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San Francisco Bay Conservation and Development Commission
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February 6, 2026

California State Lands Commission
100 Howe Avenue, Suite 100-South
Sacramento, CA 95825

ATTN: Lieutenant Governor Eleni Kounalakis, Chair

SUBJECT: San Francisco Bay and Delta Sand Mining Project Final Supplemental Environmental Impact Report; State Clearing House No. 2007072036

Dear Ms. Kounalakis:

By letter dated September 29, 2025, San Francisco Bay Conservation and Development Commission (BCDC) staff submitted extensive comments on the San Francisco Bay and Delta Sand Mining Draft Supplemental Environmental Impact Report (DSEIR), which evaluates proposed 17.5 million cubic yards of sand mining in Central San Francisco Bay and Suisun Bay over the next ten years in five state lands lease areas and one private lease area in Central San Francisco Bay and Suisun Bay per the California Environmental Quality Act (CEQA). BCDC's mission is to protect and enhance San Francisco Bay and advance the Bay's responsible, productive, and equitable use—for today and for generations to come—as we face a changing climate and rising sea levels. As the California State Lands Commission (SLC) is aware, the proposed sand mining activities would occur within the BCDC's Bay jurisdiction, apart from a portion of the eastern end of the Suisun Lease area, and the proposed projects will require BCDC permits for the mining activities within our jurisdiction.

SLC's Final Supplemental Environmental Impact Report (FSEIR) is an important component of CEQA environmental review that will inform BCDC's permitting process, as a responsible agency, and it is therefore critical that the FSEIR provides a full and reasoned examination of the impacts of the proposed activities under the current and future conditions. BCDC staff is disappointed that many of their comments provided in our September 29th letter were not adequately addressed between the DSEIR and the FSEIR. The most concerning remaining issues are provided for your consideration below.

1. **Current Mining Activity and Proposed Project**. The State Lands Commission previously reviewed and authorized 22 million cubic yards of sand mining over ten years in its 2012 CEQA document and lease. However, over the past ten years, only approximately 8 million cubic yards of sand was actually mined by both companies combined.

   The DSEIR and the FSEIR describe the Proposed Project of 17 million cubic yards of sand mining over ten years as a reduction of mining activity. However, this actually represents a 60% increase in mining activity from existing conditions, not a reduction in mining.

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2. **Changes in the Environment**. In the DSEIR and the FSEIR, the SLC asserts that the environmental conditions and mining conditions have not changed sufficiently since 2012 to warrant a full review of the project under CEQA. The purpose of a supplemental EIR is to document the minor additions or changes necessary to make an original EIR adequate for a revised project, and it only needs to contain the information necessary to meet that purpose.

   Since 2012 there have been notable and well documented changes to the environment, including changes in both ocean and Bay conditions due to climate change; changes in wildlife use of the Bay (whales, seabirds, and fish); reductions in populations of threatened and endangered species; new and proposed threatened and endangered species listings by state and federal agencies; new scientific information that documents the lack of sand being transported into the Bay; confirmation that sand present is relic and a non-renewable resource; continued erosion in the Bay, specifically mined areas eroding at a faster rate than other areas; lowering of the Bay floor, and erosion along the outer coast. The SEIR does not fully evaluate the continuation of mining at the proposed higher rate compared to the rate of mining that has actually occurred in the last 20 years and the significant changes in the environment.

3. **Reduced Mining Alternative**. The SLC staff included consideration of a "reduced mining alternative," (mining approximately 700,000 cy per year) which was determined to be the environmentally superior alternative. Yet, the SLC staff found that this level of mining would not reduce impacts but rather have the same impact of mining 1.75 million cubic yards per year. This statement is illogical and ignores the magnitude and intensity of mining impacts on the marine environment, particularly regarding entrainment, disturbance and loss of habitat. The SLC analysis regarding why this alternative was selected is lacking documentation and scientific credibility.

4. **Sand Transport, Provenance, and Budget Studies.** Over the past five years, significant efforts were undertaken to better understand the sand availability, transport mechanisms, origins, and impacts from mining. The findings report, "*San Francisco Bay Sand Budget, Transport, Provenance, and Bathymetric Change Studies and Potential Physical Effects of Sand Mining Activities*." (Stantec 2024) integrates the results of multiple sand studies and develops key findings based on the combined knowledge and expertise of five of the foremost experts in the sediment and hydrology fields in the region. This research and synthesis further informs the sand system and conditions within the Bay's sand shoals and bedforms. It states:

   "1. The volume of mined sand is significant relative to the Bay sand budget (it represents the largest outflow of sand from the Bay), including the net sand discharge to the ocean. Additionally, sand is mined faster than it is replenished; therefore, sand is a non-renewable resource over the long term."

   2. Suisun bay sand is not replenished; thus, it is a finite resource, and the bed is being lowered. The bathymetric, modeling, and budget studies all support this. Sand mining effects in lease areas appear to be highly localized, with effects diminishing

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with distance from the event location. The effect is pronounced in areas of negligible sand transport such as Suisun Bay, where the depressions caused by mining persist in the bed over time."

3. Central Bay sand is relic (i.e., deposited between 20,000 and 6,000 years ago as sea levels rose and the river discharge point migrated through the Bay to its present location in the Delta) and is part of a large bay-ocean reservoir of sand. Sands derived from the watersheds of the Sacramento and San Joaquin rivers are no longer a significant source to the Bay and ocean, and large volumes of sand do not move through the system during times of high flows (e.g., wet winters), as was previously assumed. Effects of mining to beaches and ecologically important shoals remain unquantified."

4. San Francisco Bay and the Pacific Ocean share a common pool of sand which sand mining reduces. In each tidal cycle, a huge amount of sand is transported between the Bay and the ocean effectively linking the two sand deposits into a shared pool. The size of this shared pool of sand, and thus the significance of the reduction due to mining, is unknown."

Yet the FSEIR and the response to comments dismisses this critical information and does not adequately incorporate the new science into its evaluation of mining effects.

5. **Biological Impacts**. The definition of take of a federally listed species is "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct." Incidental take is unintentional but still constitutes take for both federal and state listed species. In the evaluation of impacts to threatened and endangered species in the FSEIR, it is concerning that Impact BIO-8, Delta and longfin smelt entrainment impacts, was reduced from a Class I impact to a Class II impact simply due to the existing use of fish screens on the water intake pipes of the mining equipment. While the fish screens reduce entrainment of juvenile and adult fish, eggs and larval fish are likely entrained when mining in spawning areas. BCDC had also recommended that Impact BIO-9, impacts to salmon, green and white sturgeon, be revised to a Class I impact, noting that the only difference between Impacts BIO-8 and BIO-9 that was apparent is use of the term would or could take listed species, and that these fish can also be entrained by the mining equipment. Rather than address BCDC's concerns and revise the impact classification, the SLC chose to downgrade the impact valuation of the most vulnerable species to the mining activity.

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In Impact BIO-3, Potential sand mining impacts on benthic habitat, infauna, epifauna, and foraging habitat, the DSEIR characterizes the impacts from mining as temporary, short-term changes in habitat composition and associated wildlife. Because mining occurs regularly and repeatedly in the same location, the bedforms that constitute sand shoal habitat and the benthic communities that live in and on the sand cannot and do not recover. In other words, the impacts are not temporary or short-term.

6. **Mineral Resources.** The SLC included a modeling analysis of the additional ten years of mining proposed at the 1.7 million cubic yard per year rate. It included analysis of the sand supply and proposed mining for each lease area. This analysis found that some lease areas do not have sufficient sand available above 90 feet to support the volume of mining proposed. Despite this analysis, the SLC finds that mining at this increased level would not have significant impacts on mineral resources. This analysis parallels analysis completed by Deltares, which found that replenishment of sand in some lease areas was very low to nonexistent. Only in one evaluated area did the replenishment substantially exceed the extraction rate. The SLC dismisses its own finding, stating only that there is uncertainty in the modeling, and determines that mining at a significantly increased rate from current extraction levels would not have significant impacts.

Thank you for reviewing and considering these supplemental BCDC comments prior to SLC's certification of the FSEIR. If you have any questions regarding our comments or information presented herein, please do not hesitate to contact our staff, Brenda Goeden at 415-352-3623 or via email at brenda.goeden@bcdc.ca.gov.

Sincerely,

DocuSigned by:
Larry Goldzband
FD166E908010417...

LAWRENCE J. GOLDZBAND
Executive Director
San Francisco Bay Conservation and
Development Commission

LG/BG/sr

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California State Lands Commission
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February 3, 2025

Dear Commissioners,

I would like to comment on the Final Supplemental Environmental Impact Report (SEIR) for the San Francisco Bay and Delta Sand Mining Project. Farallon Institute scientists have been monitoring upper trophic predators and their prey in the Central San Francisco Bay since 1996 (with Point Reyes Bird Observatory prior to 2011).

There have been many substantial changes in circumstances since the 2012 EIR, which have not been adequately characterized by the Final SEIR, despite numerous comments submitted on the Draft SEIR. Unfortunately, since many comments have been inadequately addressed in the Final SEIR, the ecology of relevant organisms in the Central San Francisco Bay and effects of the Revised Project on this ecology have not been sufficiently considered. Given this shortcoming, Farallon Institute can only support the No Project Alternative.

We provide detailed rebuttal to the Master Response in the Final SEIR for Marine Birds, below. As time and funding did not allow detailed rebuttal to all the issues still remaining in the Final SEIR, we have included again a list of issues previously identified, many of which were not remedied in the Final SEIR.

In conclusion, there are still many parts of the Final SEIR which are erroneous or inadequate. Limited and flawed studies on lower-taxa biology (benthic infauna), and lack of studies on any higher taxonomic level makes it impossible to adequately consider biological or ecosystem effects of sand mining activities. Moreover, this SEIR makes no attempt to analyze other sources of existing available evidence of the last 10-12 years to assess potential impacts to biological resources, especially seabirds and other marine wildlife.

Thank you for the opportunity to comment.

Sincerely,

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Master Response J, Marine Birds

p. II-31 "The commenters shared how global climate change has resulted in the decline of Pacific herring (*Clupea pallasii)* which has impacted sea bird colonies on the Farallon Islands, causing them to migrate and establish colonies on Alcatraz Island."

This is erroneous. Pacific herring populations have declined, and insufficient attention has been given to the potential impacts of the Revised Project on herring and their eelgrass breeding habitat in adjacent Richardson Bay. But seabird colonies on the Farallon Islands and Alcatraz are related to availability of Northern anchovy (*Engraulis mordax*) and other demersal fishes[1], not Pacific herring.

This lack of attention to detail and lack of understanding of regional marine ecology pervades the SEIR.

p. II-31 "…When a proposed project risks exacerbating those environmental hazards or conditions that already exist, an agency must evaluate the potential impact of such hazards on future residents or users. In those specific instances, it is the *project's* impact on the environment—and not the *environment's* impact on the project—that compels an evaluation of how those future residents or users could be affected by exacerbated conditions."

