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REVISITING THE STATUS, DATA, AND FEASIBILITY OF ALTERNATIVE FISHERY MANAGEMENT STRATEGIES FOR CALIFORNIA COASTAL CHINOOK SALMON

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Revisiting the status, data, and feasibility of alternative fishery management strategies for California Coastal Chinook salmon

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Abstract

California Coastal Chinook salmon (CC-Chinook) are listed under the United States Endangered Species Act (Threatened) in 1999 and frequently limit ocean salmon fishery seasons in California and Oregon. Owing to historical and ongoing data deficiencies, a cap on the age-4 ocean harvest rate for Klamath River fall Chinook (KRFC) salmon has been used as a proxy to limit ocean harvest impacts on CC-Chinook. Explorations of the feasibility for developing alternative ocean fishery management strategies were undertaken in 2012 and 2015 (O'Farrell et al., 2012, 2015). Both efforts found that until more comprehensive ocean fishery harvest and spawner escapement data become available for CC-Chinook, few prospects exist for developing a new fishery management strategy. In this report, we re-assess the currently available data and ask whether the data for CC-Chinook are now sufficient to support the development of a new fishery management strategy. We found that there have been substantial improvements to the amount and quality of spawner escapement data over the past ten years, and modest improvements to ocean fishery data from Genetic Stock Identification (GSI) studies. However, as was the case in 2012 and 2015, the current data are insufficient to implement an age-structured assessment using cohort reconstruction methods or development of an abundance index. The development of abundance or ocean harvest indices is also hampered by a lack of consistent CC-Chinook ocean harvest estimates. However, since 2012, new analyses of GSI data presented in this report provide a better understanding of the ocean distribution differences between CC-Chinook and KRFC. Furthermore, we identify a relative harvest rate index that could be estimated given modest increases in escapement and ocean harvest data collection. We end with a reinforcement of the recommendations provided in O'Farrell et al., (2015), with some modifications given the information available at the current time.

Introduction

In 2012, three authors of this report published a NOAA Technical Memorandum titled "California Coastal Chinook salmon: status, data, and feasibility of alternative fishery management strategies" (O'Farrell et al., 2012), which evaluated the California Coastal Chinook salmon (CC-Chinook) spawner escapement and ocean fishery data available at the time. The Evolutionarily Significant Unit (ESU) status and the ocean fishery consultation history were also described. Furthermore, we posed a variety of questions, and attempted to answer these questions, to inform the ability to directly manage ocean fisheries that encounter CC-Chinook. At the time that O'Farrell et al., (2012) was published, and continuing to the time of publication of this report, fishery management for CC-Chinook has relied on the use of a surrogate salmon stock: Klamath River fall Chinook (KRFC). The ocean fishery consultation standard has remained a maximum age-4 ocean harvest rate of 0.16 as of November 2022. The core conclusions of O'Farrell et al., (2012) were that the data available at the time left few options for the development of alternative fishery management strategies and that further work needed to be performed to develop new sources of data that could allow for changes to CC-Chinook fishery management.

Following the publication of O'Farrell et al., (2012), a joint National Marine Fisheries Service (NMFS) and California Department of Fish and Wildlife (CDFW) workshop titled "California coastal Chinook salmon fishery management: future prospects" was convened. A second Technical Memorandum documenting the findings of that workshop was then published (O'Farrell et al., 2015). The goals of the workshop were to "...identify the level of information necessary to allow for development of an abundance-based fishery management (ABM) approach and evaluate the feasibility of collecting that level of information for the CC-Chinook salmon Evolutionarily Significant Unit (ESU)." ABM refers to a fishery management approach where allowable exploitation rates can vary with stock abundance. General conclusions from the workshop were (1) a substantial increase in data would be needed to implement an abundance-based management approach, (2) there are substantial technical difficulties to escapement estimation within the ESU, and (3) there are substantial difficulties for implementing a marking and tagging program within the ESU. However, an ABM approach could be possible if these difficulties could be overcome, and steps toward resolving these difficulties were identified.

In the ten years following publication of O'Farrell et al., (2012) there have been several efforts to increase the information level for CC-Chinook salmon. In this report, we describe the changes in data richness for spawner escapement and ocean harvest of CC-Chinook. We update information regarding the CC-Chinook ESU status and ocean fishery consultations. We then re-evaluate the questions posed in the 2012 Technical Memorandum regarding the feasibility of alternative fishery management strategies, such as ABM. Finally, we revisit the feasibility of developing an alternative management approach for CC-Chinook, given the data currently available or that might be collected in the near future.

ESU status

California Coastal Chinook salmon (CC-Chinook) were originally considered part of a larger ESU that included coastal populations in southern Oregon and northern California (Myers et al., 1998). However, a subsequent biological review concluded that CC-Chinook constituted a distinct ESU that included populations from Redwood Creek, Humboldt County, south to and including the Russian River in Sonoma County (NMFS, 1999). This ESU was listed as threatened in 1999 (64 FR 50394; see O'Farrell et al., 2012 for elaboration on the listing history).

Since the original listing, NMFS has conducted status review updates every five years, as required by law. The first of these reviews, concluded in 2005, was performed by a Biological Review Team (BRT), which reaffirmed the threatened status of CC-Chinook salmon (Good et al., 2005). Beginning in 2011, these 5-year reviews have proceeded in two phases. First, NMFS Southwest Fisheries Science Center has performed viability assessments, collecting and analyzing new information regarding ESU viability and evaluating extinction risk relative to viability criteria outlined in Spence et al., (2008) and, beginning in 2016, recovery criteria defined in the recovery plan for CC-Chinook salmon (NMFS, 2016a). In addition to evaluating extinction risk, viability assessments consider information related to ESU boundary delineation.

Next, biologists from NMFS West Coast Regional office consider these viability assessments in conjunction with new information related to five ESA listing factors, including (1) the present or threatened destruction, modification, or curtailment of the species' habitat or range; (2) overutilization for commercial, recreational, scientific, or educational purposes; (3) disease or predation; (4) inadequacy of existing regulatory mechanisms; or (5) other natural or man-made factors affecting the species' continued existence. Based on this analysis, the region staff determine if the ESU should have its status changed or be removed from the endangered species list.

The three most recent viability assessments have all concluded that new information collected since prior assessments was insufficient to conclude that extinction risk for CC-Chinook salmon had changed significantly. Williams et al., (2011) concluded that there was no new evidence to suggest a significant change in extinction risk, but noted that the lack of population-level estimates of abundance for the majority of independent populations hindered the assessment, and that until more comprehensive data are collected, the viability of CC-Chinook salmon remained highly uncertain. They also cited the apparent extirpation of populations in the North-Coastal and Coastal diversity strata (with the exception of the Russian River) as significant concerns for maintaining ESU connectivity. The 2016 viability assessment likewise concluded that there was a lack of compelling evidence to suggest a significant change in extinction risk (Spence 2016). The review again highlighted the scarcity of available long-term spawner data, but noted that new information generated by implementation of California's Coastal Monitoring Plan (CMP; Adams et al., 2011) had improved understanding of population status, particularly for populations in the North-Coastal and Coastal diversity strata, where spawning ground surveys demonstrated that small numbers of Chinook salmon continued to spawn in most of these watersheds. Based on the review of biological information and the five listing factors, NMFS determined no change in listing status was warranted after each of these reviews (NMFS, 2011, 2016b; 76 FR 50447, 81 FR 33468).

The most recent viability assessment, which considered data up through the 2018–2019 spawning season, concluded that overall extinction risk remained moderate for the ESU and had not changed appreciably since the prior review (Spence, in press). The report noted improved data availability since the prior viability assessment, with new sonar-based monitoring for several key independent populations and continued spawning ground surveys for Mendocino Coast populations. As of this writing, the final status review has not yet been published by NMFS West Coast Region.

Available spawner data

Prior status reviews and viability assessments for CC-Chinook salmon have noted the paucity of long-term population-level estimates of abundance for CC-Chinook populations anywhere in the ESU (Myers et al., 1998; Good et al., 2005; Williams et al., 2011). These early assessments relied to a great extent on (1) spawner indices that represented only a small portion of the population or available spawning habitat and (2) counts such as those made at Van Arsdale station, which, in addition to representing only a small proportion of the Upper Eel River population, are believed to be strongly influenced by year-to-year variation in hydrological conditions that affect the ability of fish to reach the station and pass above it (Lacy et al., 2016). For these early reviews, little information was available on the status of populations along the Mendocino Coast.

The availability of spawner information has improved considerably with implementation of California's CMP in various regions (Adams et al., 2011). For Chinook salmon, increased emphasis has been placed on obtaining population-level estimates of abundance for "independent populations" that are identified as important components of ESU recovery. Independent populations are defined as those that historically had a high probability (>95%) of persisting at time scales of 100 years or more (Bjorkstedt et al., 2005). In some cases, funding for programs has been intermittent, which reduces the utility of data for fishery management purposes. But overall, data availability is much greater than it was when O'Farrell et al., (2012) was published. Here we review the current status of various monitoring programs that currently produce, or have recently produced, information on Chinook salmon spawner abundance that was used in the most recent viability assessment (Table 1).

Table 1. California Coastal Chinook freshwater data sources examined in the most recent 5-year viability assessment, which included data through the 2018–2019 spawning season (Spence, in press). Populations are as defined by Bjorkstedt et al., (2005), as modified by Spence et al., (2008), and are arranged from north to south in the table.

Location	Population	Data
Redwood Creek	Redwood Creek	Adult spawners have been estimated using a sonar camera in most years since 2010. Estimates have averaged 2,946 fish (range 1,455–4,541) over the period of record with no significant trend. Additionally, standard spawning ground surveys have been conducted in all but one year since 2011. Surveys occur in a sample frame designed for coho salmon and so do not encompass all potential Chinook spawning habitat. Redd estimates have averaged 774 (range 290–1,063) from 2011–2019.
Mad River	Mad River	Adult spawners have been estimated using a sonar camera in most years since 2014. Estimates have averaged just 6,520 fish (range 2,169–12,667) over the period of record.
Freshwater Creek	Humboldt Bay	Partial counts have been made at a weir since 1994. These are partial both because Freshwater Creek is considered only part of the Humboldt Bay population and because fish may pass uncounted through (jacks) or around the weir, especially during periods of high flow. Counts have averaged 29 fish (range 0–154) over the period of record and have declined in recent years, likely due to the discontinuation of a local hatchery program in the watershed in the early 2000s.
Eel River	Upper Eel River	Counts of adult spawners have been made at Van Arsdale Station since the late 1940s, which represents a small and potentially variable (depending on streamflow) proportion of the total Upper Eel River population. Counts prior to 1997 are confounded both by variation in stream flows and trap operation, as well as the influence of hatchery activities; thus, only counts since 1997 were used in the most recent viability assessment. Since 1997, natural origin Chinook have been counted separately and have averaged 627 fish (range 26–3,471). Additionally, a new sonar-based program was initiated in 2018, which provides a much more robust estimate of the Upper Eel River Chinook population. Estimates over four years have averaged 4,435 fish (range 3,844–4,953).
Mattole River	Mattole River	Spawner surveys have been conducted since 2013, replacing a spawner "index" that was based on surveys that varied among years in spatial and temporal extent. Surveys occur in a sample frame designed for coho salmon and so do not encompass all potential Chinook spawning habitat. Redd estimates within the frame have averaged 862 (range 418–2,202) from 2011–2019.
Mendocino Coast – North-Central Coastal diversity stratum	Ten Mile, Noyo, Big, and Albion	Spawner surveys have been conducted since 2009, with one missing year for the Big and Albion rivers. Population estimates for individual populations have usually ranged from zero to a few tens of fish; however, estimates for the Ten Mile and Noyo rivers have neared or exceeded 100 fish on occasion. The aggregate abundance for the North-Central Coastal stratum has averaged 101 fish (range 0–728) between 2009 and 2022.

