.g., sablefish, kelp greenling, cabezon, and lingcod) was higher ($R^2 = 0.69$) than compared to rockfish species; however, the limited number of roundfish species used in this study brings into question whether this relationship would hold if more roundfish species were added. Additionally, life history may have a limited influence on the potential variability in σ during the projection period compared to model structure, estimated versus fixed parameters, and data quantity and quality.

While evidence of a clear linkage between M or h and σ across species groups was not found in this work, that does not mean that these fundamental population dynamics parameters would not be expected, on average, to be highly influential in model uncertainty. Numerous simulation studies have identified that the treatment (e.g., estimated or fixed at misspecified values) of M or/and h are highly influential in the performance of stock assessment models and their ability to estimate population quantities in order to meet management goals ([Lee et al., 2012](#); [Johnson et al., 2015](#); [He and Field, 2019](#); [Hordyk et al., 2019](#)). Additionally, within West Coast groundfish Decision Tables, M is the most commonly used parameter to develop alternative states of nature since it is often identified as being the most uncertain and having large impacts on estimates of spawning output and stock status. Many West Coast stock assessments are unable to estimate M and/or h within the assessment model, and thus estimated uncertainty around unfished or final spawning output is likely under-estimated relative to the true uncertainty.

Additionally, M or h can be used to evaluate the potential risk of overfishing to a population. Under the assumptions of a Beverton-Holt

stock-recruitment relationship, populations with higher h , generally, can sustain higher fishing mortality and have the ability to quickly increase in population size if mortality is reduced. Additionally, M scales the yield curve where species with higher M values have higher yield at maximum sustainable yield relative to species with lower M values (e.g., select rockfish). Hence, the risk of potentially setting future harvest greater than the OFL is not symmetric across M and h ([Mildenberger et al., 2021](#)). If a species with lower h and M (i.e., generally species with slower life history dynamics; [King and McFarlane, 2003](#)), experiences overfishing resulting in the population declining below management targets, typically, a longer period of time is required to rebuild the population relative to species with faster life history dynamics ([Wetzel and Punt, 2016](#)). However, it will generally take several years of overfishing to lead to that result. Additionally, in the short-term, species with higher M can be more susceptible to overfishing given that, generally, a higher portion of the stock is taken in each year at F_{MSY} particularly if there is high scientific uncertainty ([Mildenberger et al., 2021](#)). The option of using the rate of increase in σ of 0.52 M rather than 0.075 across values of M may be reasonable, particularly when M values are far from 0.15 yr^{-1} , the average value across species examined here.

This analysis accounted for within-model uncertainty, but only limited among-model uncertainty, based on defining a low state of nature model that had a final year spawning output at a pre-specified level below the base model. This provides an incomplete picture of the inherent uncertainty around stock assessment results. West Coast groundfish data-rich assessments have variable within-model uncertainty due to the treatment of parameters (e.g., estimated versus fixed) and the quantity and quality of data included in the model. Due to fixing of certain parameters and other assumptions within stock assessment models, the estimated within-model uncertainty measured by the standard deviation around the estimated final spawning output in most, if not all data-rich stock West Coast groundfish assessments is below the associated estimate of σ from [Ralston et al. \(2011\)](#). Additionally, the estimated within-model uncertainty around the OFL in most, if not all, has also been below the corresponding σ of 0.50 from [Privitera-Johnson and Punt \(2020\)](#) analysis. [Bi et al. \(2022\)](#) noted that the largest source of variation in estimated quantities was observed when sequential stock assessments had major changes in parameter assumptions. Similar large variations in estimated quantities for West Coast groundfish stocks has occurred in subsequent assessments when knowledge around key parameters evolves across time ([Thorson and Wetzel, 2015](#); [Gertseva et al., 2021](#)).