"If global events result in the change of a species' occurrence within the project study area, then discussion of its occurrence is an important aspect of the environmental setting, but unless the project causes or exacerbates those existing environmental conditions, then those conditions are outside the scope of the EIR. Global environmental concerns such as climate change are considered in a CEQA assessment when a proposed project could contribute to a cumulatively significant impact, or when existing environmental conditions (potentially influenced by global events) could be worsened by the project (i.e., generation of additional greenhouse gas emissions)."

"…There is no evidence that the environmental changes that could occur if the Revised Project is approved would cause a new significant impact or a more severe significant impact than was disclosed in the 2012 EIR due to loss of foraging habitat or food relied upon by San Francisco Bay bird colonies."

This conclusion is erroneous. Environmental hazards and conditions do already exist, which the proposed project risks exacerbating. Large-scale climate events have resulted in the change of multiple species' occurrence within the project study area in recent years as well a reduced habitat outside of the project study area, rendering habitat within the project study area proportionally much more important.

[1] Sydeman et al. 2001. https://www.sciencedirect.com/science/article/abs/pii/S0079661101000283\
Saenz et al. 2006. http://www.marineornithology.org/article?rn=681
Ainley et al. 2018. https://doi.org/10.1016/j.biocon.2017.11.021
Robinson et al. 2019. http://www.marineornithology.org/article?rn=681

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The Revised Project impacts the waters, benthos, and organisms in the Central Bay. Specifically for marine birds, the Revised Project will likely both directly and indirectly reduce foraging habitat and food relied upon by San Francisco Bay bird colonies. Cumulative impacts of increasing pressures, including the Revised Project activities in key seabird foraging habitat, are therefore likely to significantly increase negative pressures on San Francisco Bay bird colonies.

Environmental changes have caused habitat compression throughout central California, reducing suitable offshore habitat and concentrating marine wildlife populations inshore[2]. This has caused profound habitat shifts for wildlife such as seabirds and marine mammals which previously utilized both nearshore and offshore regions; wildlife has moved inshore into a much smaller area in which to breed and feed[3]. This inshore movement has occurred despite higher concentrations inshore of human disturbance, disruption, and direct harm[2,3]. In other words, instead of previous breeding and feeding locations which occurred more distant from human activities, wildlife is now risking dangerous encounters with human activities due to insufficient food elsewhere.

Wildlife is therefore under increasing pressure due to both climate change and increasing human population/activities. Cumulative impacts of increasing pressures, including the Revised Project activities in key wildlife foraging habitat, are therefore likely to cause significant negative effects. Yet, this SEIR makes no attempt to analyze available evidence of the last 10-12 years to assess potential impacts to biological resources, especially seabirds and other marine wildlife. Available evidence was not analyzed as a part of the technical studies and reports on which the SEIR bases its erroneous statement on page II-25 that "the approval of the Revised Project would not significantly interfere with the trusts upon which public lands are held..." Furthermore, we should mitigate human activities since we cannot control climate change impacts. Disregarding available evidence of the last 10-12 years completely ignores the agency's mandate to evaluate the potential of such hazards on future residents and users.

Impacts to the smaller wildlife breeding and feeding areas resulting from habitat compression are proportionally much more important for a species. For example, Brandt's cormorants normally breed and feed, and humpback whales normally feed, offshore out to the continental shelf break and the Farallon Islands; however, both of these species have increasingly moved inshore and specifically into Central San Francisco Bay[2,3]. For cormorants, the proportion of the regional meta-population inside the Central Bay versus outside the Central Bay in the Gulf of the Farallones has been increasing and hit a record high in 2024[4]. These spatial shifts have been linked to food availability. Reducing available food and foraging opportunities in this remaining habitat of the Central Bay is likely to have negative consequences on marine predators, as this habitat is already affected by climate change and other stressors caused by concentrated human activity.

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[2] Santora et al. 2020. https://doi.org/10.1038/s41467-019-14215-w
Schroeder et al. 2022. https://doi.org/10.1016/j.ecolind.2022.109520

[3] Ainley et al. 2018. https://doi.org/10.1016/j.biocon.2017.11.021
Markowitz et al. 2024. https://doi.org/10.1002/aqc.4107

[4] Post et al. 2024. Breeding Ecology of Brandt's Cormorants and Western Gulls on Alcatraz Island, 2024. Final Report to the Golden Gate National Recreation Area (GGNRA) National Park Service (NPS).

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Seabirds are constrained to a central place (the breeding colony) during incubation and chick rearing, and thus must feed themselves and their offspring with prey obtained in a small radius around the colony. Alcatraz Island cormorants are almost exclusively observed flying to/from more marine waters of the west-Central Bay during foraging trips, not towards more estuarine waters of the east Central or Upper/Lower Bays[5]. The mean foraging radius of Brandt's cormorants is 10km (~6 miles[6]). The distance from Alcatraz to the Golden Gate Bridge is ~3.5 miles. Thus by approximation, the Central Bay sand mining lease areas overlap with roughly 50% of the feeding area for Brandt's cormorants breeding at Alcatraz Island.

Sand mining activities in the Central Bay have caused direct impacts and also potentially indirect impacts for cormorants and other seabirds foraging pelagically in flocks (including Western gull, Western grebe, common murre, brown pelican, various tern species, and California sea lion, as well as other wildlife such as pigeon guillemots, pelagic cormorants, harbor seals, and gray whales which forage benthically in the Central Bay sand mining lease areas[5,7]. Sand mining activities can directly disturb both foraging flocks and foraging individuals, and create turbid water plumes observed for many daylight (e.g., foraging) hours, which inhibit foraging by visual predators such as seabirds. Sand mining activities also disturb benthic habitat around Alcatraz and likely reduce seabird food supply in the limited foraging range around the breeding colony (since seabirds must return to provision offspring many times each day).

In years of poor marine conditions, cormorant breeding population size and reproductive success on Alcatraz can drastically decrease. Notable examples of this were in 2009-2012, 2016, 2019, and 2024[7].

Thus, the Revised Project may potentially have a more significant negative effect on Alcatraz seabirds and other wildlife which are restricted to foraging habitat around a central place (e.g., seabird breeding colony, seal haulout), particularly during years of poor marine conditions.

p. II-32. "One commenter noted that National Park Service data indicates that 4,825 nesting Brandt's cormorants (*Phalacrocorax penicillatus*) were present on Alcatraz Island in 2023. This is approximately a 25 percent increase over the 3,900 birds mentioned in the SEIR (Section 3.2, page 3.2-5). The commenter's statement further validates the information presented in Section 3.2.2.3 of the SEIR that sea bird colonies on Alcatraz Island have seen an upward trend likely due to prey availability and reproductive success. As the evidence suggests, these species do not appear to be lacking in prey or adequate foraging habitat

[5] Saenz et al. 2001. Telemetry of Alcatraz-Breeding Brandt's Cormorants, 2001. Final Report to the Golden Gate National Recreation Area (GGNRA) National Park Service (NPS).
J. Thayer/Farallon Institute, pers. obs.

[6] Briggs et al. 1987. Bird communities at sea off California 1979-1983. Studies in Avian Biology No. 11.
Peck-Richardson et al. 2017. Three-dimensional foraging habitat use and niche partitioning in two sympatric seabird species, *Phalacrocorax auritus* and *P. penicillatus. Marine Ecology Progress Series.* https://doi.org/10.3354/meps12407

[7] Thayer et al. 2025. Breeding Ecology of Brandt's Cormorants and Western Gulls on Alcatraz Island, 2025. Final Report to the Golden Gate National Recreation Area (GGNRA) National Park Service (NPS).

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nor does the Revised Project appear to pose any direct or indirect impacts on these colonies. Even with the acknowledgement that regional population changes have occurred, the evidence indicates that the Alcatraz colonies are healthy and growing, and as one commenter stated, "…Alcatraz numbers still exceeded those of the largest colony for this species (SE Farallon Island) despite restrictions at/around SEFI on fishing, light pollution, benthic disturbance, and human access/disturbance." The Revised Project would not displace cormorant populations or otherwise result in significant nesting impacts as the Revised Project would not occur onshore. As discussed in Section 3.2.5.2 of the SEIR, minor direct impacts to the free movement or foraging of birds, fish, and other marine species could occur during active sand extraction activities. As disclosed in the SEIR, the temporary displacement of birds and other wildlife species from active mining areas would have a less-than-significant impact.

This conclusion is erroneous. First, there is not "free movement" of foraging birds breeding on Alcatraz. As described above, breeding seabirds are central place foragers since they have to return many times per day to the breeding colony to participate in nest duties and provision young[8]. And second, it is well established in the literature that reduction of seabird prey around a breeding colony negatively impacts the colony[9], so the assertion that there would be no significant impacts because the Revised Project does not occur onshore is meaningless.

As stated above, by approximation, the Central Bay sand mining lease areas overlap with roughly 50% of the feeding area for Brandt's cormorants breeding at Alcatraz Island. Sand mining activities in the Central Bay have caused direct impacts through dispersal of predator feeding flocks and creation of turbid water plumes inhibiting visual foraging[5,7], and likely indirect impacts due to disruption of the benthos and mortality of larval and juvenile fishes in this important estuarine fish nursery area.

Furthermore, the response above about "healthy populations" is out of context and does not follow a logical argument. The response mentions 3900 cormorants in 2023 but provides no citation in the SEIR; this number is incorrect, so it is meaningless to calculate any percentage between an incorrect number and the documented number of cormorants on Alcatraz in 2023 (which was 4,825[7]; the original comment was intended to correct the erroneous information in the Draft SEIR). At any rate, there was an 8% increase in the population from 2022 to 2023. The subsequent year there was a severe population decrease of -288% from 2023 to 2024. And the response about Alcatraz numbers exceeding Farallon Island numbers in 2024 indicates not a healthy population given such severe decreases, but simply a more drastic decrease on the Farallon Islands than on Alcatraz.

Valid ecological interpretation is completely lacking in this response.

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[8] See Orians, GH & NE Pearson. 1979. On the theory of central place foraging. In: Horn, DJ, RD Mitchell, GR Stairs (Eds.), *Analysis of Ecological Systems*.

[9] Anderson et al. 1982. https://www.jstor.org/stable/3544527?origin=crossref
Gulka et al. 2020. https://www.jstor.org/stable/26988303
Fayet et al. 2021. https://besjournals.onlinelibrary.wiley.com/doi/pdf/10.1111/1365-2656.13442

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Fluctuations in both cormorant breeding population size and reproductive success have been increasing[4,7], meaning wider and wider variations in numbers of breeding birds and numbers of surviving offspring, parameters both related to available prey around the colony. These variations have gotten particularly wider in the past 15 years; extreme fluctuations from 0 to 15,000 birds, and from a mean of 0 to more than 3 chicks fledged per pair. Variations in prey fish upon which cormorants depend (including anchovy and other benthic fishes) have also gotten more pronounced[10]. This is not surprising given climate change, but increasing variability is not a positive outlook. Reducing even further these already increasingly-variable prey resources will likely significantly negatively impact Alcatraz cormorants and other wildlife dependent on these resources in years of poor marine conditions (such as El Niño events or marine heat waves) when survival margins are thin and marine predators are most vulnerable (see Gibble et al. 2010 and Piatt et al. 2020 on cormorant and other seabird extreme mortality events)[11].