Location	Population	Data
Mendocino Coast – Central Coastal diversity stratum	Navarro and Garcia rivers	Spawner surveys have been conducted since 2009, with one missing year for both populations. Population estimates for individual populations have usually ranged from zero to a few tens of fish; however, estimates for the Garcia River have exceeded 100 fish on several occasions. The aggregate abundance for the Central Coastal stratum has averaged 37 fish (range 0–178) between 2009 and 2021.
Russian River	Russian River	Counts have been made using video and, more recently sonar, since 2000, though the camera was not operable for portions of the 2015, 2016, and 2017 seasons, so estimates were derived by other means and are not directly comparable to other years. The long-term average abundance is estimated at 2,723 fish (range 625–6,739).

In addition to the sonar-based program, conventional spawning ground surveys have been conducted in Redwood Creek in all but one year between the 2010–2011 and 2017–2018 spawning seasons. The estimates produced are basin-wide redd abundances (uncorrected for fish/redd ratios). The sampling frame includes all potential Chinook spawning habitat; however, high flows can limit the frequency of surveys in the mainstem of Redwood Creek, leading to potential underestimation of Chinook salmon redds (Seth Ricker, CDFW, pers. comm.). Redds that cannot definitively be assigned to species are assigned using a known nearest neighbor (KNN) algorithm (CDFW 2021). This monitoring program has estimated an average of 774 redds over nine years of record (Table 2).

Redwood Creek is also the site of a life-cycle monitoring (LCM) station, which produces estimates of outmigrating Chinook smolts using mark-recapture methods at a rotary screw trap. These data enable estimation of survival indices including adult-to-adult survival, freshwater survival (adult-to-smolt), and marine survival (smolt-to-adult) (Deibner-Hanson, 2019). This station represents the only LCM station in the North Coast region that is in a watershed with a sizable Chinook salmon population.

In the Mad River basin, estimates of adult Chinook salmon abundance have been produced since the 2013–2014 spawning season using a long-range ARIS sonar located at river mile 7.0. As with the Redwood Creek program, estimates are derived by analyzing 20 minutes of every hour of footage, and species apportionment is achieved primarily using snorkeling, augmented with angler creel surveys (past and present), hook-and-line sampling, and professional judgment (Sparkman, 2020). In the first six years of operation, the average estimated abundance of Chinook salmon was 6,520 fish (Table 2). This monitoring effort provides far more reliable estimates of abundance than the long-running Canon Creek spawning index, which produced an index based on maximum live-dead counts observed during the spawning season and was recently discontinued.

Spawning ground surveys using CMP protocols have been conducted in the Mattole River since the 2012–2013 spawning season. Prior to this, a Chinook salmon redd index was produced based on spawning ground surveys that varied among years in spatial extent and intensity. Since then, the sampling program has been more refined, but the sampling frame has varied, focusing on coho salmon spawning areas in some years, but including additional reaches within the Chinook sampling frame in others. The estimates produced are basin-wide redd abundances (uncorrected for fish/redd ratios). Redds that cannot definitively be assigned to species are assigned using a KNN algorithm (CDFW, 2021). Over the seven years for which data are currently available, redd estimates have averaged 862, though analysis of trends is not considered appropriate due to variation in the sampling frame through time (Table 2).

Table 2. Population estimates for California Coastal Chinook salmon considered in the most recent NMFS 5-year viability assessment (Spence in press). ND indicates no data available due to lack of funding; NA indicates data were collected but final estimates are not currently available. SGS indicates estimates based on spawning ground surveys.

Year	Redwood Creek (sonar)	Redwood Creek (SGS[1])	Mad River (sonar)	Freshwater Creek (count)	Eel River mainstem (sonar)	Eel River Van Arsdale (count)	So. Fork Eel River (sonar)	So. Fork Eel River (SGS[1])	Mattole River (SGS[1])	Mendocino NCC diversity stratum (SGS[2])	Mendocino CC diversity stratum (SGS[2])	Russian River (video)
1997-1999						91						
2000-2005				90		443						3,839
2005-2006				22		620						2,607
2006-2007				18		697						3,407
2007-2008				7		478						2,021
2008-2009				2		496				102	0	1,129
2009-2010	2,438			2		518				58	16	1,800
2010-2011	ND	783		19		2314		1,829		33	14	2,502
2011-2012	1,455	866		1		2436		68		84	0	3,173
2012-2013	3,401	940		2		3471		855	418	28	9	6,739
2013-2014	3,487	963	2,169	0		214		233	988	0	0	3,152
2014-2015	ND	1063	7,489	8		588		781	535	168	0	1,420
2015-2016	1,839[3]	740	5,786	2		102		418	331	50	178	3,020
2016-2017	ND	ND	7,186	4		435		1,458	929	728	165	1,062
2017-2018	4,541	850	12,667	9		232		867	2,202	115	28	2,063
2018-2019	2,919	468	3,825	1	3,844	94	3,831	404	633	50	ND	1,219
2019-2020	2,380	290	NA	NA	4,231	156	2,441	135	NA	0	0	922
2020-2021	ND	ND	NA	NA	4,710	63	ND	14	NA	0	0	625
2021-2022	ND	ND	NA	NA	4,953	457	ND	NA	NA	0[4]	65	NA
Average	2,946	774	6520	29	4,435	627	3,136	642	862	101	37	2723

1. Estimates produced are frame-wide estimates of total redd abundance, uncorrected for spawner:redd ratios
2. Estimates produced are frame-wide estimates of total spawner abundance, derived by estimating total redd abundance and then multiplying by spawner:redd ratios generated at life-cycle monitoring stations, or by 2.5 for the Mendocino Coast diversity strata.
3. Partial estimate due to extensive downtime for equipment (J. Deibner-Hanson, Stillwater Sciences, pers. comm.); value not included in the average.
4. Estimate is preliminary and subject to change.
5. Provisional estimate for this stratum based on redd assignments is 0; however, some live Chinook and carcasses were observed (S. Gallagher, CDFW, pers. comm.).

Additional Chinook adult spawner information has been collected at the Freshwater Creek life cycle monitoring station (LCM) since 2000–2001. This program produces a partial count for the Humboldt Bay independent population, which also includes other tributaries to Humboldt Bay (i.e., Jacoby Creek, Elk River, and Salmon Creek). The counts are incomplete because fish can pass around the weir under high flow conditions, and smaller jacks can pass through the weir. Over the period of record, the average Chinook count has been 29 fish, though in the last 10 years of the time series, fewer than 5 fish have been observed on average; this decrease follows the termination of a small hatchery program that operated in Freshwater Creek up until the early 2000s (Table 2). Spawning ground surveys have also been conducted across the larger Humboldt Bay Region since the 2010–2011 spawning season, but these surveys typically produce few observations of Chinook salmon.

Two efforts to monitor Chinook salmon and other salmonids in the South Fork Eel River (considered part of the lower Eel River independent population) have been initiated in the last 12 years. A sonar camera was operated from mid-November through mid-to-late May during the 2018–2019 and 2019–2020 spawning seasons. The installation site in 2018–2019 was originally near the confluence of the mainstem Eel River, below the first major tributary (Bull Creek); however, in 2019–2020 it was moved 10 miles upstream to take advantage of onsite power, which excluded some potential spawning habitat. For this program, species apportionment methods based on direct observation have not been fully developed. Instead, it is assumed that fish passing the station prior to January 1 are all Chinook salmon, whereas those arriving after January 1 are either coho salmon or steelhead. This assumption likely introduces some bias in the Chinook salmon estimates, as coho salmon (and probably steelhead) likely enter the South Fork Eel prior to January 1 in most years. But the extent of that bias remains unknown. The sonar program was not funded in the 2020–2021 or 2021–2022 seasons, but may be resumed in 2022–2023. Estimates of Chinook adults have averaged 3,136 in the two years of operation, but likely are biased high due to the occurrence of coho salmon in the South Fork prior to January 1 (Table 2).

Standard spawning ground surveys have also been conducted in the South Fork Eel River basin since the 2010–2011 spawning season. The sample frame for these surveys has generally targeted coho salmon, excluding portions of the mainstem South Fork Eel River, which supports Chinook salmon spawning but cannot be safely or effectively surveyed during typical winter flows. Thus, the estimates underestimate the total number of Chinook spawners in the subbasin. The estimates produced are frame-wide redd abundances (uncorrected for fish/redd ratios), and have averaged 642 over the 11 years of record (Table 2). These monitoring efforts have replaced the Sproul Creek maximum live/dead counts that were previously the only available information on Chinook salmon numbers in the South Fork Eel, but represented only a small portion of the available spawning habitat.

The North Mountain Interior diversity stratum includes the Upper Eel River fall-run Chinook population and the portion of the Lower Eel River population that includes the Van Duzen River. A sonar-based program for the Upper Eel River population was initiated in the 2018–

2019 season and has continued to the present, with the camera being operated from mid-November to May in most years (Table 1). The camera has been installed just above the confluence with the South Fork Eel River, so potentially represents the entirety of the Upper Eel River population. As with the South Fork Eel River sonar program, species apportionment is not based on direct observation but instead on the assumption that fish passing the station prior to January 1 are Chinook salmon and those passing afterwards are other species. In this instance, the assumption likely leads to less bias in Chinook estimates compared with the South Fork Eel River, since few coho salmon return to the Upper Eel River, and the degree of overlap between the Chinook and steelhead run timing is considered small. In at least one year (2019-2020), September rains resulted in some Chinook passing through the location before the DIDSON camera was installed; however, the fraction of the total population that passed early was believed relatively small. In the four years of operation, the average Chinook estimate is 4,435 fish (Table 2).

In addition to the sonar program, counts of Chinook salmon have been made at Van Arsdale station at river mile 156 in the Upper Eel River since the late 1940s. The number of Chinook salmon that reach the station in a given year is considered highly dependent on flow conditions (Lacy et al., 2016), including mandated flow releases that have been in effect since 2004. Additionally, early counts were influenced by both station operations and the planting of hatchery fish, which ceased in 2004. Counts of returning natural-origin Chinook salmon have been kept separate from hatchery fish since the 1996–1997 spawning season, and have averaged 627 fish during this period (Table 2).