An additional source of uncertainty that was not accounted for in this analysis was the stochasticity in annual recruitment during the projection period. Both the low and base states of nature assumed deterministic recruitment with annual recruitment being determined from the stock-recruitment curve during the projection period. Additionally, most assessments fix the recruitment equal to the stock-recruitment curve at the end the modeled period due to a lag in the information within composition data to inform the estimation of realized recruitment (i.e., typically 2–5 years depending upon the growth and the minimum size observed in the composition data), increasing the number of years that the true recruitment can deviate from assumed deterministic recruitment (i.e., typically a total of 12–15 years including the projection period). In long lived stocks, that are late to mature, this assumption may have limited impact within the ten-year projection period if none or only a limited proportion of recruits mature in time to contribute to the spawning output. However, for shorter lived species with young age- or size-at-maturity, if there are strong or weak recruitment years at the start of the projection, there could be a meaningful departure in the realized versus predicted recruitment by the end of the projection period. It is important to note that other studies have shown that for species with high recruitment variability, in the absence of fishing, the probability of the stock falling below management targets and thresholds is non-zero ([Punt et al., 2008](#)). Additionally, the time between subsequent assessments is a key factor in potentially having higher

variations in estimated quantities between assessments particularly for coastal pelagic species which generally have high recruitment variation by year (Bi et al., 2022). Future simulation work could be done to further understand and quantify how life history attributes (e.g., M , h , size-at-maturity, and the magnitude of recruitment variation) can affect divergence in the true population and model predictions. This work could build upon other simulations that measure the impact of model misspecification on end year and short-term harvest estimates (e.g., Johnson et al., 2015; Stawitz et al., 2019).

Implementing science-based approaches for accounting for uncertainty when setting ABCs (or ACLs) to prevent overfishing is a key component of U.S. fisheries management (Methot et al., 2014). In the face of continuing environmental change, linking potential future variation in life history, productivity, and fishing mortality will be critical to understanding and quantifying future uncertainties in fish populations. Generally, assessments of West Coast groundfish assume time-invariant life history. In the presence of short- or long-term directional changes in environmental conditions, determining harvest limits based on static biological reference points may be less precautionary (Travers-Trolet et al., 2020). Dynamically determining biological reference points based on current environmental conditions impacting productivity of fish populations, particularly with the presence of continued directional changes, can reduce the probability of overfishing (Berger, 2019; O'Leary et al., 2020). However, for many assessments, the ability to estimate a time-invariant M or h , let alone the ability to detect time-varying changes, is limited. Focusing on oceanographic conditions and their linkage to recruitment success across the short-term may be a viable first step (Tolimieri et al., 2018; Haltuch et al., 2020). Depending upon the predictive power of identified drivers, they could potentially be used to predict future recruitments a year out at a time based on expected oceanographic conditions. This could reduce one source of uncertainty when using assessment projections to inform management advice. However, large sources of model uncertainty remain that will need to be accounted for when setting harvest limits in order to meet management goals.

As noted above, there are multiple future lines of research to be undertaken to improve projections of both the fish population sizes and to adequately account for uncertainty in estimates of OFLs when setting ABCs to prevent overfish. Nonetheless, in 2019, the PFMC and its SSC adopted the approach of increasing the σ by a constant 0.075 proportion of the σ_{default} for all data-rich (category 1) and data-moderate (category 2) groundfish assessments (which have a different σ_{default}), and this was first implemented for the 2021–2022 management cycle. The SSC and the PFMC emphasized the need to revisit this approach and consider whether to use parameter/life history covariates or information from individual assessments to modify this approach for future management cycles.

Accounting for uncertainty is key when determining harvest limits to prevent overfishing. This work provides an initial approach to explicitly account for additional uncertainty in setting future harvest limits as the time since the last assessment increases for West Coast groundfish, and a framework that could be applied elsewhere. As new populations are assessed and there are new assessments for previously assessed populations there will continue to be opportunities to re-evaluate estimates of time-varying uncertainty to meet the objectives of management and to prevent overfishing.

CRediT authorship contribution statement

Chantel R. Wetzel: Conceptualization, Methodology, Software, Validation, Formal analysis, Writing – original draft, Visualization.
Owen S. Hamel: Methodology, Formal analysis, Validation, Writing – original draft, Visualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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