The evidence directly opposes the statement in the Final SEIR that "these species do not appear to be lacking in prey or adequate foraging habitat, nor does the Revised Project appear to pose any direct or indirect impacts." The Revised Project would potentially directly displace and negatively impact foraging cormorants and other wildlife restricted to foraging around a central place (e.g., seabird breeding colony, harbor seal haul-out, etc.), and result in significant breeding and survival impacts through negative effects on the benthos and prey surrounding the central place.

Context previously provided in comments to the Draft SEIR:

Climate change has been increasing rapidly since 2012, from changes in basic ocean physics (e.g., upwelling along the California coast[12]) to increasing frequency and intensity of marine heat waves[13], and cumulative impacts of El Niño events, Pacific Decadal Oscillation extremes and global warming[14]. Scientists are now viewing a 1.5°C temperature increase as inevitable, with severe effects on marine ecosystems[15].

The Central San Francisco Bay (SFB) is an important transition zone from the ocean to estuary. Such areas have exceptionally high biodiversity and trophic transfer (the transfer of energy and nutrients from the physical system up to top predators), supporting a richer community than either adjacent

[10] Raimonet & Cloern 2017. https://doi.org/10.1111/gcb.13546,
Thayer et al. 2022. https://doi.org/10.5281/zenodo.6385640
Hieb et al. 2025. https://doi.org/10.6073/pasta/50dd4452a476db0aa3521ecdbb9a7a51

[11] Gibble et al. 2010. The 2009 central California Brandt's Cormorant (*Phalacrocorax penicillatus*) die-off and breeding failure event. [Abstract] Long Beach, CA: Pacific Seabird Group 2010 annual meeting, 17-20 February 2010
Piatt et al. 2020. https://doi.org/10.1371/journal.pone.0226087

[12] Bograd et al. 2023. https://doi.org/10.1146/annurev-marine-032122-021945

[13] Oliver et al. 2018. https://www.nature.com/articles/s41467-018-03732-9

[14] Sen Gupta et al. 2020. https://www.nature.com/articles/s41598-020-75445-3

[15] Santana-Falcón et al. 2023. https://doi.org/10.1038/s43247-023-01002-1,
Reisinger et al. 2025. https://doi.org/10.1146/annurev-environ-111523-102029

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marine or estuarine waters separately [16]. Despite this richness, the Central SFB lacks appropriate protections against anthropogenic threats, largely due to the complex nature of the system and challenges translating science to management and policy[17].

The Central SFB is being utilized by increasing numbers of marine organisms, serving as a climate refuge when ocean conditions are poor, including endangered species and benthic foraging species[18]. Sand mining effects have been inadequately considered on the myriad species which overlap with the mining lease areas in the Central SFB. The SEIR has not adequately considered:

- Central SFB and especially West-Central SFB concentrations of certain species
- Central SFB species' demographic changes since 2012
- Regional context, especially important since the Central SFB is a migration corridor through which all species moving in/out of the bay have to pass, and it is a transition zone with high use by marine as well as bay species
- Metapopulation changes, particularly important for endemic species, and/or those species which are declining in other portions of their range (making the Central SFB relatively more important)
- Proportion of overlap of mining lease areas, and cumulative duration of mining activities, with respect to organisms which are increasingly using the Central SFB and especially regarding limited-range organisms
- Appropriate analyses or summaries of rare, seasonal, or intermittently-pulsed species, conducted in a manner by which to be able to detect signals of change

Additionally, the SEIR contains unreferenced citations and instances of unreliable/unscientific citation sources (e.g., SF Gate). We have encountered inaccurate information in the SEIR, and instances where important data was omitted.

The Central SFB is a dynamic habitat and organisms are specifically adapted and attracted to this. Some studies exist - studies since 2012 investigated physical movement of sand, and limited (and flawed) studies on low-taxa biology (benthic infauna). Changes to abundance or ecology of demersal fishes, crabs, and benthic-pelagic webs have not been evaluated, yet studies of biological effects of sand mining activities on these higher taxa are possible. Direct studies of use of similar highly-dynamic sand wave

[16] Cloern et al. 2007 www.pnas.org_cgi_doi_10.1073_pnas.0706151104;
Raimonet & Cloern 2017 https://doi.org/10.1111/gcb.13546,
Intergovernmental Panel on Climate Change 2019 https://www.ipcc.ch/srocc/chapter/chapter-5/,
Preston et al. 2025 https://doi.org/10.1038/s44183-025-00128-3
[17] Columbano et al. 2025 https://doi.org/10.15447/ sfews.2025v23iss3art1
[18] Saenz et al. 2006, https://doi.org/10.5038/2074-1235.34.1.681,
Stern et al. 2017 https://doi.org/ 10.1578/AM.43.6.2017.691,
Seher et al. 2022, https://doi.org/10.3354/meps14028,
Markowitz et al. 2024, https://doi.org/10.1002/aqc.4107

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habitats in other bays/estuaries have yielded surprising results, including the high importance of these habitats for many species, including groundfish and salmon[19].

In summary, inadequate characterization of the above species, population demographics, food web interactions and spatio-temporal information makes it impossible to adequately consider biological or ecosystem effects of sand mining activities.

[19] Greene et al. 2017, https://doi.org/10.3390/geosciences7040107, Greene et al. 2020, https://doi.org/10.1016/B978-0-12-814960-7.00014-2, Greene et al. 2021, https://doi.org/10.1016/oceano.2121.06.002, Chaliflour et al. 2019, https://doi.org/10.3354/meps13064; Baker et al. 2021, https://doi.org/10.1016/j.fishres.2021.106067, Baker et al. 2024, https://doi.org/10.2139/ssrn.4896945

The Marine Mammal Center®

February 4, 2026

California State Lands Commission
100 Howe Avenue, Suite 100-South
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Via Email: CSLC.CommissionMeetings@slc.ca.gov
Attn: Christopher Huitt, Senior Environmental Scientist

Dear Mr. Huitt,

We appreciate the opportunity to submit comments on the State Lands Commission Supplemental Environmental Impact Report (SEIR) for the San Francisco Bay and Delta Sand Mining Project prior to the Commission's upcoming meeting on February 9, 2026. As described in the SEIR, the environmental review is limited to evaluating the current revised project against the project approved in the 2012 Environmental Impact Report (EIR) and concludes that only minor changes are necessary for the 2012 EIR to remain applicable. However, there are essential issues and changed circumstances that are not adequately addressed, or not addressed at all, in the SEIR and supporting biological documentation, including Appendix G (Biological Resources Supplemental Information, August 2025). These deficiencies must be addressed for the environmental analysis to be adequate under CEQA. In particular, the environmental document does not sufficiently account for the recent, regular use of San Francisco Bay by species newly listed under the U.S. Endangered Species Act, nor the substantial decline of existing threatened and endangered species that has occurred since the prior review, including humpback whales (*Megaptera novaeangliae*) and gray whales (*Eschrichtius robustus*), among others discussed below. The environmental review requires a new, thorough evaluation of potential impacts.

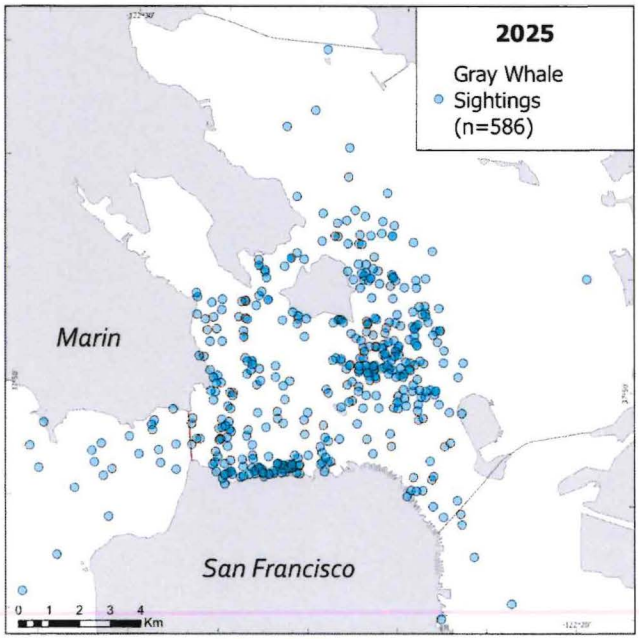
As subject matter experts with respect to the use of San Francisco Bay by large whales, we can provide additional relevant input that relates to biological resources, both aquatic species and habitat. We are concerned that the SEIR, and specifically Appendix G Biological Resources Supplemental Information dated August 2025, does not adequately describe the environmental impacts of the project.

The Marine Mammal Center (The Center) is the world's largest marine mammal teaching hospital, treating hundreds of marine mammal patients each year, and as such is an ocean conservation organization with a global impact. With more than fifty years of experience, we learn from every patient in our care about their health, the health of their species and their ocean home, and about the connection to human health. We are on the front lines of ocean change and are witness to the many stressors that marine mammals face along the California coast and in Hawai'i.

The Center monitors the presence and health of marine mammals in San Francisco Bay, and we have observed an increased use of the Bay by two species of large whales of note: humpback whales (*Megaptera novaeangliae*) and gray whales (*Eschrichtius robustus*).

Humpback Whales: Beginning in 2016, we observed an unprecedented influx of humpback whales into the Bay, and our research efforts showed they commonly overlap the Central Bay sand mining lease parcels. Our photo-identification of individuals confirmed these whales belong to two distinct population segments listed as endangered and threatened under the U.S. Endangered Species Act. Since 2016, we have seen the whales return every year seasonally (April–November) to use the Bay as foraging habitat. The SEIR Appendix G states their regional occurrence is seasonal (May through November) and their potential presence in the study area is "Moderate-High" as "recently, humpback whales have been observed more frequently feeding in the Central Bay." We have frequently observed these whales lunge-feeding on northern anchovy (*Engraulis mordax*), a fish species which the SEIR lists as present in the sand mining locations. Our detailed findings on humpback whales in the Bay have been published in a peer-reviewed journal (Markowitz et al., 2024).[1]

Gray Whales: Beginning in 2018, we noticed a sharp increase in members of the Eastern North Pacific population of gray whales entering the Bay, a phenomenon that appeared to be linked to a NOAA-declared Unusual Mortality Event (2019-2023) for gray whales. Every year since 2018, we have surveyed the Bay to track the presence and locations of gray whales, and we have found these whales occur annually from February through June. Our photo-identification effort has resulted in a catalog of 120 individual gray whales, supplemented by compiled visual sightings. Photo-identification records confirmed that several individuals have remained in San Francisco Bay for several weeks, even months (the maximum residency time reached 75 days). During 2025, the average observation period per whale was 24 days, measured from first to last observation. Our most recent map of gray whale sightings, focusing on the 2025 season, is presented below.