Maximum live-dead counts have been made in Tomki Creek, a tributary of the Upper Eel River, since the 1970s. Beginning in 2000–2001, changes were made to estimation methods, so numbers before and after this change are not comparable. As with Van Arsdale station, the numbers of Chinook entering Tomki Creek appear to be dependent on flow conditions during the year. Because of this dependence and the fact that Tomki Creek represents only a small portion of the total Upper Eel River population that is monitored through the sonar program, the Tomki Creek data were not reported in the most recent viability assessment.

The North-Central Coastal diversity stratum includes watersheds from Usal Creek in the north to Big Salmon Creek in the south. Within this stratum, three independent Chinook salmon populations (Ten Mile, Noyo, and Big rivers) and one dependent population (Albion River) are currently monitored as part of the Mendocino Coast CMP monitoring effort (Table 1). The Mendocino Coast sampling frame covers both the North-Central Coastal diversity stratum and portions of the Central Coastal diversity stratum, and includes 339 reaches that are potential spawning habitat for coho salmon and steelhead, of which 113 are also identified as potential Chinook spawning habitat. Redds that cannot definitively be assigned to species in the field are assigned to species using a KNN algorithm (CDFW, 2021). Total population estimates are derived by multiplying redd estimates by 2.5, as spawner:redd ratios could not be calculated from life-cycle monitoring stations, which are located in smaller watersheds or subbasins upstream of where the majority of Chinook spawning typically occurs (McClary et al., 2019).

Overall, Chinook numbers in all of these watersheds tend to be small, typically ranging from no fish to a few tens of fish. However, occasionally Chinook spawner estimates in the Ten Mile and Noyo rivers have approached or exceeded 100 fish (Table 2). The composite estimate for the North-Central Coastal stratum has averaged 101 fish over the 14 years of record, though it should be noted that in 2018–2019, neither the Big River nor Albion River were surveyed due to insufficient funding.

The Central Coastal diversity stratum includes watersheds from the Navarro River in the north to the Russian River in the south. The Mendocino Coast CMP program also produces Chinook salmon adult estimates for two independent populations in this stratum: the Navarro and Garcia river populations (Table 1). Methods are the same as those described for the North-Central Coastal stratum, with redd assignments to species being made using a KNN algorithm, and expansions from redd counts to population estimates produced by multiplying redd estimates by 2.5. Chinook salmon occurrence in these two watersheds has been more sporadic, with Chinook being recorded in only 2 of 12 years in the Navarro River and 6 of 12 years in the Garcia River. Stratum-level estimates of Chinook salmon abundance (including only the Navarro and Garcia rivers) have averaged 37 fish over the 13 years of record (Table 2).

Estimates of Chinook salmon for the Russian River population have been made since the 1999–2000 spawning season based on footage from video cameras installed on fish ladders at Mirabel Dam on mile 24.7 of the mainstem Russian River. Since 2011, sonar cameras have been paired with the video cameras to improve accuracy during periods of high turbidity (Lacy et al., 2016). Counts during three recent years (2014–2015 to 2016–2017) were interrupted by repairs to the dam and counting station (Sonoma Water and Sea Grant, 2022); thus, estimates were derived using information from other monitoring efforts and are not considered comparable to other years. Some spawning by Chinook salmon does occur downstream of the video cameras, but the majority is believed to occur upstream. In the 22 years of operation, Chinook counts have averaged 2,723 fish (Table 2).

In addition to the adult counting station, a rotary screw trap has been operated below the fish ladder in order to enumerate outmigrating Chinook smolts (and other salmonids) since 2013 (excepting 2015 and 2016 when dam repairs were taking place). The period of trap operation, however, has been inconsistent among years, which has likely contributed to both bias in smolt estimates (when a significant portion of the smolt run is missed) and low precision of estimates (due to high within-season variability in trap efficiencies) (Sonoma Water and Sea Grant, 2022). More consistent trap operation would increase the precision of estimates and enable estimation of smolt-to-adult survival rates.

Ocean fishery consultation history

Portions of this section have been excerpted from O'Farrell et al., (2012).

The 2000 Biological Opinion established the CC-Chinook consultation standard consisting of a maximum *preseason-projected* KRFC age-4 ocean harvest rate of 0.17 (NMFS, 2000). The harvest rate cap was reduced to 0.16 shortly thereafter owing to modifications made to the KRFC cohort reconstruction model (KRTAT, 2002).

Reductions in ocean salmon fisheries as a result of KRFC harvest allocations and Sacramento River winter Chinook conservation concerns occurred prior to the development of the CC-Chinook consultation standard in 2000. In 1993, establishment of 50:50 sharing of the tribal and non-tribal allowable harvest of KRFC permanently constrained ocean fisheries that impact the stock. In particular, commercial fisheries in the Klamath Management Zone (the combined KO and KC management areas) and Fort Bragg (FB) were sharply reduced (see Figure 1 for a current map of ocean fishery management areas). In 1996, the first constraints on ocean fisheries to protect Sacramento River winter Chinook were introduced, which resulted in reduced fishing effort south of Point Arena, California in many years. Because CC-Chinook were thought to have an ocean distribution somewhat intermediate between KRFC and Central Valley Chinook (NMFS, 2000), it was inferred that harvest rates on CC-Chinook declined as a result of these ocean fishery constraints. Furthermore, it was concluded that KRFC ocean harvest rates should be comparable to CC-Chinook total harvest rates because there is no legally retained river harvest of CC-Chinook and their ocean distributions likely have substantial overlap.

The combined constraints to ocean fisheries resulting from KRFC tribal/non-tribal sharing and Sacramento River winter Chinook conservation concerns since 1996 led to a focus on the fisheries that occurred between 1996 and 1999 when developing the CC-Chinook consultation standard. In this four-year time period, the maximum postseason-estimated KRFC age-4 ocean harvest rate estimate was 0.17 (range: 0.11–0.17). This result, coupled with limited information indicating that spawner abundance of some CC-Chinook populations had improved since 1996, suggested that ocean fisheries operated at the 1996–1999 scale were sufficient to allow for persistence of CC-Chinook when they are at low abundance (NMFS, 2000). This cap on the preseason-projected KRFC age-4 ocean harvest rate was also justified by the concern that in years of high KRFC abundance, the preseason-projected age-4 ocean harvest rate could otherwise exceed 0.20, which was identified in NMFS (2000) as a level that had the potential to appreciably reduce the likelihood of the survival and recovery of CC-Chinook. Such a scenario would likely result in increased CC-Chinook ocean harvest rates if there was expansion of fishing in California north of Point Arena.

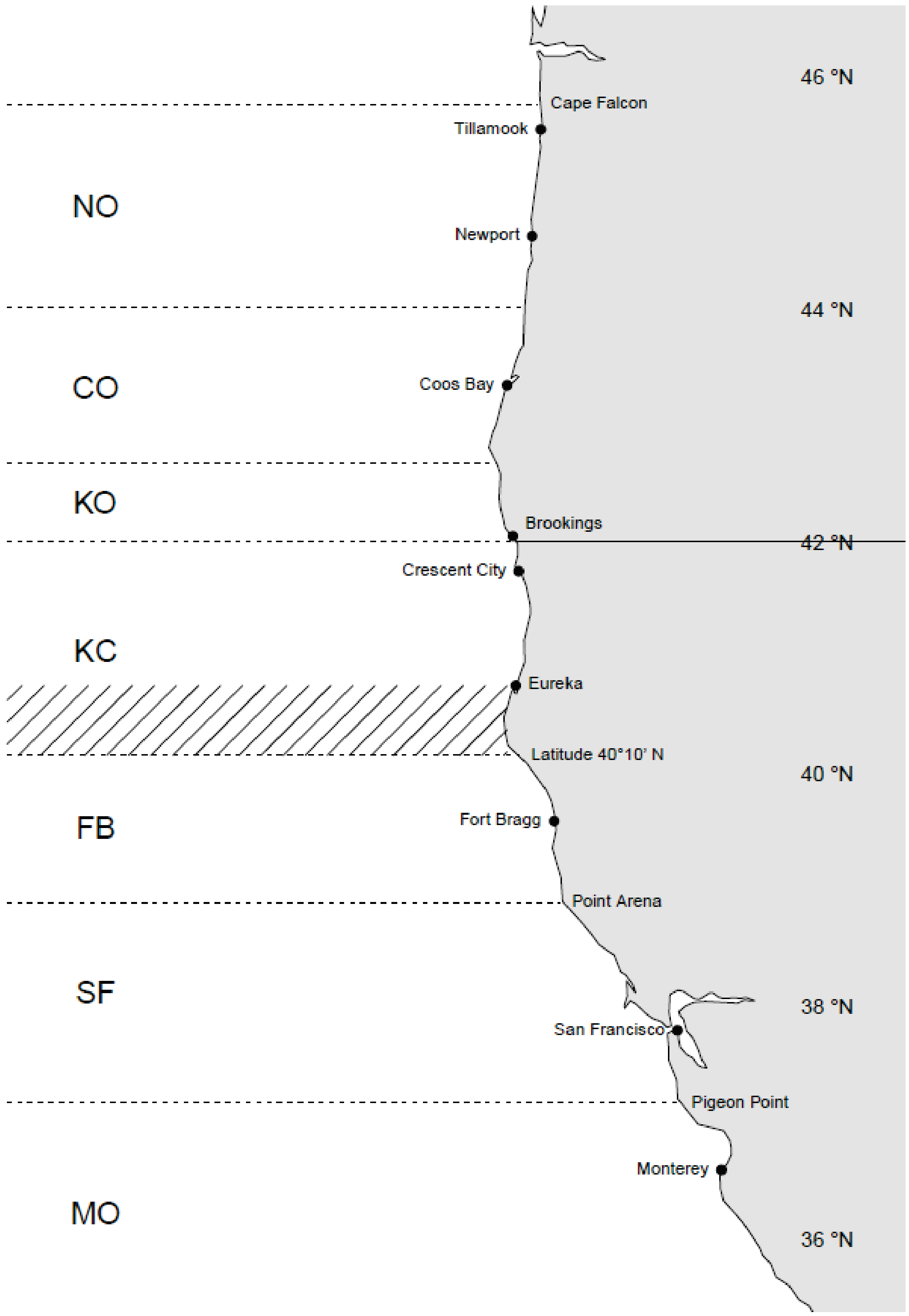

Figure 1. Map of the seven ocean fishery management areas, as well as major ports and landmarks, for the area south of Cape Falcon, Oregon. The area from Humboldt South Jetty (Eureka) to Latitude 40°10' N, denoted by the striped area, has been closed to commercial salmon fishing since 1992. The KO and KC management areas comprise the Klamath Management Zone.

In addition to limiting the preseason-projected age-4 ocean harvest rate of KRFC, the Reasonable and Prudent Alternative in the 2000 Biological Opinion stipulated that (1) NMFS must continue to evaluate the KRFC harvest rate as an indicator of the harvest rate on CC-Chinook and (2) NMFS shall cooperate with the states and PFMC to ensure that ocean fisheries are monitored and sampled for stock composition, including coded-wire tags (CWTs) and other biological information (NMFS, 2000).