[1] Markowitz, T.M., Keener, W., Webber, M.A., Payne, A.R., Lane, R.S., Fahlbusch, J.A., & Calambokidis, J. (2024). New urban habitat for endangered humpback whales: San Francisco Bay. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 34(3), e4107. https://doi.org/10.1002/aqc.4107

We have also noted that many of these whales in the Bay are in poor body condition, appearing to have reduced body fat (blubber). Gray whales are notable for two reasons. First, they have a very long roundtrip migration (~16,000 – 22,000 km) from breeding grounds in Mexico to feeding habitat in Alaska, and during migration they generally do not engage in feeding. Second, unlike most whales that feed in the water column, gray whales specialize in bottom feeding; their main source of food comes from invertebrate prey species living on the seafloor bottom or as benthic infauna. The SEIR Appendix G states their regional occurrence is seasonal (February through May) and their potential presence the study area is noted as "Rare-Low" in the study area, while mentioning "there has been an increase in sightings of foraging gray whales in Central Bay near the Golden Gate."

The foraging behavior we have observed suggests that many of the gray whales stopping along their migration route to enter the Bay are thin and hungry, and thus attracted to the potential food resource in the bayfloor substrate. Specifically, we have tracked whales in the Central Bay that dived to the bottom where they generated mud plumes as they attempted to suction feed and presumably strain food items through their baleen plates. To shed light on whether these whales were actually ingesting prey species, we are collaborating with the Smithsonian Environmental Research Center in Marin County to characterize stomach contents from eight deceased gray whales that stranded in San Francisco Bay since 2019. Preliminary results of the macro- and microscopic analyses, as yet unpublished, revealed that the gray whale diet in the Bay consists of epibenthic species and benthic infauna, primarily crustacean gammarid amphipods, decapods (crabs) and isopods, as well as some eelgrass and algal sea lettuce.

It is our understanding that sand mining using suction dredging has a deleterious effect on benthic invertebrate communities, and therefore the operations are in competition with the gray whales. The loss of a significant food resource for gray whales, already nutritionally stressed, across the combined area of the lease parcels in the Central Bay warrants further analysis by the Commission. Although the SEIR mentions the increased use of the Bay by gray whales, it does not detail the potential impacts of sand mining on this charismatic species, California's official state marine mammal. Also, the Central Bay has seen new use by U.S. Endangered Species Act listed humpback whale subpopulations that forage in the water column on northern anchovy. Their disturbance by sand mining operations is also worthy of additional analysis.

We trust the Commission will be open to considering impact mitigation alternatives to protect large whales in the Bay, such as reductions in the areal extent or volume of material mined, and limiting operations to months when fewer whales tend to be present (November to February). The Marine Mammal Center is open to working with the Commission and stakeholders to develop suitable science-based approaches to this important issue.

We look forward to your response to our comments. Should you have any questions regarding the information presented here, please contact me by email at majerusc@tmmc.org.

Sincerely,

Cecily Majerus

Cecily Majerus, CEO

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SAN FRANCISCO BAYKEEPER®

February 4, 2026

Transmitted via Electronic Mail

Honorable Commissioners
California State Lands Commission
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Sacramento, California 95825-8202
CEQA.comments@slc.ca.gov

Re: Please Reject Certification of the Supplemental Environmental Impact Report for San Francisco Bay and Delta Sand Mining Project (State Clearinghouse No. 2007072036)

Dear Commissioners:

On February 9, 2026, you will be asked to certify a Supplemental Environmental Impact Report (SEIR) that authorizes continued commercial extraction of sand from San Francisco Bay. On behalf of San Francisco Baykeeper's 5,000+ members and supporters, I respectfully urge you to reject this certification.

The science has fundamentally changed since 2012, and continuing to mine the Bay's limited sand at these unsustainable levels would violate your public trust obligations to manage California's natural resources for the benefit of all citizens.

Reject the Certification Because New Studies Show These Companies are Mining Ancient, Irreplaceable Sand from the San Francisco Bay Floor

Recent breakthrough scientific studies commissioned by the U.S. Geological Survey, the California State Coastal Conservancy, and the San Francisco Bay Conservation and Development Commission (BCDC) have revealed what Bay sand mining extracts. This is not renewable sand being replenished regularly by rivers—it is ancient sand from deposits that took millennia to form and cannot be replaced under current environmental conditions.

The Independent Science Panel (ISP) synthesis concludes that Central Bay sand is relic, deposited roughly 6,000 to 20,000 years ago as sea levels rose and the river discharge point migrated through the Bay. The ISP further concludes that sand is mined faster than sediment is replenished, and that Bay sand is therefore non-renewable over the long term.

Every cubic yard extracted represents permanently depleting irreplaceable geological sand that accumulated over centuries. The Bay's ancient sand took millennia to form. Once it is taken out of the Bay, it is gone forever.

The ISP, formed at the request of the SF Bay Conservation and Development Commission (SF BCDC), conducted a comprehensive research program that documented that sand mining constitutes the single largest sand loss from the Bay system:

- Current authorized extraction: 2.04 million cubic yards annually
- Natural sand supply from all sources: ~460,000 cubic yards annually
- Mining rate: approximately 4 times greater than natural replenishment

This represents fundamentally unsustainable resource extraction that creates permanent sediment deficits year after year. Since operations began in the 1930s, over 100 million cubic yards have been extracted — this is sand that will never return to the Bay system.

Reject the Certification Because Your Public Trust Duty Requires Protection of Public Resources

The California Supreme Court has made clear that the State holds tidelands and submerged lands in trust for the benefit of all the people. As trustees, you have a fiduciary duty to preserve these resources for public uses including navigation, commerce, fisheries, recreation, and environmental protection. You cannot subordinate these public interests to private commercial profit.

The SEIR reveals a direct conflict with your trust obligations.

San Francisco has invested $175 million in the Ocean Beach Climate Adaptation Project to protect vital wastewater infrastructure threatened by coastal sand erosion. The city must continuously replenish this sand at costs exceeding $1 million per application to combat the erosion. Meanwhile, the State proposes to authorize private companies to extract the very sand that creates this public crisis. Facilitating private profit by removing sand that the public must pay to replace represents a breach of your fiduciary duty as trustees.

BCDC — your co-trustee agency with primary jurisdiction over Bay resources — criticized the SEIR for failing to incorporate the ISP synthesis and emphasized the ISP's findings that 1) mined sand is the largest outflow of sand from the Bay, 2) sand is mined faster than it is replenished (making it non-renewable over the long term), 3) Suisun Bay sand is not replenished and the bed is being lowered, and 4) that Central Bay sand is relic — all while effects of mining to beaches and ecologically important shoals remain unquantified.

Reject the Certification Because Endangered Species that Depend on Bay Sand will be Impacted

The environmental baseline has changed dramatically since 2012. Marine mammal populations that were absent or minimal at the time the original EIR was issued now depend on Bay resources for survival:

- **Gray Whales:** Now feed in Central Bay mining areas in a period coinciding with a NOAA-designated Unusual Mortality Event, claiming ~700 North Pacific gray whales.
- **Harbor Porpoises:** Returned to the Bay after 60-year absence and are now year-round residents.
- **Humpback Whales:** Now use the Bay as seasonal foraging habitat; whales photographed in the Bay included individuals from ESA-listed threatened and endangered distinct population segments.
- **Southern Resident Killer Whales:** Critically endangered population of only 74 individuals now documented in Bay waters.

These federally protected species depend on benthic prey communities in the very areas targeted for continued sand mining. The Marine Mammal Protection Act and Endangered Species Act require consultation and analysis regarding the impacts to these species, but the SEIR defers this analysis to future permitting. This is an improper deferral of environmental review under CEQA.

Reject the Certification Because Climate Change Makes Continued Sand Mining Untenable

Scientific evidence shows that atmospheric rivers — the storms that drive Bay and coastal erosion — are intensifying under climate change, with precipitation projected to increase 26-37% by 2100 in the Bay Area. South Ocean Beach has already experienced severe erosion over the past 15 years, with storm-driven waves threatening critical infrastructure.

Authorizing the continued extraction of irreplaceable ancient sand in the face of accelerating climate-driven erosion is fundamentally incompatible with protecting California's coast and Bay system for future generations.

Reject the Certification Because it is Contrary to California's International Conservation Commitments

In October 2025, Governor Newsom announced that California would join the International Union for Conservation of Nature (IUCN) as a subnational member — among the first US states to do so. The United States itself is an IUCN member through the Department of State and multiple federal agencies including NOAA, U.S. Fish and Wildlife Service, and the National Park Service.

At the 2021 IUCN World Conservation Congress in Marseille, member governments and organizations adopted Resolution WCC-2020-Rec-029: "For the urgent global management of marine and coastal sand resources." This resolution recognizes the global crisis of unsustainable sand extraction and calls for improved governance of coastal and marine sand resources.

By joining IUCN and accepting the responsibilities of membership, California has signaled its commitment to align with international conservation standards. Authorizing another decade of unsustainable sand mining in San Francisco Bay would directly contradict California's stated commitment to IUCN principles and Resolution 029's call for urgent action on coastal sand resource management.

Your decision on sand mining will demonstrate whether California's IUCN membership represents a genuine commitment to international conservation leadership or merely symbolic participation.

Reject the Certification Because Viable Alternatives Exist

The SEIR summarily dismisses a "No Project" alternative and fails to analyze reasonable alternatives such as:

- Phased reduction of mining to sustainable levels matching natural replenishment.
- Geographic restrictions protecting areas of highest ecological value or sediment transport significance.
- Temporal restrictions during critical biological periods for marine mammals and fisheries.

- Enhanced mitigation requiring sand replacement or marsh restoration to offset losses.

CEQA requires meaningful analysis of alternatives that could reduce environmental impacts. The SEIR's cursory dismissal violates this core requirement.

Reject the Certification Because Your Decision Will Set a Precedent

You are being asked to authorize another decade of extracting ancient, irreplaceable sand at rates that exceed natural supply by 4:1, despite breakthrough scientific evidence showing this practice is fundamentally unsustainable.

Your decision will demonstrate whether California's public trust doctrine and IUCN commitments have substance or are merely aspirational.

Other jurisdictions have acted on similar science. The Netherlands prohibited commercial sand extraction from the ecologically sensitive Wadden Sea, transferring all operations to offshore areas beyond 20-meter depth. German states prohibit sand extraction from Wadden Sea National Parks. These decisions recognize that some natural resources are too valuable and irreplaceable to sacrifice for commercial profit.