In 2005, a reinitiation of consultation was undertaken following a very high postseason estimate of the KRFC age-4 ocean harvest rate for 2004 ocean salmon fisheries (NMFS, 2005). Although the CC-Chinook consultation standard at the time specified the cap on the preseason-projected KRFC age-4 ocean harvest rate of 0.16, and the Klamath Ocean Harvest Model (KOHM) projected harvest rate in 2004 was 0.15, the postseason estimate of the rate in 2004 was 0.52. As a result of the preseason projection being exceeded by such a large amount, an effort to determine the causes of this overage was undertaken. This analysis identified the primary cause to be underpredicted contact-rates per-unit-effort in the KOHM (NMFS, 2005). The 2005 consultation therefore specified that the PFMC and Salmon Technical Team would modify the KOHM for the 2006 season to more heavily weight recent-year contact-rate-per-unit effort estimates for forecasting purposes. In practice, this led to using data from 2003 forward for making contact-rate-per-unit-effort predictions for the commercial fishery in the FB, San Francisco (SF), and Monterey (MO) management areas (PFMC, 2006). This practice of using data from 2003 forward for forecasting contact-rates-per-unit-effort in the KOHM for the commercial salmon fishery in those areas continued through the 2020 ocean fishery planning process.

In 2021, after eight consecutive years of under-predicting the KRFC age-4 ocean harvest rate, and substantial under-predictions in 2018–2020, another modification was made to the KOHM. Contact rates per unit effort were again estimated using a shorter, more contemporary data range. For management year 2021, contact rates per unit effort in both commercial and recreational fisheries were estimated using data from 2013–2020. A description of the contact rate per unit effort modification, and the analysis supporting it, is described in Appendix B of PFMC (2021).

Despite the contact rate per unit effort modifications made in 2021, the postseason estimate of the 2021 KRFC age-4 ocean harvest rate was again underpredicted by a substantial margin. For 2022 fishery planning, the data range used for estimating contact rates per unit effort was further shortened to 2015–2021 in both the commercial and recreational fisheries in an attempt to better characterize contemporary fishery encounters and harvest (PFMC, 2022a, Appendix D). In March of 2022, NMFS reinitiated consultation on the effects of fisheries managed under the Pacific Coast Salmon Fishery Management Plan (PFMC, 2022b) on CC-Chinook. NMFS completed a biological opinion (NMFS, 2023) on February 28, 2023 and determined that the conservation objective, implemented as a *postseason* limit of 0.16 on the KRFC age-4 ocean harvest rate, was not likely to jeopardize the continued existence of CC-Chinook.

However, the KRFC age-4 harvest rate was again underpredicted by a substantial amount in 2022 (PFMC, 2023). NMFS again reinitiated consultation in March 2023, with the expectation that the consultation would be completed prior to implementation of the 2024 salmon fishery regulations. Outcomes of that consultation are unknown at the time of this writing.

Available ocean data

Ocean fishery data richness for CC-Chinook remains low relative to many other managed salmon stocks in California. However, considerably more data are available or have been analyzed than when O'Farrell et al., (2012) was written. The data that do exist include CWT recoveries from ocean fisheries and genetic stock identification (GSI) data from both commercial and recreational ocean fisheries. GSI data have been collected at-sea, at the docks as part of ocean fishery sampling, and in times and areas closed to salmon retention. However there has not been a sustained effort to collect GSI data from all relevant times, areas, and fishery sectors where CC-Chinook are likely to be harvested.

O'Farrell et al., (2012) described the history of coded-wire-tag tagging for populations within the CC-Chinook ESU, and noted that CWT recoveries from fish released in the Eel River basin were used in the 2000 Biological Opinion. The analysis of these data suggested that the distribution of Eel River Chinook was more southerly than KRFC, but more northerly than Central Valley Chinook. Following the 2000 Biological Opinion, there have been few CC-Chinook CWT recoveries (287 ocean recoveries and zero freshwater recoveries). The availability of CWT data for CC-Chinook has not increased since the publication of O'Farrell et al., (2012); there have been no coded-wire tagged CC-Chinook released or recovered since that time as hatchery programs in the ESU were phased out by the early 2000s.

GSI can allow for genetic-based estimates of the stock of origin for fish sampled in ocean fisheries. For CC-Chinook, the current GSI baseline (Clemento et al., 2014) is able to discriminate between Eel River and Russian River fish with a high degree of confidence, with other populations within the ESU likely to be assigned to the Eel River population (Carlos Garza, NMFS Southwest Fisheries Science Center, personal communication). GSI data were collected via dockside sampling of the recreational fishery in California by CDFW for 1998–2002, and opportunistically by willing commercial salmon fishermen along much of the California and Oregon coast since 2006, with data collected through 2019 analyzed and presented later in this document. In 2010, the GSI project sampled nearly all months and management areas south of Cape Falcon, Oregon, both during the course of regular commercial fisheries and as non-retention sampling in months and areas closed to commercial fishing, and the age composition of the 2010 samples from Oregon and California was estimated by reading scales (see Kormos et al., 2011, for California age structure estimates). In other years, the GSI sampling project was conducted primarily during months and areas open to commercial fishing, but age compositions have rarely been estimated, and coverage has been patchy in time and space. These data have yielded estimates of the contribution rate of CC-Chinook to the sampled catch (or fish sampled and released in the case of non-retention sampling) at fine spatiotemporal scales, which are discussed further in the following sections. However, there are some potential limitations to the inferences that can be derived from these data. For example, it is unclear how well GSI data collected during non-retention sampling can be expected to represent the harvest stock composition that would have resulted from normal retention fishing (see Bellinger et al., 2015 for further discussion of this issue). In addition, it is unclear how well fishermen participating in voluntary GSI sampling represent the entire fleet. Nonetheless, these data have been used in comparisons of size-at-age and spatial distributions of CC-Chinook and Klamath Chinook (Satterthwaite et al., 2014) and other published studies of ocean catch composition and stock-specific spatial distributions (Bellinger et al., 2015; Satterthwaite et al., 2015), as well as more recent, currently unpublished analyses. A summary of these spatial distribution results is provided in the “Can abundance of CC-Chinook be inferred from other stocks?” section below.

A relatively small amount of GSI data exists from ocean fisheries prior to 1998, including GSI sampling on small temporal and spatial scales in California and Oregon and test fisheries in California (Winans et al., 2001).

Feasibility of alternative fishery management strategies

The feasibility of implementing an alternative fishery management strategy for CC-Chinook was evaluated in O’Farrell et al., (2012). That evaluation was primarily focused on the ability to implement an abundance-based fishery management approach, where allowable fishing mortality rates depend on stock abundance. In the sections that follow we revisit the set of questions posed in O’Farrell et al., (2012) and evaluate whether the conclusions from the 2012 Technical Memorandum have changed in the intervening ten years. The text from O’Farrell et al., (2012) is reprinted in italics in the sections below.

Can total escapement of CC-Chinook be estimated?

Aggregate escapement estimates for the CC-Chinook ESU cannot be made owing to relatively low levels of sampling and a lack of randomized sample site selection across the available spawning habitat. For those areas that are sampled, measures of escapement are nearly all confined to indices of relative abundance, some of questionable quality, and they therefore do not provide population or region-level estimates of total escapement (Williams et al., 2011). At the individual population level, the video counts made on the Russian River potentially provide an estimate of Russian River escapement, although the amount of spawning downstream of the video counting station is not estimated.

The level of escapement data for CC-Chinook has increased substantially since 2012. However, a lack of consistent funding for key monitoring programs (e.g., Redwood Creek and the South Fork Eel River) has resulted in data series gaps. There has been a lack of monitoring in certain important watersheds, such as the Van Duzen River. However, some of these gaps have been addressed recently with SONAR monitoring systems. SONAR-based monitoring of the mainstem Eel River, upstream of the South Fork, has been ongoing since the 2018–2019 run year, while SONAR monitoring commenced for the Middle Fork Eel River in 2021–2022 (Smith, 2022) and the Van Duzen River in the 2022–2023 run year (Seth Ricker, CDFW, personal communication). While there have been substantial increases in escapement monitoring, a total escapement estimate for the ESU cannot be made at this time. However, such an estimate is not necessarily a prerequisite for the development of an alternative fishery management strategy.

Can ocean harvest of CC-Chinook be directly estimated?

Stock proportions are estimable from GSI data, and the product of local stock proportions and local total catch could yield an estimate of total CC-Chinook harvest. Uncertainty in total CC-Chinook catch arising from genetic assignment and sampling uncertainty can be quantified using the methods described in Satterthwaite et al., (In prep.), which have been incorporated into the computer program gsi sim2. However, expanding stock proportions from a genotyped subsample to estimate total CC-Chinook harvest requires that the genotyped subsample is representative of the fishery management stratum harvest as a whole. Thus, the fishermen who participate in GSI sampling must be fully representative of the fleet, in terms of where and how they fish, or the GSI samples need to be collected via a comprehensive dockside sampling scheme similar to that currently employed for CWTs.

While GSI data can, in theory, allow for estimation of CC-Chinook ocean harvest in an adequately sampled management stratum, estimates of total ocean catch cannot be made from the data currently available. The estimation of total catch would require stock proportion estimates from all month/area/fishery strata within the CC-Chinook range. These GSI data do not currently exist. The low proportion of CC-Chinook expected in many strata (Winans et al., 2001) adds to the difficulty, as small proportions are difficult to estimate with adequate precision unless sample sizes are impractically large (Allen et al., In prep.).

Since the publication of O’Farrell et al., (2012), there have been improvements to methods used to quantify uncertainty in catch estimates and additional GSI data have been collected. Uncertainty in total CC-Chinook catch arising from genetic assignment and sampling uncertainty can now be quantified using published methods (e.g., Satterthwaite et al., 2014; Jensen et al., 2022). With regard to new GSI data, genotype assignments corresponding to dockside genetic sampling from California recreational fisheries conducted by CDFW in 1998–2002 are now available, and collection of genetic data by commercial fishermen in California and Oregon has continued to varying degrees. In addition, tissue samples have been collected (but not analyzed) in more recent dockside sampling of both the recreational and commercial fisheries in California.

Estimation of total ocean catch using GSI data would require sufficient sampling from all month/area/fishery strata where significant numbers of CC-Chinook are harvested. Given existing data gaps, estimates of total ocean catch cannot be made, as was the case at the time of publication of O’Farrell et al., (2012). Furthermore, the low proportion of CC-Chinook expected in many strata (Winans et al., 2001; Bellinger et al., 2015; Satterthwaite et al., 2015a) complicates estimation of total catch because small proportions are difficult to estimate precisely unless sample sizes are large (Allen-Moran et al., 2013). It may be appropriate, or at least acceptable, to use a coarser time/area stratification in estimating fishery impacts on CC-Chinook than what has been usual practice for actively managed California Chinook salmon stocks if annual impact rate estimation is the goal, rather than precise estimates of stratum-specific contributions to total impacts. This would allow for smaller total sample sizes, which would reduce total sampling costs. Establishing *a priori* criteria for the precision and level of stratification desired in harvest estimates would aid in determining appropriate sample sizes and assessing their feasibility. Such criteria might be informed by an assessment of the uncertainty in estimates arising from accepted methods used for actively managed salmon stocks, perhaps particularly those that are ESA-listed.