Recommendation

For the reasons detailed in our comprehensive technical comment letter attached here, San Francisco Baykeeper respectfully urges the Commission to:

1. Reject certification of the SEIR as inadequate under CEQA, and

2. Direct staff to prepare a revised SEIR that:
    - Accurately analyzes the significance of extracting ancient, non-renewable sand resources.
    - Properly evaluates impacts to endangered species under changed baseline conditions.
    - Analyzes reasonable alternatives including phased reduction to sustainable levels.
    - Complies with the public trust doctrine by prioritizing public benefits over private profit.

The Independent Science Panel has provided the definitive scientific evidence. The question is whether you will act on that evidence to protect California's irreplaceable natural heritage, or whether you will authorize another decade of unsustainable extraction that benefits private companies while imposing costs on the public.

Please protect the Bay’s sand as a public trust resource for all Californians and for future generations.

Most Sincerely,

Sejal Choksi-Chugh
Executive Director

February 4, 2026

Transmitted via Electronic Mail

California State Lands Commission
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CEQA.comments@slc.ca.gov

Re: Comments on Final Supplemental Environmental Impact Report for San Francisco Bay and Delta Sand Mining Project (State Clearinghouse No. 2007072036)

Dear Commissioners and Staff:

San Francisco Baykeeper submits these comments on the *San Francisco Bay and Delta Sand Mining Project: Final Supplemental Environmental Impact Report* ("Final SEIR" or "FSEIR") for the San Francisco Bay and Delta Sand Mining Project ("Project"). These comments supplement Baykeeper's September 29, 2025 comments on the Draft Supplemental Environmental Impact Report ("Draft SEIR").[1]

1. Introduction

The California State Lands Commission ("CSLC," "State Lands," or the "Commission") has prepared the Final SEIR pursuant to the California Environmental Quality Act (CEQA)[2] to evaluate proposed lease renewals that would authorize commercial sand mining operations in San Francisco Bay through 2035. This Final SEIR responds to public comments on the Draft SEIR but fails to adequately address substantial evidence demonstrating that the Project will cause significant environmental impacts not disclosed in the 2012 Environmental Impact Report ("2012 EIR") or the Draft Supplemental Environmental Impact Report: San Francisco Bay and Delta Sand Mining Project, State Clearinghouse No. 2007072036 ("Draft SEIR"), and fails to adequately identify, describe, or require mitigation of those impacts.

These comments address critical deficiencies in the Final SEIR's environmental analysis, including:

(1) Arbitrary reclassification of significant impacts without factual justification;
(2) Improper baseline manipulation;
(3) Failure to account for substantial new scientific evidence and changed circumstances; and
(4) Inadequate consideration of public trust obligations.

[1] Draft and Final 2011/2012 EIRs for the San Francisco Bay & Delta Sand Mining Project available at: https://www.slc.ca.gov/ceqa/sf-delta-sand-mining/.

[2] Cal. Pub. Resources Code § 21000 et seq.; 14 Cal. Code Regs. § 15000 et seq.

2. The Final SEIR Arbitrarily Reclassifies Impact BIO-8 from Significant and Unavoidable to Potentially Significant Without Substantial Evidence

CEQA requires lead agencies to make significance determinations based on substantial evidence in the record.[3] The improper deferral of significance determinations violates CEQA's fundamental purpose of informed public participation and decision-making. The 2012 EIR classified Impact BIO-8 (entrainment and mortality of Delta Smelt and Longfin Smelt) as "Significant and Unavoidable" (Class I).[4] The Draft SEIR maintained this classification, acknowledging that "there still remains a potential for take of listed delta or longfin smelt to occur, and therefore the impact remains a Significant Impact (Class I)."[5]

The Final SEIR downgrades this impact to "Potentially Significant" (Class II) in Table ES-1 without adequate justification.[6] Specifically, the downgrade of Impact BIO-8 from Class I to Class II relies on Incidental Take Permits (ITPs) that are unchanged from the Draft SEIR. The Final SEIR never reconciles how identical equipment and operations avoid significant smelt take and constitute new mitigation for White Sturgeon, which Impact BIO-9 admits will suffer potential significant impacts.[7] This inconsistency in the Final SEIR, without any explanation or evidence for the change, violates CEQA and demonstrates a gap in the proposed mitigation measures.

a. The Final SEIR Fails to Adequately Respond to Comments

Comment responses 08-30, 08-32, and 08-42 address Baykeeper's comments on Delta and Longfin smelt. According to Comment response 08-30, "based on consideration of updated ITPs and approved mitigation measure variances (described in Appendix F – Existing 2012 Project Mitigation (as Amended) and Incidental Take Permits), the impact conclusion for BIO-8 has been downgraded to Class II (Potentially Significant). See response 07-3 for details and updated SEIR text."

A Final SEIR must respond substantively to comments. Baykeeper asserted that there were "internal contradictions regarding impact significance that violate CEQA's requirement for clear, consistent analysis," and specifically listed where those contradictions occurred. Essentially, the SEIR states that there will be no new significant impacts and no increase in impacts from the 2012 EIR, yet several significant impacts are listed, both in the Draft and Final SEIRs. *See* Baykeeper Comments at 17. The only response to this comment in the Final SEIR is that "[n]o contradiction is indicated between the Executive Summary and the analysis of direct and indirect effects of the Revised Project." Comment Response 08-30. There is nothing else substantive in the Response.

[3] Cal. Pub. Resources Code § 21082.2.

[4] California State Lands Commission, Final Environmental Impact Report: San Francisco Bay and Delta Sand Mining Project (2012), § 4.1.3.

[5] Draft SEIR, § 3.2.5.2, p. 3.2-31.

[6] Final SEIR, Table ES-1, Part III (2025).

[7] Final SEIR, § 3.2.5.2, 3.2-35 to 36 (added language in Final SEIR relying on ITP conclusion of no take for salmonids to conclude that "no additional mitigation is required for impacts on green sturgeon, white sturgeon, Chinook salmon, or steelhead trout").

Impacts listed in the SEIR are, by definition, new impacts as compared to the 2012 analysis. They are not impacts from the mining that occurred 20 years ago, but rather predicted impacts from sand mining that has yet to occur, including impacts to Longfin Smelt and Delta Smelt. In addition, Baykeeper's, and many others', comments addressed a wide range of new impacts that were not addressed in the Response to Comments.

b. Minimal CEQA standard (significance must be supported in the SEIR record)

CEQA requires the lead agency to disclose and support its significance conclusions with substantial evidence and to avoid a "permit-now/analyze-later" approach that relies on future approvals in lieu of clear, enforceable mitigation commitments and standards in the SEIR.[8] The SEIR must also serve CEQA's informational function—an EIR is an "environmental alarm bell" intended to alert the public and decision-makers to environmental consequences before approval.[9]

The "new" evidence cited in the FSEIR is merely the existing ITPs and fish screens, which were already present when the DSEIR declared the impact significant.[10] The Final SEIR therefore lacks substantial evidence and continues to fail to adequately analyze changed conditions since 2012 (e.g., species have continued to decline, increasing significance of impacts; updates to critical habitat and recovery plans; and new science on climate change impacts).

c. The Final SEIR record supports Class I—not a downgrade

The Draft SEIR classified Impact BIO-8 (entrainment and mortality of Delta and Longfin smelt) as Significant (Class I) and stated that "there still remains a potential for take… and therefore the impact remains a Significant Impact (Class I), consistent with the 2012 EIR."[11] If the Final SEIR now lists BIO-8 as Class II ("Potentially Significant") in the Executive Summary table, that is a material change that must be supported by new, clearly explained evidence in the record for the Final SEIR.[12] Absent a documented analytic basis (e.g., a new quantitative entrainment demonstration or enforceable mitigation performance standards shown to reduce impacts below the significance threshold), the reclassification is illegally arbitrary.

d. Future ITP actions do not substitute for the Commission's CEQA duty to disclose residual impact significance

To the extent the Final SEIR suggests that future incidental take permitting by California Department of Fish and Wildlife will reduce BIO-8 to below-significant levels for CEQA purposes, that reasoning is

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[8] California Code of Regulations, title 14, § 15126.4(a)(1)(B) (mitigation discussion; limits on deferring mitigation/analysis without enforceable standards and commitments). Legal Information Institute, Cornell Law School. Retrieved January 30, 2026, from https://www.law.cornell.edu/regulations/california/14-CCR-15126.4

[9] *Laurel Heights Improvement Assn. v. Regents of Univ. of Cal.* (1988) 47 Cal. 3d 376, 392.

[10] *See, e.g.*, Final SEIR Errata p. 13, striking language describing the Project's significant impact due the entrainment and mortality of Delta and Longfin smelt even with the fish screen in place and adding the following: "because the Revised Project operations include use of the fish screen, the potential for take of listed delta or longfin smelt is substantially reduced when compared to the 2012 Project."

[11] Draft SEIR, Impact BIO-8 discussion and Class I conclusion.

[12] Final SEIR, Executive Summary / Table ES-1 listing for Impact BIO-8.

inadequate. The Commission must make and support the CEQA significance determination in the EIR itself. However, the Final SEIR fails to identify any new quantitative entrainment analysis, monitoring results, or enforceable performance thresholds demonstrating that incidental take is reduced below CEQA significance thresholds. Instead, it relies on the same mitigation regime that the Draft SEIR expressly found inadequate to avoid a Class I impact.

Relying on future consultations and permits to determine significance and mitigation does not satisfy CEQA. CEQA requires agencies to analyze and shape the project to reduce or avoid environmental impacts before deciding to approve it.[13] Likewise, agencies cannot defer formulation of mitigation measures to the future.[14] To the extent mitigation is deferred, it must adhere to specific performance standards.[15]

There are no specific performance standards for this mitigation described in the Final SEIR: "As determined by CDFW in their issuance of an ITP for aggregate mining activities, Applicants will provide compensatory mitigation to a CDFW-approved mitigation or conservation bank covering smelt habitat." FSEIR 3.2-35. Moreover, State Lands relies on ITPs (which by the text of the Final SEIR itself only address "'taking' of salmonid species") for the conclusion that no additional mitigation is required. But no ITP has been issued yet for White Sturgeon, and Green Sturgeon are not salmonids (sturgeon belong to the family *Acipenseridae* and are considered primitive, cartilaginous fish, whereas salmon and trout belong to the *Salmonidae* family). FSEIR 3.2-36. Moreover, CDFW's own comments note population crashes despite these measures; the FSEIR ignores the agency with expertise, CDFW, in favor of the Applicant's legal argument.

The Final SEIR also lacks specific performance standards for White Sturgeon, which it finds to be potentially significantly impacted. FSEIR 3.2-36. As a result, "the Revised Project would be required to implement a new Mitigation Measure BIO-9a (as distinct from MM BIO-9a in the 2012 EIR) to reduce the potential impact to White Sturgeon to less than significant." *Id*. No further performance standards are provided, and the only rationale given is that another species, salmonids, were found not to be taken by aggregate mining activities by CDFW, as noted above. *Id*. Emergency regulations for the protection of White Sturgeon as a listed species under the California Endangered Species Act have been on the books for nearly two years, yet the Final SEIR neither adequately evaluates sand mining impacts to this species nor sets specific performance standards for protecting it from those potentially significant impacts.

[13] *See, e.g.*, *Sundstrom v. County of Mendocino* (1988) 202 Cal.App.3d 296; *Madera Oversight Coalition, Inc. v. County of Madera* (2011) 199 Cal.App.4th 48, 49 (mitigation measure proposed to verify that certain archaeological sites are historical resources for CEQA purposes was an unlawful deferral of environmental analysis: "post certification verification procedure [is contrary to law and] allows for an environmental decision to be made outside an arena where public officials are accountable.").

[14] *Id*.

[15] *Sierra Club v. Cty. of Kern,* No. F071133, 2018 Cal. App. Unpub. LEXIS 4685 (July 10, 2018).

e. Worsening conservation status heightens—not reduces—concern

The San Francisco Bay–Delta distinct population segment of Longfin Smelt was listed as endangered under the federal ESA in 2024.[16] In that context, a downgrade from Class I to Class II requires especially clear record support explaining why impact severity is now deemed lower than what the Draft SEIR and prior EIR record indicate.

While the Final SEIR references project restrictions during Chinook Salmon migration, it still fails to analyze such restrictions for other special status fish species including Delta Smelt, Longfin Smelt, Green Sturgeon, and White Sturgeon. White Sturgeon are particularly vulnerable to projects like sand mining because they live almost their entire lives in the Bay, on the bottom. Boat strikes and dredging are potential sources of impact to both Green Sturgeon and White Sturgeon.[17] In the period since the 2012 EIR, Longfin Smelt and White Sturgeon have both been tagged for special protections because their populations have been dropping so precipitously. Worsening population numbers for these species require a closer look at mitigation measures and impacts, not a step back.

The U.S. Fish and Wildlife Service listed Longfin Smelt as a federally endangered species in 2024, based in part on the inadequacy of existing regulatory protections (including those resulting from protection under the California Endangered Species Act).[18] As USFWS explained, the species was at a 20% likelihood of extinction by 2035 and a 50% likelihood of extinction within 30 years.[19] The FSEIR does not account for this substantial increased risk to the species, nor the role of the Project in causing harm to Longfin Smelt. These risks are worse now than they were when the ITP was adopted in 2014 and when the initial EIR was adopted in 2012.

The California Department of Fish and Wildlife (CDFW) found in 2024 that California White Sturgeon, the name CDFW used for California Endangered Species Act listing purposes, were in trouble.[20] In 2025 CDFW released a study showing new abundance estimates, which are much lower than previous abundance estimates and represented a big decline from where the population was when it

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[16] U.S. Fish and Wildlife Service. (2024, July 30). *Endangered and threatened wildlife and plants; Endangered species status for the San Francisco Bay–Delta distinct population segment of the longfin smelt. Federal Register, 89*(146), 61029–61049 (effective Aug. 29, 2024). Retrieved January 30, 2026, from https://www.govinfo.gov/content/pkg/FR-2024-07-30/pdf/2024-16380.pdf

[17] In 2009, the San Francisco Estuary Institute prepared a study for the U.S. Army Corps of Engineers regarding San Francisco Estuary dredging impacts on green sturgeon (Stanford et al. 2009). Direct impacts include entrainment from hydraulic dredging, exposure to contaminated sediments, water quality impacts via sediment resuspension and sedimentation, disturbance from underwater noise, and changes to habitat (e.g., bed leveling). Indirect impacts include modifications to prey base, increased occurrence of ship propeller strikes, and predation by invasive species. Impacts to Green Sturgeon are likely amplified for California White Sturgeon, because California White Sturgeon spend most of their lives in the San Francisco Estuary, whereas Green Sturgeon migrate through the estuary more quickly.

[18] *See* USFWS 2024a, 89 FR 61029 (published rule listing Longfin Smelt as endangered) (attached).

[19] *See* USFWS 2024b (Longfin Smelt Species Status Assessment) (attached).

[20] *See* CDFW 2024, Petition Evaluation for White Sturgeon (Acipenser Transmontanus), Report to the Fish and Game Commission (March 2024) at 16 (attached) ("the Department has determined that the Petition and other relevant information provide sufficient scientific information to indicate that the petitioned action to list the White Sturgeon as threatened may be warranted. Therefore, the Department recommends the Commission accept the Petition for further consideration under CESA").

was first petitioned for listing under the California Endangered Species Act.[21] So risks for White Sturgeon are also worse now than when the initial EIR was adopted in 2012.

The same is true for Delta Smelt, which are subject to continuing impacts that are expected to produce long-term declines (i.e., they are non-viable) according to the Final Environmental Impact Statement (FEIS) analysis for the recently adopted Central Valley Project Long-Term Operations Plan (and there is some indication that the impacts under 2024 operations are greater than the impacts of the previous, 2019, plan).[22] Figure F.4-9 of that document shows that Delta Smelt population growth will remain below replacement (<1) on average under every variant of the new operations (which have since been adopted).[23] The lower panel of that figure shows that new operations are actually worse than previous operations for Delta Smelt.[24] This worsening condition must be taken into account in any new environmental analysis, but has not been adequately considered in the Final SEIR.

According to NMFS, Green Sturgeon also continue to be at risk of extinction, and activities like sand mining post a threat to the species.[25] Chapman et al. (2019), for instance, found that 81% of detected green sturgeon were observed in at least one dredged area or dredged material placement site.[26] Winter-run Chinook Salmon and spring-run Chinook Salmon are also facing worse risks now than in 2012. Again, risks are worse now than when the initial EIR was adopted in 2012 and are unaccounted for in the Final SEIR.

3. The Final SEIR Manipulates the Environmental Baseline to Obscure Actual Project Impacts

By increasing production by 60% over historic average mined volumes of sand, according to BCDC,[27] the Project is functionally a major expansion masked as a reduction. This is made possible through manipulation of the environmental baseline.

a. Legal Requirements for Baseline Selection

CEQA requires that environmental impacts be evaluated against a description of the existing physical environment that serves as the baseline for significance determinations. The CEQA Guidelines provide that the environmental setting "will normally constitute the baseline physical conditions" and further specify that an "existing conditions baseline shall not include hypothetical conditions… that might be allowed, but have never actually occurred, under existing permits or plans."[28] California

[21] See CDFW 2025, 2024 White Sturgeon Population Estimate (attached).
[22] *See* Reclamation 2024, Delta Smelt Life Cycle Model analysis from Reclamation's 2024 FEIS of CVP long term operations (attached).
[23] *Id*. at page F.4-19.
[24] *Id*.
[25] *See* NMFS 2022 Southern Distinct Population Segment of North American Green Sturgeon (Acipenser medirostris) 5-Year Review: Summary and Evaluation at 16 (attached) ("No changes to the species status or threats are evident since the last review based on the reviewed information on abundance and demographic trends").
[26] *Id*. at 12.
[27] *See* BCDC, Comment Letter on Draft SEIR (Sept. 29, 2025) at p. 3, *infra* § 3(b).
[28] Cal. Code Regs., tit. 14, § 15125(a)(1)–(3).

courts have similarly held that a baseline should reflect actual physical conditions rather than inflated "allowable" scenarios that minimize apparent impacts.[29]

The Final SEIR asserts that it need not evaluate the existing physical environmental setting in its baseline because it is only modifying a previously approved project. FSEIR II-7 (Master Response C). By assuming that "the originally approved project" exists, the FSEIR departs into the realm of fantasy to evaluate a project that never actually existed; this leaves the underlying purpose of CEQA unfulfilled. According to the California Supreme Court, "[p]rior operating permits do not in themselves establish a baseline for CEQA review of a new project."[30] As the Court explained,

> A long line of Court of Appeal decisions holds, in similar terms, that the impacts of a proposed project are ordinarily to be compared to the actual environmental conditions existing at the time of CEQA analysis, rather than to allowable conditions defined by a plan or regulatory framework. This line of authority includes cases where a plan or regulation allowed for greater development or more intense activity than had so far actually occurred, as well as cases where actual development or activity had, by the time CEQA analysis was begun, already exceeded that allowed under the existing regulations. In each of these decisions, the appellate court concluded the baseline for CEQA analysis must be the "existing physical conditions in the affected area" (*Environmental Planning & Information Council v. County of El Dorado, supra*, 131 Cal. App. 3d at p. 354), that is, the "'real conditions on the ground'" (*Save Our Peninsula Committee v. Monterey County Bd. of Supervisors, supra*, 87 Cal.App.4th at p. 121; s*ee City of Carmel-by-the-Sea v. Board of Supervisors, supra*, 183 Cal. App. 3d at p. 246), rather than the level of development or activity that could or should have been present according to a plan or regulation.[31]

The cases cited by the California Supreme Court are as follows:

(i) *Environmental Planning & Information Council v. County of El Dorado* (1982) 131 Cal. App. 3d 350, 354, 357–358 (effects of a proposed area plan for land development must be compared to the existing physical conditions in the area, rather than to development permitted under the county's general plan);
(ii) *City of Carmel-by-the-Sea v. Board of Supervisors* (1986) 183 Cal. App. 3d 229, 246–247 (effects of rezoning must be compared to the existing physical environment, rather than to development allowed under a prior land use plan);
(iii) *County of Amador v. El Dorado County Water Agency* (1999) 76 Cal.App.4th 931, 955 (baseline for water diversion project was actually existing stream flows, not minimum stream flows set by federal license);
(iv) *Save Our Peninsula Committee v. Monterey County Bd. of Supervisors* (2001) 87 Cal.App.4th 99, 121 (water use baseline for analysis of proposed land development

[29] *Communities for a Better Env't v. S. Coast Air Quality Mgmt. Dist.*, 48 Cal. 4th 310 (2010); *Neighbors for Smart Rail v. Exposition Metro Line Constr. Auth.*, 57 Cal. 4th 439 (2013).

[30] *Communities for a Better Environment v. South Coast Air Quality Management Dist.* (2010) 48 Cal.4th 310, 320.