Can a cohort reconstruction be completed for CC-Chinook?

Cohort reconstructions allow for the estimation of abundance, maturation rates, harvest, harvest rates, and many other metrics used to assess stock status and the effect of fisheries on a population (Hilborn and Walters, 1992; O'Farrell et al., 2012). The basic method for cohort reconstruction is the sequential rebuilding of abundance from the end of a cohort's life, when abundance is zero, to an earlier age, usually prior to recruitment to ocean fisheries, by accounting for removals due to fishing, natural mortality, and maturation. Most commonly, the core data used for cohort reconstructions are CWT recoveries (properly expanded for nonexhaustive sampling and, in some cases, the fraction of fish released with marks and tags) from freshwater escapement surveys, river harvest surveys, and ocean harvest surveys. However, CWT data are not strictly necessary so long as the core information requirements of cohort reconstruction are met. Here we evaluate whether the data currently available for CC-Chinook satisfy these requirements.

Freshwater data requirements for cohort reconstruction include age-specific escapement and river harvest data. For CC-Chinook, age-specific escapement data do not exist. As noted in Section 7.1, total escapement cannot be estimated from the current data, and there has been no known effort to estimate age structure. There are no records of freshwater CWT recoveries in the RMPC CWT database. Freshwater harvest of CC-Chinook is prohibited. Thus, because age-specific escapement data do not exist for CC-Chinook, the freshwater data requirements for cohort reconstruction are not met.

The ocean data required for cohort reconstruction is age-specific harvest. Minimal CWT data exist, and the release of marked and tagged CC-Chinook ceased after the 2002 brood year. GSI has the potential to identify CC-Chinook in the ocean harvest, but to date no estimates of total CC-Chinook ocean harvest exist. Furthermore, minimal information exists on the age structure of ocean-harvested fish identified as CC-Chinook via GSI. To meet the ocean data requirements for cohort reconstruction, recreational and commercial fisheries would both need a carefully planned sampling scheme to generate estimates of total harvest for each of the stocks identifiable by GSI. These harvest estimates would also need to be age-specific, likely requiring extensive scale-aging and careful consideration of the resultant uncertainties. These harvest *estimates would need to be combined with estimates of escapement for the same stock units identifiable by GSI; i.e., a cohort reconstruction could not proceed if ocean harvest was identified to the level of the ESU (or "reporting group") while escapement was measured for only a single river.*

Ocean harvest, maturation rates, and other vital rates can be estimated from cohort reconstructions performed on untagged, natural populations (i.e., CC-Chinook) if a suitable CWT indicator stock exists. To perform natural-origin cohort reconstructions in this manner, age-specific river return estimates for the natural population and information from reconstructed cohorts of the CWT indicator stock are needed. Assuming equality in the ocean fishery contact or exploitation rates between the CWT indicator stock and the natural population, natural-origin cohort reconstructions proceed using the natural population's age-specific river return estimates and the ocean fishery contact or exploitation rate estimates borrowed from the indicator stock. For example, cohort reconstructions of natural-origin KRFC are performed using ocean fishery contact rates estimated from reconstructed hatchery-origin KRFC release groups coupled with age-specific natural-origin KRFC river return estimates (Mohr, 2006). For CC-Chinook, within-ESU indicator stocks do not exist because marking and tagging of CC-Chinook no longer occurs. Other neighboring stocks with CWT programs, such as KRFC and Central Valley Chinook stocks, are not likely to be appropriate indicator stocks for CC-Chinook because of differences in marine stock distributions (see Section 7.5 for more information on the ocean distribution of CC-Chinook). Furthermore, age-specific river return data are not available for CC-Chinook.

Therefore, owing to both freshwater and ocean data deficiencies, cohort reconstructions can not be performed for CC-Chinook at the current time, and as a result estimation of abundance, exploitation rates, and maturation rates is hindered. Moreover, without a time series of historical abundance estimates, preseason forecasting of abundance using traditional methods (i.e., sibling regressions) is also not possible at the current time.

Freshwater data requirements for cohort reconstruction include age-specific escapement and river harvest data (to the extent that river harvest occurs). For CC-Chinook, age-specific escapement data currently do not exist, to our knowledge. There remains no record of freshwater CWT recoveries for CC-Chinook in the RMPC CWT database. Freshwater harvest of CC-Chinook remains prohibited. As a result, the freshwater data requirements for cohort reconstruction remain unmet.

The ocean data required for cohort reconstruction are age-specific harvest. Low levels of CWT data exist, and there are no new sources of CWT data since the publication of O'Farrell et al., (2012). While GSI has the potential to identify CC-Chinook in the ocean harvest, there are no new estimates of total CC-Chinook ocean harvest, and CC-Chinook identified by GSI would need supplemental age analysis (e.g., via scale reading) to inform a cohort reconstruction.

The core data needed for cohort reconstructions continue to be unavailable for CC-Chinook. Without a time series of historical abundance estimates, preseason forecasting of ocean abundance using traditional methods (such as sibling regressions) is also not possible at the current time. However, the ongoing development of ecosystem indicators relevant to predicting salmon production and abundance (CCIEA, 2022) has the potential to provide information on the likely abundance of cohorts still in the ocean. Such information might serve to provide abundance forecast skill comparable to accepted methods for other stocks, given the modest performance record of traditional abundance forecast methods (Satterthwaite and Shelton, 2023).

Can an abundance index for CC-Chinook be estimated?

The minimum data requirements for cohort reconstruction are not met by many West Coast salmon stocks. However, in some cases, the data do allow for the estimation of an abundance index and a crude exploitation rate based on the ratio of total catch to the sum of total catch and total escapement (Hankin and Healey, 1986). For example, the Sacramento Index (SI) has been used for assessment of Sacramento River fall Chinook, and the forecast SI is used to define annual exploitation rate targets or limits for that stock (O'Farrell et al., In prep.). The SI is defined as the sum of total escapement, total ocean harvest, and total river harvest.

For CC-Chinook, the lack of total escapement and ocean harvest data currently precludes the estimation of an abundance index for the entire ESU. A more realistic goal might be to estimate an abundance index for an indicator population within the ESU. A leading candidate would be the Russian River population, which appears to have the most complete estimate of annual escapement and can be identified in ocean fisheries with GSI methods. However, ocean catch of Russian River Chinook would need to be estimated using GSI-derived stock proportions collected from all months/areas/fisheries. Because Russian River Chinook make up only a fraction of CC-Chinook abundance, estimating the proportion of Russian River Chinook in the ocean harvest would require even larger sample sizes for acceptable precision than for the aggregate CC-Chinook ESU.

Estimates of catch in all months/areas/fisheries currently do not exist. As a result, the extent and resolution of ocean catch data precludes estimation of an abundance index (analogous to the SI) for any of the populations within the CC-Chinook ESU.

Indices of ocean abundance and exploitation rates similar to those derived from cohort reconstructions have been developed and used in the management of some stocks, including Sacramento River Fall Chinook (O'Farrell et al., 2013), the primary contributor to ocean harvest off California. This stock is assumed to have a relatively low diversity in its adult age structure, and efforts are underway to improve the assessment of this stock by including age data. It is unknown how significantly it would compromise the understanding of fishery impacts on CC-Chinook if age structure was neglected.

Ideally, harvest and escapement would be estimated at the same level of resolution (e.g., for the Russian River component of the ESU, which is the only component that can be separately identified via GSI with a high degree of confidence), but given the low abundance of Russian

River Chinook, this may result in the need for very intensive GSI sampling to yield adequate sample sizes. Given the precedent for using harvest rate indices in place of direct estimates of age-specific impact rates, there may be merit in considering the utility of a harvest rate index for CC-Chinook derived as the ratio between total ocean catch of CC-Chinook (ESU-wide) and summed escapement for a core set of indicator streams (for example, the Upper Eel River, Mad River, and the Russian River). To the extent that the core set of streams made up a constant fraction of ESU-wide escapement, variation in this index would reflect variation in the impacts of fisheries on CC-Chinook, although the index could not be interpreted in an absolute sense (i.e., it would not be constrained to be less than 1.0). As longer time-series of suitable escapement estimates become available, it might be possible to develop defensible assumptions about the correlation structure in escapement to different CC-Chinook watersheds that could inform simulations exploring the likely level of noise in this index attributable to variation in the proportion of total escapement returning to index reaches that are adequately sampled. Estimates of CC-Chinook ocean catch could come from GSI sampling and analysis. However, extensive and regular GSI sampling has not occurred consistently in California and Oregon in recent years, and it is unknown whether such an effort will be pursued in the future. The Pacific Coast Salmon Fishery Management Plan (PFMC, 2022b) expressed an intent to develop the programs necessary to support stock-specific management within five years of an ESU being listed, which for CC-Chinook would have been 2004.

Increased freshwater monitoring offers some promise for a reliable escapement time series for a core set of indicator streams that could contribute to a CC-Chinook index of abundance. However, a time series of CC-Chinook ocean harvest estimates is not currently available. Until more consistent ocean harvest estimates can be generated, the ability to estimate an ocean abundance or harvest rate index will remain elusive.

Can abundance of CC-Chinook be inferred from other stocks?

Equivalent (or approximately so) estimates of catch-per-unit-effort (CPUE) for two or more stocks in a month/fishery stratum would suggest similar local abundance if the following conditions are met. First, the stocks would need to have equal catchability. Second, the fishing fleet should not differentially capture one stock over another (i.e., the fleet would randomly sample the aggregate local abundance). If these conditions were met, similarity in local CPUE would indicate similarity in local abundance. However, this would not necessarily imply similarity in total ocean abundance between these stocks. More information regarding the spatial distributions of the stocks at sea would be needed to make such an inference. For example, if the distributions of the stocks were identical, then CPUE similarities may imply similar abundance. If the distributions were not identical, differences in fishing effort in space and time could lead to misleading inferences with regard to abundance. To illustrate this point, consider the following hypothetical scenario. A similar CPUE is estimated for Klamath Chinook and CC-Chinook in FB for August, while KC is closed to fishing and sampling. If the above assumptions hold and distributions were identical, this result would correctly imply a similar ocean abundance for these stocks. If, however, the underlying spatial distribution of Klamath Chinook results in the bulk of their abundance being located in KC and the bulk of CC-Chinook *abundance located in the FB management area, a similar CPUE between the stocks for FB in August would indicate that the Klamath stock is much more abundant. Inferring similar abundance from similar local CPUE would therefore draw an incorrect conclusion.*

Differences in estimated Klamath Chinook and CC-Chinook ocean spatial distributions have been identified from an analysis of contacts per unit effort based on GSI data from 2010 and 2011. Satterthwaite et al., (In prep.) found that contacts per unit effort were similarly distributed for Klamath Chinook and CC-Chinook early in the year (analysis possible only in 2010), but late in the year (July or August) contacts per unit effort were relatively higher for CC-Chinook in the FB area and for Klamath Chinook in the KC area (this pattern held qualitatively in both 2010 and 2011). The comparison was confounded by the closure of the area between Humboldt South Jetty and Horse Mountain, an area that has been closed to commercial salmon fishing since the early 1990s, largely for the purpose of protecting CC-Chinook populations. This result must be interpreted with caution since it is limited to two years' data, and likely more complicated patterns would emerge in time. We might expect a high concentration of CC-Chinook in the closed area as spawners return to the Eel and Mattole rivers with mouths in that area but cannot test this hypothesis directly with the data at hand. The GSI-based estimates of CC-Chinook spatial distributions made in Satterthwaite et al., (In prep.) are not inconsistent with the CWT-based inferences made for CC-Chinook in NMFS (2000), despite the differences in data and methods used.

In sum, there is potential for evaluating relative local ocean abundance of KRFC and CC Chinook with CPUE data, and such data currently exist from the GSI sampling program in 2010 and 2011. However, CPUE data must be interpreted cautiously when inferring relative, range-wide abundance because of uncertainty in differences in catchability and spatial distributions. If such problems could be resolved, and more data become available, relative CPUE measures could be used to make inferences about stock abundance. However, it seems unlikely that this approach could be useful for fishery management. The CPUE data necessary to infer CC-Chinook abundance would come from the fishery, and the bulk of the fishery occurs after the preseason fishery planning process. Relative CPUE measures from fisheries conducted the previous fall (September– November) could be investigated, though interpretation of these data is likely problematic because of the potential for run-timing differences between KRFC and CC-Chinook. Peak river mouth return of KRFC occurs around September 1 (O'Farrell et al., 2010), while CC-Chinook may only be able to enter certain natal rivers after the first large winter storms, which typically arrive in November (Bjorkstedt et al., 2005).

If abundance of CC-Chinook and other stocks (i.e, KRFC, [Sacramento River fall Chinook] *SRFC) are highly correlated, then preseason forecasts of the more data-rich stocks could potentially be used to infer relative abundance of CC-Chinook. The only CC-Chinook data series judged to be of adequate quality to represent total escapement is from the Russian River. Examination of the pairwise relationships between Russian River video counts and river mouth return estimates of adult KRFC, age-4 KRFC, and SRFC indicate low correlation (Figure 3). For these comparisons, river mouth returns for KRFC and SRFC were compared to the Russian River escapement estimates because the Russian River population is not subject to river fisheries and, assuming little river natural mortality, the river mouth return and escapement* *values should be comparable. Ignoring the correlation between adult and age-4 KRFC, the highest correlation exists between the Russian River and SRFC, although this correlation coefficient was not statistically significant (p = 0.098). The Russian River lies at the southern end of the CC-Chinook ESU and is the most proximate CC-Chinook population to Central Valley Chinook stocks; a lower correlation with SRFC might be expected for other CC-Chinook populations. The relatively low correlation between KRFC or SRFC and the Russian River population suggests that using KRFC or SRFC abundance forecasts to infer abundance of CC-Chinook would be problematic.*

Similarities and differences in estimated Klamath Chinook and CC-Chinook ocean spatial distributions have been identified from an analysis of contacts per unit effort based on GSI data. Note that the genetic baselines used for these analyses (Seeb et al., 2007; Clemento et al., 2014) do not distinguish the fall versus spring runs of Klamath Chinook; however, abundances of Klamath Spring Chinook are generally low relative to Klamath Fall Chinook (PFMC SRKWWG, 2020, Appendix A) and differences in the spatial distributions inferred from hatchery-origin coded-wire tagged Trinity River Hatchery Spring Chinook compared to tagged Klamath Fall Chinook were small (Satterthwaite and O'Farrell, 2018) relative to the precision possible given typical GSI sample sizes. Comparisons based on commercial fishery recoveries are also confounded by the closure of the area between Humboldt South Jetty and Horse Mountain, an area that has been closed to commercial salmon fishing since the early 1990s, largely for the purpose of protecting CC-Chinook populations (the southern boundary of the closure area was moved 5 nautical miles north of Horse Mountain to Latitude 40°10' N, beginning in 2021). Despite these caveats, some broad inferences seem generally supported, as described below.

Satterthwaite et al., (2014) performed a detailed comparison of spatial patterns in CPUE for CC-Chinook versus Klamath Chinook using GSI data collected by commercial fishermen participating in the West Coast Salmon GSI Cooperative in 2010 and 2011 during normal operations as well as limited non-retention sampling during times fisheries were otherwise closed, when fishermen were instructed to mirror their typical practices as closely as possible. Spatial patterns in CPUE were generally similar early in the year, but by August and September there tended to be divergence in where peak CPUE occurred, with Klamath Chinook showing a sharp peak in the California Klamath Management Zone (KMZ, Eureka and Crescent City), while CPUE of CC-Chinook was more spread out and sometimes peaked in the Fort Bragg management area. In June (2010 data only), 95% Bayesian credible intervals on the ratio between CPUE in Eureka and CPUE in Fort Bragg for the two stocks substantially overlapped. In July, CPUE in Fort Bragg was significantly higher than CPUE in Eureka for CC-Chinook but not for Klamath Chinook in 2010, and was significantly higher in Fort Bragg than the California KMZ for both stocks but by a significantly smaller amount for Klamath Chinook in 2011. In August of both 2010 and 2011, the ratio between CPUE in Eureka or the California KMZ versus Fort Bragg was significantly higher for Klamath Chinook than for CC-Chinook, as was the case in September 2010 (there were no data from September 2011).

Bellinger et al., (2015) described patterns in stock proportions and CPUE inferred across a broad range of stocks based on the same 2010 GSI data informing Satterthwaite et al., (2014), but looking at all stocks identifiable from the genetic baseline used. This paper provides context on the proportional contribution of CC-Chinook and Klamath Chinook to commercial fishery catches in different times and areas, but does not discuss CC-Chinook versus Klamath Chinook specifically.

Satterthwaite et al., (2015) used GSI data collected from dockside sampling of California recreational fisheries by CDFW during 1998–2002 to document patterns in stock composition and stock-specific CPUE for stocks encountered in California ocean fisheries that could be identified using the Clemento et al., (2014) baseline. The paper did not focus on direct comparisons of CC-Chinook versus Klamath Chinook, but did note that CPUE of CC-Chinook was generally highest in Fort Bragg during August or September of most years, while CPUE of Klamath Chinook was always highest in Eureka or Crescent City during these months, similar to the patterns identified in the 2010 and 2011 troll data by Satterthwaite et al., (2014). Figure 2 shows spatial patterns in CPUE calculated from the same GSI dataset used in this paper, specifically for CC-Chinook and Klamath Chinook. Stock-specific CPUE estimates were obtained by multiplying year-, month-, and area-specific estimates of stock composition based on GSI sampling by estimates of total recreational ocean Chinook catch (from records maintained by CDFW and PFMC) divided by total recreational fishing effort (angler-days). Stock proportions (mean and 95% credible intervals) were estimated using the statistical model implemented in the R package zoid (Jensen et al., 2022) to account for genetic assignment and sampling uncertainty, but this analysis assumes that total catch and effort are known without error.

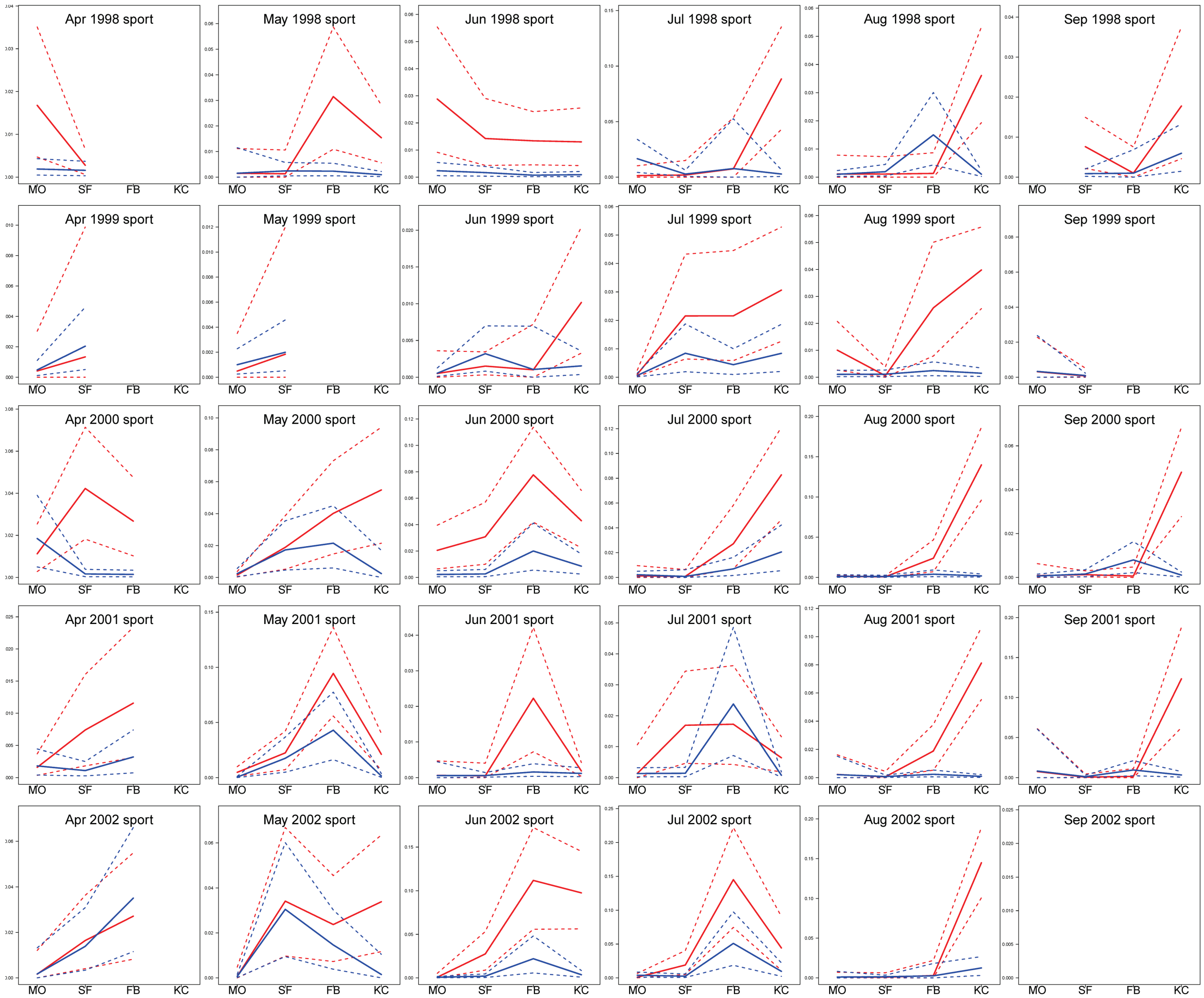

Figure 2. Spatio-temporal variation in stock-specific CPUE of CC-Chinook (blue) and Klamath Chinook (red) for recreational fisheries off California in 1998–2002. Solid lines reflect point estimates of stock composition from GSI samples applied to total Chinook catch and effort for a specific month-management area (areas are as defined in PFMC [2021, Figure 6-1] except that they are restricted to California waters), while dashed lines reflect use of the upper or lower bound of the 95% credible interval on stock proportions.