[31] *Id*. at 320-321 (footnotes omitted).

was actual use without the project, not what the applicant was entitled to use for irrigation);

(v) *San Joaquin Raptor Rescue Center v. County of Merced* (2007) 149 Cal.App.4th 645, 658 (baseline for proposed expansion of a mining operation must be the "realized physical conditions on the ground, as opposed to merely hypothetical conditions allowable under existing plans");

(vi) *Woodward Park Homeowners Assn., Inc. v. City of Fresno* (2007) 150 Cal.App.4th 683, 693, 706–710 (effects of a large office and shopping center development must be compared to the current undeveloped condition of the property, rather than to an office park that could be developed under existing zoning).[32]

State Lands' reliance on a Court of Appeals case from 1999, over 10 years before the California Supreme Court decided *Communities for a Better Environment v. South Coast Air Quality Management Dist.,* in 2010 is not reasonable. FSEIR II-8 (Master Response C, citing *Fairview Neighbors v. County of Ventura* (1999) 70 Cal.App.4th 238). While it may be reasonable to rely on permitted levels that closely approximate actual operations (as was the case in the *Fairview Neighbors* court's evaluation of fluctuating truck trips from a mining operation with a difference of less than 5% between the busiest times at the mine and the permitted number),[33] there is no close approximation here. The Final SEIR must use a baseline that reflects actual physical conditions; because it does not, it is in violation of CEQA.

b. The Final SEIR Uses Authorized Volumes Rather Than Actual Extraction Volumes

The Final SEIR baseline relies on authorized extraction volumes rather than actual mining volumes that have occurred. The San Francisco Bay Conservation and Development Commission ("BCDC") explained in its Draft SEIR comments that the State's analysis relies on conditions that do not exist, because actual extraction since 2012 has been substantially lower than fully authorized annual volumes.[34] In its comments, BCDC reported that the maximum annual volume mined since 2012 was approximately 976,000 cubic yards, and that average annual mining volume across lease areas was approximately 700,000 cubic yards.[35]

This baseline selection affects impact characterization. Using fully authorized lease volumes as the baseline makes the Revised Project appear to reduce impacts (e.g., a claimed "~14% reduction" relative to fully authorized annual volume), while obscuring that—measured against actual historic operations—the Revised Project represents a substantial increase in annual mining activity.[36] BCDC

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[32] *Id*. at 321 fn 6.

[33] "The production previously permitted under CUP-1328 is the daily average of 810 truck trips. Indeed, in 1989, peak operation of the mine under CUP-1328 generated 837 daily truck trips." *Fairview Neighbors v. Cty. of Ventura*, 70 Cal. App. 4th 238, 243 (1999) (further noting that, because of the difference, "a supplemental EIR, a narrowed EIR under the concept of 'tiering' or a partial exemption may have been reasonable" in these circumstances).

[34] Comment Letter A4, FSEIR II-62 to II-71 (BCDC, Comment Letter on Draft SEIR (Sept. 29, 2025) at p. 3).

[35] *Id*. at II-64.

[36] Id.

estimated that comparing the Revised Project's annual volume to the historic average indicates an increase of approximately 60%.[37] The difference between the asserted 14% reduction and the actual 60% increase must be disclosed and analyzed, not hidden.

c. The Inflated Baseline Frustrates CEQA's Informational Purpose

By establishing a baseline that exceeds actual operations, the Final SEIR presents a comparison that impairs informed public review. CEQA's baseline requirement is intended to ensure decision-makers and the public can understand the magnitude of change a project would cause relative to existing conditions.[38] The baseline approach identified by BCDC obscures the actual scale of change associated with higher annual extraction rates and the associated biological and physical disturbance pathways evaluated in the SEIR.

4. The Final SEIR Fails to Adequately Respond to Substantial New Scientific Evidence Demonstrating Significant Impacts

Since certification of the 2012 EIR, a substantial body of new scientific work—commissioned in part through the BCDC/State Coastal Conservancy sand studies process and reflected in the Independent Science Panel's ("ISP") synthesis and updated technical reports—has materially changed the understanding of San Francisco Bay sand supply, transport connectivity, and the sustainability of mining from existing lease areas. CEQA requires a subsequent EIR when new information of substantial importance shows that the project will have significant effects not discussed in the prior EIR or that such effects will be substantially more severe than previously disclosed.[39]

Sand is a non-renewable resource in San Francisco Bay. The demonstrated non-replenishment of that resource leads to bed lowering in the Bay. In turn, that bed lowering leads to the loss of benthic habitat as well as associated and additional fish, bird, and marine mammal impacts. Moreover, the sediment deficit created by sand mining leads to significant impacts to shoreline resilience and substantial public trust impacts.

a. Independent Science Panel Findings Constitute Substantial New Information

In June 2024, the ISP convened through the BCDC/State Coastal Conservancy sand studies effort issued a synthesis identifying key findings relevant to the physical effects and sustainability of Bay sand mining.[40] The ISP's "Key Observations" include (among others) that:

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[37] *Id.* (noting baseline exceeds actual operations by approximately 1.34 million cubic yards; Revised Project implies approximately 1.05 million cubic yard increase over approximately 700,000 cubic yard average).

[38] *Id.* (recommending revision to analyze impacts of increased mining activity using actual average annual mining volumes as baseline).

[39] CEQA Guidelines, Subsequent EIR Standard. 14 Cal. Code Regs. § 15162(a)(3). Also see Pub. Res. Code § 21166 (limits on subsequent review absent specified triggers, including new information).

[40] Independent Science Panel (ISP) Synthesis. Battalio, B., et al. (2024, June 3). San Francisco Bay Sand Budget, Transport, Provenance, and Bathymetric Change Studies and Potential Physical Effects of Sand Mining Activities: Independent Science Panel Findings Report – Best Available Science. [hereinafter "ISP Synthesis"]. Available at: https://www.bcdc.ca.gov/wp-content/uploads/sites/354/2024/01/2024-06-20-SF-Bay-Sand-Budget-Transport-Provenance-and-Mining-Impacts.pdf

I. The volume of mined sand is significant relative to the Bay sand budget and is mined faster than it is replenished, making sand effectively non-renewable over the long term;
II. Suisun Bay sand is not replenished, and mining-related bed depressions can persist where transport is negligible; and
III. Central Bay sand is largely "relic" (deposited on the order of ~20,000 to 6,000 years ago) and Delta-derived sands are no longer a significant source to the Bay and ocean under modern conditions as previously assumed.

These are not minor refinements—they are foundational findings about resource renewability, transport connectivity, and persistence of mining effects.

b. New Sand Budget Results Undercut Prior Renewable Supply Assumptions

A recent sand budget analysis determined that during 2001–2020, total sand inflow to the Bay averaged approximately 0.45 Mt/yr, with approximately 81% derived from local Bay Area tributaries.[41] The study found net average Delta sand supply to be zero.[42] The same analysis identified sand mining as the largest sand loss from the Bay, averaging approximately 1.17 Mt/yr and comprising approximately 75% of total sand outflows.[43] These findings indicate that authorized extraction removes sand at a rate approximately 2.6 times greater than total annual sand inflow (1.17 Mt/yr ÷ 0.45 Mt/yr = 2.6).[44] This mass balance is inconsistent with assumptions that authorized extraction rates are sustained by modern replenishment processes.

The July 2025 report further indicates that Bay–ocean sand exchange averaged near zero (best estimate approximately 0.034 Mt/yr, within a range of uncertainty that includes both inflow and outflow). This finding does not support a basis for concluding that mining impacts are not significant, given that mining rates vastly outstrip the rate of sediment arriving and depositing into San Francsico Bay.

c. Fingerprinting Evidence Supports Disconnected Sediment Sinks and Limited Modern Delta Contribution

The University of Texas-led fingerprinting study tested what it characterized as the "prevailing hypothesis" of a continuous sand transport pathway from the Sacramento–San Joaquin Delta through the Bay to the outer coast. [45] The provenance analysis instead documented contrasts supporting disconnected sediment sinks, termed depocenters. Specifically, Central Bay and outer coast sands were found to be indistinguishable, while Suisun Bay and San Pablo Bay showed limited Delta sand contribution. These conclusions are consistent with the July 2025 sand budget finding of

[41] McKee, L.J., et al. (2025, July). Sand Budget and Sand Transport in San Francisco Bay (SFEI Contribution #1125). Available at: https://www.sfei.org/documents/sand-budget-and-sand-transport-san-francisco-bay

[42] *Id.*

[43] *Id.* at Executive Summary.

[44] *Id.*

[45] Malkowski, M.A., Sickmann, Z.T., Jaffe, B., Fregoso, T., & Mohrig, D. Fingerprinting sand and its transport history through San Francisco Bay: Implications for sand mining and its environmental effects. Report available at: https://www.sfei.org/sites/default/files/biblio/2025-10/UT%20Statrigraphy_Report_FINAL_clean_0.pdf

zero net Delta sand supply. [46] The combined evidence does not support earlier assumptions regarding sediment connectivity and transport pathways through the Bay system.

d. Replenishment and Persistence of Mining-Related Depressions

Independent analyses of bathymetric change and replenishment patterns in and around mined areas indicate that recovery is variable and incomplete over the analysis periods examined. The Deltares "ring analysis" of a Central Bay lease area documented partial recovery ranging from approximately 11% to 28%, depending on the time period and ring location analyzed.[47] These results indicate that mined areas do not fully restore to pre-mining bathymetry within the timeframes evaluated.[48]

The Independent Science Panel synthesis noted that in areas with negligible sediment transport—specifically identifying Suisun Bay—mining-related depressions can persist and the bed can experience progressive lowering.[49]

e. Subsequent EIR Analysis Is Required

CEQA Guidelines section 15162 requires a subsequent EIR where new information of substantial importance—unknown and not reasonably knowable at the time of the prior EIR—shows new significant effects or substantially more severe effects than previously disclosed.[50] Here, the ISP synthesis and the superseding July 2025 sand budget, together with fingerprinting and replenishment analyses, provide substantial new information indicating that:

(i) Mining is significant relative to the Bay sand budget and is not balanced by modern replenishment;
(ii) Suisun Bay sand is not replenished and mining effects can persist;
(iii) Delta-derived sands are not a significant modern source, and a simple “conveyor belt” model is not supported as a general premise; and
(iv) Replenishment in mined areas is incomplete within relevant timeframes.

A supplemental document that primarily responds to comments, without incorporating the implications of this new evidence into a defensible subsequent-impact analysis, fails CEQA’s requirements when faced with substantial new scientific information of this magnitude.

[46] McKee, L.J., et al. (2025, July). Sand Budget and Sand Transport in San Francisco Bay (SFEI Contribution #1125).

[47] Elias, E., & Roelvink, F. (2023). Understanding Impacts of Bay Sand Mining on Sand Supply and Transport in San Francisco Bay and Outer Coast—Task 2.2: Lease Area Ring Analysis and Bedform Dynamics (Deltares Report No. 11206299-000-ZKS-0003).

[48] *Id.*

[49] ISP Synthesis, Executive Summary.

[50] CEQA Guidelines, Subsequent EIR Standard. 14 Cal. Code Regs. § 15162(a)(3). Also see Pub. Res. Code § 21166 (limits on subsequent review absent specified triggers, including new information).