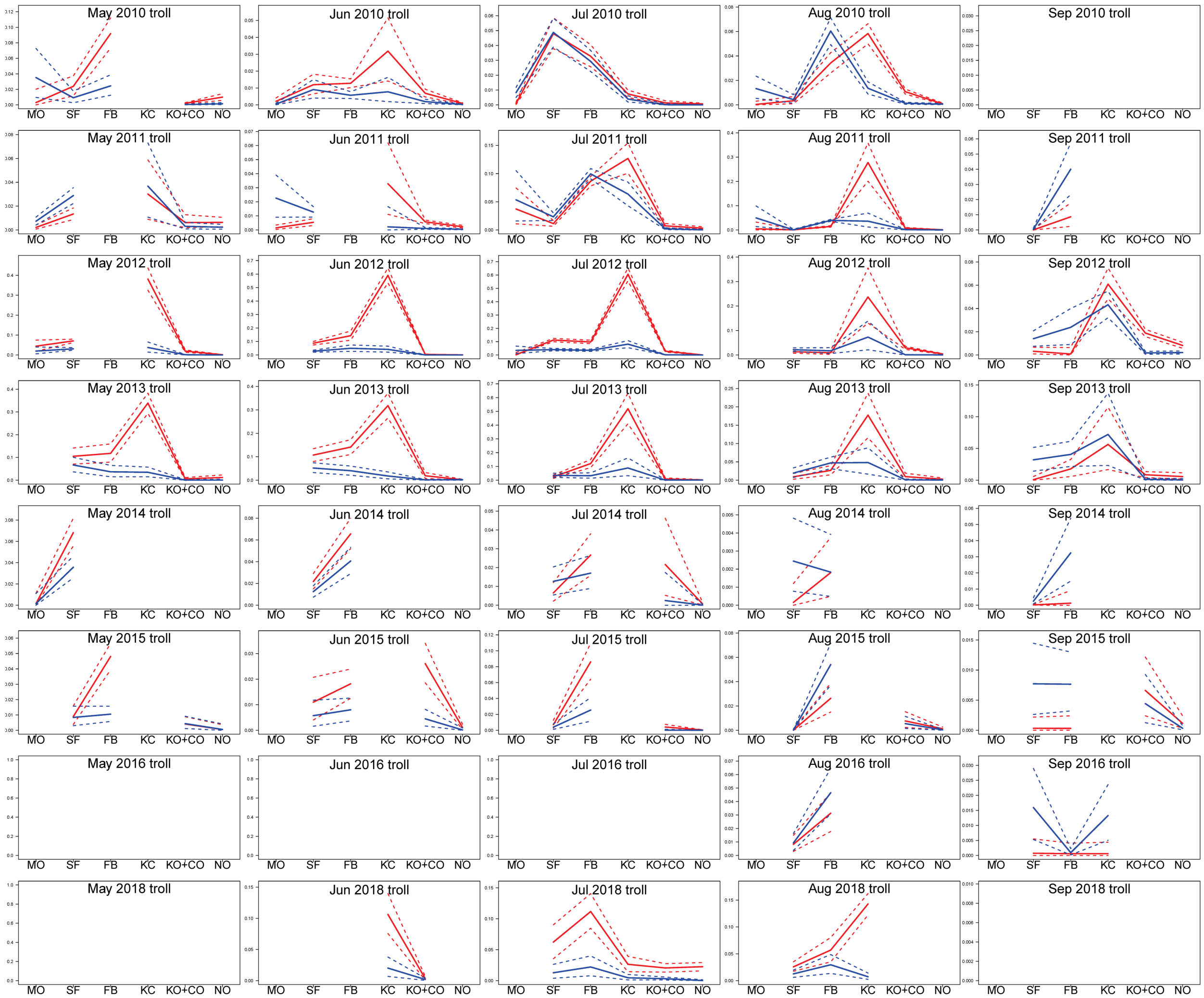

Figure 3. Spatio-temporal variation in stock-specific CPUE of CC-Chinook (blue) and Klamath Chinook (red) for commercial troll fisheries off California and Oregon south of Cape Falcon in 2010–2018. Solid lines reflect point estimates of stock composition from GSI samples applied to total Chinook catch and effort for a specific month-management area (areas are as defined in PFMC [2021, Figure 6-1] except that the KMZ is split at the state boundary and the Oregon portion of the KMZ is merged with CO for sample size reasons), while dashed lines reflect use of the upper or lower bound of the 95% credible interval on stock proportions.

Additional GSI data have been collected from commercial troll fisheries by the West Coast Salmon GSI Collaboration since 2011, although spatial and temporal (i.e., monthly) coverage has varied across years. Figure 3 summarizes spatio-temporal patterns in stock-specific CPUE based on zoid-estimated stock proportions from these data applied to PFMC records of total Chinook catch and effort in different management areas. Stock proportions were only estimated for year-month-area strata where at least 10 fish were genotyped. Only the core fishery months of May – September are presented due to limited data from other months, and months where only one area was sampled or years when only Oregon waters were sampled are not shown because they provide limited information on how CPUE of the focal stocks varies through space. Uncertainty in total Chinook catch and effort is not accounted for. No results are shown for September 2010 because only non-retention sampling was performed, but see Satterthwaite et al., (2014) for patterns then, with Klamath Chinook CPUE in Eureka much higher than in Fort Bragg while CC-Chinook CPUE was higher in Fort Bragg than in Eureka.

As shown in both Figure 2 and 3, for every year with data, the highest CPUE (in both recreational and commercial fisheries) in August and September for Klamath Chinook was always in the KC management area (i.e., the California portion of the Klamath Management Zone [KMZ], where the main ports are Eureka and Crescent City). In contrast, CPUE for CC-Chinook in August and September in FB (the Fort Bragg management area, which also contains Shelter Cove) was often (but not always) comparable to or even higher than CPUE in KC. Patterns earlier in the year defy easy summarization, and some year-specific differences were apparent, but no patterns as consistent as those for August and September were identified.

These patterns in CPUE may have relevance to evaluation of the continued applicability of the existing CC-Chinook ocean fishery consultation standard, or to evaluating the level of concern for CC-Chinook to associate with recent patterns of the Klamath Ocean Harvest Model (KOHM) under-predicting age-4 ocean harvest rates for KRFC. If the proportion of the total modeled age-4 ocean harvest of KRFC that takes place in Fort Bragg versus the KMZ in August and September has stayed fairly stable, the existing consultation standard has likely provided a fairly consistent level of protection for CC-Chinook. If the proportion of impacts attributable to KMZ fishing in August and September has increased, this may mean increasing protection for CC-Chinook, whereas if the proportion of impacts attributable to KMZ fishing in August and September has decreased it may mean diminished protection for CC-Chinook. Recently, bias has been identified in the preseason application of the KOHM, with a tendency to under-predict ocean harvest rates on KRFC. Depending on where the under-prediction is most severe, this could cause relatively less concern for CC-Chinook protection if the under-prediction is worst in the KMZ during August (noting that the KOHM is not used to plan September fisheries); or the most concern if under-prediction is severe for Fort Bragg during this same time period.

While there is potential for evaluating relative local ocean abundance of KRFC and CC-Chinook with CPUE data informed by GSI, existing GSI data typically are not sufficiently complete in space and time. CPUE data must be interpreted cautiously when inferring relative, range-wide abundance because of uncertainty in differences in catchability and spatial distributions. It might be possible to resolve these problems by identifying core area-time combinations for comparisons, and consistently sampling them; or through fitting a model with both year and area effects like Satterthwaite et al., (2013), interpreting the year effect as an index of annual abundance (based on a representative month, or developing a method to share information across months). If these approaches were deemed acceptable, and more data become available, relative CPUE measures could be used to make inferences about stock abundance. However, there would be challenges in using this sort of approach for preseason fishery planning. The CPUE data necessary to infer CC-Chinook abundance would come from the fishery, and the bulk of the fishery occurs after the preseason fishery planning process. Relative CPUE measures from fisheries conducted the previous fall (September–November) could be investigated, though interpretation of these data is likely problematic because of the potential for run-timing differences between KRFC and CC-Chinook. Peak river mouth return of KRFC occurs around September 1 (O’Farrell et al., 2010), while CC-Chinook may only be able to enter certain natal rivers after the first large winter storms, which typically arrive in November (Bjorkstedt et al., 2005), especially in more southerly populations. However, if it is possible to forecast the CPUE index (perhaps using ecosystem indicators, as described above), this could provide information in time to be used in the preseason planning process. In-season management, as is done for some northern salmon stocks, might also be an option, although applications to date have largely been limited to terminal fisheries (Michielsen and Cave, 2019; Staton and Catalano, 2019).

In O’Farrell et al., (2012), it was noted that if abundance of CC-Chinook and other stocks are highly correlated, then preseason forecasts of the more data-rich stocks could potentially be used to infer relative abundance of CC-Chinook. In 2012, the only CC-Chinook data series judged to be of adequate quality to represent total escapement was from the Russian River. Pairwise relationships between Russian River video counts (updated through the 2020–2021 run) and river mouth return estimates of adult KRFC, age-4 KRFC, and SRFC are displayed in Figure 4. Correlations between Russian River and KRFC age-4, and Russian River and SRFC, are similar to those reported in O’Farrell et al., (2012) (0.45 and 0.50, respectively). However, the updated return estimates resulted in a substantial increase to the correlation between the Russian River and KRFC (0.079 to 0.53). These correlations are modest, but comparable to or higher than the correlations in escapement between SRFC and the other stocks in the Central Valley Fall Chinook stock complex for which SRFC is currently the indicator (0.38-0.41 for 1970-2021)[1].

[1] https://www.pcouncil.org/documents/2022/10/d-2-attachment-1-methodology-review-materials-electronic-only.pdf/#page=50

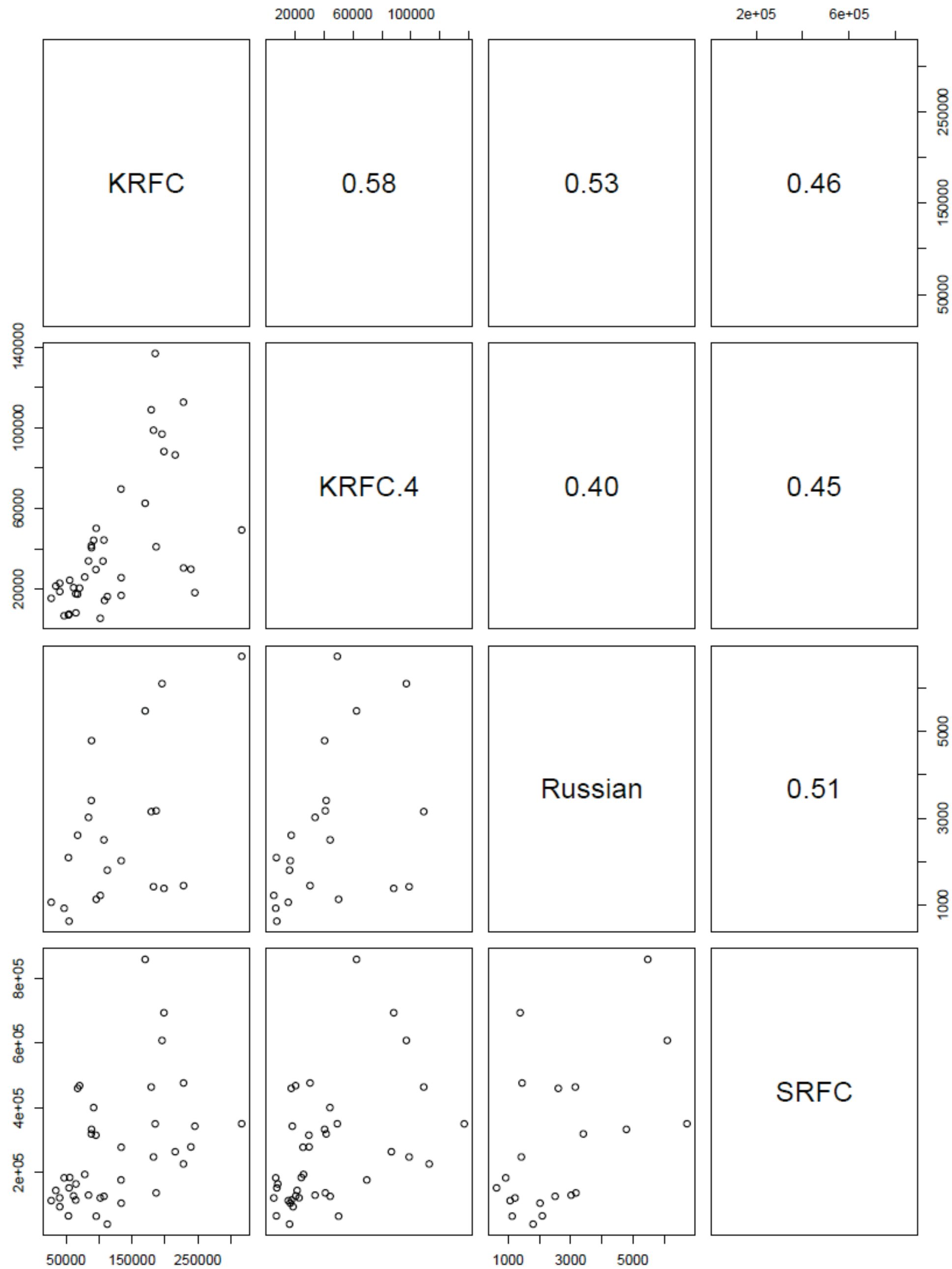

Figure 4. Pairwise comparisons between Russian River escapement estimates and river return estimates for Klamath River fall Chinook (KRFC) and Sacramento River fall Chinook (SRFC). Correlation coefficients appear above the diagonal. KRFC denotes adult (age 3–5) river return and KRFC.4 denotes age-4 river return. Correlation coefficients for comparisons between the Russian River and the three other stock components were estimated for run years 2000–2001 through 2020–2021.

What is an appropriate exploitation rate for CC-Chinook?

While we have addressed questions about the estimation of exploitation rates, of perhaps equal or greater importance is the question of what exploitation rate is appropriate for the CC-Chinook ESU. Estimation of a stock-recruitment relationship can allow for estimation of stock productivity, which defines the exploitation rates that maximize yield, allow for population persistence, or promote recovery.

Sufficient data do not exist for estimating a stock-recruitment relationship at the ESU- or population-level, again due to the lack of sufficient escapement and ocean harvest information. Future data generated from LCMs may assist in the estimation of appropriate exploitation rates, although these data are not currently available and many years of data will be necessary before stock recruitment relationships are estimable. Hence, at present, the productivity of the CC-Chinook ESU is unknown and, as a consequence, appropriate exploitation rates cannot be determined for the ESU or its constituent populations.

As was the case in 2012, sufficient data do not exist for estimating a stock-recruitment relationship at the ESU- or population-level, owing to deficiencies in escapement and ocean harvest information. The productivity of the CC-Chinook ESU remains unknown and, therefore, appropriate exploitation rates cannot be determined for the ESU or its constituent populations at the current time.

Although this informs only one of many aspects relevant to understanding exposure to fisheries and ability to sustain different levels of exploitation rates, there are some indications that maturing CC-Chinook return to freshwater later in the year than Klamath fall Chinook (Bjorkstedt et al., 2005). Consistent with this difference, Satterthwaite et al., (2014) compared size-at-age for ocean-harvested fish from the two stocks based on scale-aged fish within the 2010 GSI dataset, and found that they were similar early in the year, but that CC-Chinook tended to be larger at a given age later in the year. They hypothesized that larger fish within a cohort were more likely to mature, and so this size difference later in the year might reflect maturing CC-Chinook leaving the ocean later in the year than do Klamath Chinook, and/or a smaller proportion of CC-Chinook maturing at younger age classes. This could affect the relative exposure of the two stocks to fisheries occurring late in the year, as well as the cumulative amount of time fish from the two stocks are exposed to the risk of fishing mortality. Scale aging has generally not occurred for GSI samples collected after 2010, so it has not been possible to assess the generality of the pattern observed in 2010.

In addition to the potential for greater cumulative exposure to fishing, the difference in return timing highlights an important challenge in managing fishery impacts on CC-Chinook via a proxy stock with potentially different life history. In assessing and managing fishery impacts on Klamath Fall Chinook, ocean fishery impacts occurring after August 31 are counted against the subsequent management year and assumed to reduce potential escapement in future years rather than the current year. For CC-Chinook, it may be important to consider impacts later into the calendar year for any given management year.

Discussion

Past explorations into the feasibility of developing alternative fishery management strategies for CC-Chinook (O'Farrell et al., 2012, 2015) have identified potential alternatives for improving data sources that could allow for future changes to CC-Chinook fishery management. These included (1) full implementation of the CMP with stable funding, (2) establishment of indicator populations for the CC-Chinook ESU, and (3) development of a pilot program aimed at assessing the feasibility of a marking and tagging program for estimation of ocean harvest.

With regard to CMP implementation, substantial progress has been made for several CC-Chinook salmon populations, and the escapement information has been improved since the publication of O'Farrell et al., (2012) across much of the ESU. However, regular funding has not been available for some programs, which has resulted in gaps for some time series. Ocean salmon fishery planning along the U.S. west coast is an annual process that for many salmon stocks is informed by past estimates of abundance and exploitation rates. Many (but not all) of the methods employed rely on recent year estimates of spawner escapement, ocean abundances, and/or exploitation rates to inform acceptable levels of planned fisheries. Gaps or lags in this information limit the available tools for stock-specific fishery management. For example, concern over the dependability and timely availability of data needed for forecasting abundance of Southern Oregon Northern California (SONCC) coho stocks resulted in limited consideration of abundance-based harvest control rules for that ESU[2].

The recommendation to establish an indicator population (or populations) in previous reports focused on obtaining sufficient data to perform cohort reconstructions. However, the data requirements for cohort reconstruction (age-specific harvest and escapement) are not currently met for any CC-Chinook population, with few prospects for obtaining such data in the foreseeable future. Age-specific escapement for some populations might be estimable if collection of age data was prioritized. However, estimation of age-specific ocean harvest for these corresponding populations is likely to be more difficult. The recommendation to establish a pilot program to mark, tag, and release 200,000 CC-Chinook salmon has the potential to provide sufficient data for cohort reconstructions, but such a program has not been funded or initiated. At the current time, it appears unlikely that a substantial marking and tagging program for natural-origin CC-Chinook, or resumption of hatchery production (and associated marking and tagging of releases), will occur.

It should be noted that some actively managed salmon stocks do not meet the data requirements for cohort reconstruction. While some of these stocks may have a less diverse age structure than the unknown age structure of CC-Chinook, abundance and harvest rate indices that neglect age structure (and have other limitations in how stock-specific catch is estimated) are used annually to support fishery management decisions (e.g., O'Farrell et al., 2013). However, such an approach would require more escapement and ocean harvest data than are available for CC-Chinook at the current time.

GSI data could be used to inform a general indicator of abundance and an index of harvest

[2] https://www.pcouncil.org/documents/2021/10/f-3-a-soncc-workgroup-report-1-electronic-only-fishery-harvest-control-rule-risk-assessment.pdf/

impacts. CPUE estimated from GSI sampling in a consistent core time/area might provide an index of relative abundance, or such an index might be derived as the year effect of a model fitting time and area effects to CPUE data (Satterthwaite et al., 2013). Once a sufficient time series of such abundance indices had been obtained, the ability to forecast the abundance index could be explored. Given an ability to forecast or use recent means of the abundance index, adjustments in expected fishery impacts might be possible either through the current age-4 Klamath Fall Chinook ocean harvest rate proxy, or by regulations informed by typical patterns in time and space of CPUE inferred from GSI. Relative harvest impacts from one year to the next might also be indexed by comparing ESU-wide catch inferred from GSI to the escapement of a consistently sampled core set of indicator populations, although it would be important to communicate to all potential users of such an index that it should not be interpreted on an absolute scale.

In conclusion, we reiterate the recommendations outlined in O'Farrell et al., (2015), with some additional thoughts. Full implementation of the CMP in core Chinook spawning areas would be highly beneficial for status reviews and could be one building block for a new CC-Chinook fishery management strategy. Establishment of ocean fishery sampling programs for collection of GSI data would provide much-needed CC-Chinook catch data which could also contribute to a CC-Chinook-specific fishery management strategy. Furthermore, consistent GSI sampling could provide information for other stocks in the fishery that are not marked and tagged (e.g., natural-origin Central Valley Spring Chinook) or are not retained in fisheries but still incur fishing-related mortality (e.g., SONCC coho in California). Implementation of both the CMP and GSI sampling programs have suffered from a lack of stable funding, which greatly complicates assessment of stock status and implementation of many fishery management strategies. The recommendation to develop a pilot program aimed at assessing the feasibility of a marking and tagging program for estimation of ocean harvest still has merit. However, given the numerous difficulties anticipated in establishing such a program, and the potential for a program of that size to provide inadequate tag recoveries, it is prudent to look more closely at obtaining estimates of ocean harvest though other means.

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