5. The Final SEIR Fails to Adequately Analyze Public Trust Obligations

The public trust doctrine imposes on the State a continuing duty to protect navigable waters and the lands beneath them for the benefit of the public, including ecological values such as fisheries and habitat. In *National Audubon Society v. Superior Court*, the California Supreme Court explained that the State has "an affirmative duty to take the public trust into account" in planning and allocating resources and to protect trust interests whenever feasible.[51]

As the trustee for sovereign tidelands and submerged lands, the California State Lands Commission must make a reasoned determination that authorizing commercial extraction from trust lands is consistent with public trust purposes and does not improperly subordinate trust resources to private use. *San Francisco Baykeeper v. California State Lands Commission* confirms that this obligation is independent of (and in addition to) CEQA compliance; where the Commission authorizes sand mining on sovereign lands, it must separately address whether that authorization is a proper use of trust property.[52]

The Final SEIR's Master Response G, Public Trust Considerations, is circular in its reasoning: it claims compliance because it "considered" the issues, without actually weighing the loss of the resource against the private profit. Contrary to this, the public trust doctrine imposes "an affirmative duty to take the public trust into account . . . and to protect public trust uses whenever feasible."[53] The Commission thus has an ongoing duty to consider the impacts that sand mining has on public trust resources. (*Id.* at 447.) Because of this ongoing duty, the public trust analysis provided by State Lands is nothing more than a *post hoc* rationalization that cannot provide support for the Commission's conclusion.[54]

The growing body of scientific evidence linking sand mining to coastal erosion makes clear that sand mining is a detriment on public trust resources, even if some uncertainties remain. The Commission is legally obligated to mitigate harm to trust resources "whenever feasible." *National Audubon*, *supra,* 33 Cal.3d at 446.

The court in the earlier Baykeeper cases found that,

> although commercial sand mining is not categorically permissible as a public trust use, the SLC may authorize private uses of trust property that do not impair the trust. (*Baykeeper I, supra*, 242 Cal.App.4th at pp. 235–238.) Consistent with this common law rule, section 6900 codifies the SLC's authority to grant leases for the extraction of minerals other than oil and gas from trust lands 'when it appears to be in the public interest' and when 'it appears that the execution of such leases and the

[51] *Nat'l Audubon Soc'y v. Superior Ct.*, 33 Cal. 3d 419, 446 (1983).

[52] *San Francisco Baykeeper, Inc. v. Cal. State Lands Comm'n*, 242 Cal. App. 4th 202, 227–29 (Cal. Ct. App. 2015).

[53] *National Audubon, supra*, 33 Cal.3d at 446.

[54] *See No Oil, Inc. v. Los Angeles* (1974) 13 Cal. 3d 68, 81, citing *Citizens to Preserve Overton Park v. Volpe* (1971) 401 U.S. 402, 420 and *Environmental Defense Fund, Inc. v. Coastside County Water Dist.* (1972) 27 Cal. App. 3d 695, 706.

> operations thereunder will not interfere with the trusts upon which such lands are held or substantially impair the public rights to navigation and fishing.' (§ 6900.)"[55]

The record before the Commission now shows that sand mining activities impair public trust uses by substantially depleting the sand resource and substantially interfering with sand transport and coastal morphology at the San Francisco bar and Ocean Beach. There is no "venerable authority" (FSEIR II-25, Response to Comments) that gives the Commission discretion to ignore public trust impacts and favor a non-trust use like sand mining over traditional public trust uses.

This is not a balancing exercise, though the Response to Comments implies as much when it states it "has weighed public benefits against environmental costs." FSEIR II-25, Responses to Comments. Paying mere lip service to taking the public trust into account by disregarding the weight of scientific evidence pointing toward irreversible impacts cannot satisfy the Commissions public trust obligations. What the Commission attempts to create, here, is a sort of reverse precautionary principle to hold that if there is any sliver of uncertainty it allows the Commission complete discretion to approve non-trust activities.

There can be no doubt: sand mining in the Bay removes sand, which is a public trust resource right in the middle of wildlife migration corridors, in a heavily used recreational area, near a staggering amount of public infrastructure and beaches. Removing sand from one location necessarily impacts the sand and sediment around it. This conclusion is more than adequately supported by the weight of evidence presented by commenters and in the record of impacts to public trust uses. The public trust doctrine does not require a quantified impairment, but rather the Commission must ensure that non-trust uses can be approved "without detriment" to public trust resources and uses.[56]

The Commission repeatedly fails to address the portions of the record that specifically address sand mining impacts. Instead, it plucks quotations from the studies and the Science Panel that support its conclusion that coastal erosion has multiple, long-term factors, while conveniently ignoring the myriad conclusions in those same documents regarding the direct impact of sand mining. That other factors cause similar impacts does not eliminate the impacts caused by the direct effect of sand mining. It is clear from the studies and the other new information presented in the record that sand mining impairs and has a detrimental effect on the San Francisco Bar and Ocean Beach. The cherry-picked responses result in mischaracterization of the studies and is not substantial evidence in support of the Commission's conclusions, as is required by law.[57]

The Commission's public trust determination must be evaluated in light of the *whole* record.[58] The Commission does have discretion to choose between conflicting expert opinions, but this discretion does not give the Commission the right to consider only the portions of studies that support its

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[55] *S.F. Baykeeper, Inc. v. State Lands Com.*, 29 Cal. App. 5th 562, 580 (2018).
[56] *C. R. Co. v. Illinois, 146 U.S. 387, 456 (1892)* ("*Illinois Central*," also stated as "without impairment").
[57] *See San Franciscans Upholding the Downtown Plan v. City & Cty. of S.F.*, 102 Cal. App. 4th 656, 675, 125 Cal. Rptr. 2d 745, 757 (2002) (substantial evidence is not "unsubstituted opinion, or clearly erroneous evidence").
[58] *See Sierra Club v. California Coastal Comm'n* (1993) 19 Cal.App.4th 547, 557.

conclusions and ignore the rest.[59] Nor is there a meaningful conflict—the only fair reading of the evidence is that sand mining has substantial negative impacts on the public trust. The Commission must eliminate those to the extent feasible.

a. Substantial New Information and Changed Conditions Require a Renewed Trustee Evaluation

Here, the record contains substantial post-2012 scientific evidence and agency concerns indicating that continued extraction may impair trust resources and trust uses, including fisheries, habitat, and shoreline resilience. Where the evidentiary record materially evolves, the Commission's trustee duty requires an updated, transparent balancing of trust values against the proposed private use, supported by substantial evidence and responsive to the best available science.[60]

b. The Final SEIR's Responses Do Not Substitute for an Adequate Public Trust Determination

To the extent the Final SEIR treats public trust issues as outside the scope of the environmental review record—or collapses trustee balancing into a general statement that economic impacts are not "CEQA impacts"—that framing misses the point. While CEQA is primarily environmental and does not require the Commission to treat private economic impacts as environmental effects, the Commission's public trust duty is a separate constitutional and common law obligation that requires the trustee to evaluate whether the proposed extraction level and operational constraints are compatible with trust purposes and trust resource protection.[61]

Importantly, BCDC specifically flagged this gap: the BCDC letter states that Section 5.11 of the Draft SEIR should discuss the Commission's public trust responsibilities and quotes the Bay Plan's public trust policy stating that when the Commission takes actions on trust lands, it should ensure the action is consistent with the public trust needs of the area.[62] A response-to-comments in a CEQA document does not satisfy the trustee's obligation to make and disclose a public trust determination in a manner that is reviewable and grounded in the administrative record.

c. Major Public Investments in Shoreline Protection Reinforce the Need for Trustee Balancing

Public trust analysis must also account for contemporary public needs and the State's evolving investments in protecting shoreline infrastructure and public recreational resources. The California Coastal Commission record for the Ocean Beach project reflects an estimated total project cost of approximately $155 million, underscoring the scale of public investment being made to protect

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[59] *See, e.g.*, *Berkeley Keep Jets Over the Bay Comm. v. Bd. of Port Comm'rs* (2001) 91 Cal.App.4th 1344, 1367 ("*Berkeley Keep Jets*") (decision is not based on substantial evidence when agency does not disclose new or conflicting data or opinions or relies on outdated scientific information).

[60] *San Francisco Baykeeper, Inc. v. Cal. State Lands Comm'n*, 29 Cal. App. 5th 562, 589–91 (Cal. Ct. App. 2018).

[61] *Nat'l Audubon Soc'y v. Superior Ct.*, 33 Cal. 3d 419, 446 (1983); *San Francisco Baykeeper, Inc. v. Cal. State Lands Comm'n*, 242 Cal. App. 4th 202, 227–29 (Cal. Ct. App. 2015); *San Francisco Baykeeper, Inc. v. Cal. State Lands Comm'n*, 29 Cal. App. 5th 562, 589–91 (Cal. Ct. App. 2018).

[62] FSEIR II-62, BCDC Comment Letter on Draft SEIR (Sept. 29, 2025).

critical coastal infrastructure and manage shoreline risk.[63] In a sediment-limited system—where best available science indicates sand inputs are small relative to extraction—continued commercial removal from sovereign lands raises an unavoidable trustee question: whether and how the authorized private extraction is consistent with the State's duty to safeguard public trust resources and public shoreline resilience.[64]

6. The Final SEIR's Treatment of Marine Mammal Impacts Fails to Account for Changed Circumstances

a. Increased Marine Mammal Presence in Central Bay

The Final SEIR acknowledges that marine mammal presence and use of Central Bay has changed substantially since 2012.[65] Documented changes include the year-round return of harbor porpoises (*Phocoena phocoena*) to Central Bay after decades of absence, and increased observations of northern elephant seals (*Mirounga angustirostris*), including young-of-the-year haul-outs and strandings.[66] The Final SEIR further describes increased bottlenose dolphin sightings in warm years and identifies gray whales as increasingly common Bay visitors, including during the recent Eastern North Pacific gray whale Unusual Mortality Event.[67] These documented changes in distribution and Bay use indicate that the biological setting evaluated in 2012 has changed.

The Final SEIR's impact determinations identify "protected marine mammals"—including gray whales, humpback whales, harbor seals, and harbor porpoises—as species potentially displaced or otherwise affected by Project operations and Project-related habitat disturbance.[68] BCDC similarly recommended that the lead agency evaluate humpback whale presence and use of San Francisco Bay as an important and evolving resource issue in its environmental review.[69] Peer-reviewed research shows increased humpback whale entry into San Francisco Bay in recent years and describes the Bay as a novel foraging habitat for whales that historically were not frequent Bay users.[70]

Because Endangered Species Act protections for humpback whales are specific to distinct population segments ("DPS"), the FSEIR should have evaluated how increased whale presence, behavioral use (including foraging), and vessel interactions affect risk under federal and state protected-species frameworks, rather than relying on categorical designations.[71]

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[63] Cal. Coastal Comm'n, Staff Report, CDP Application No. 2-21-0912 (SFPUC Ocean Beach Armoring) 2 (Oct. 31, 2024) (noting City's estimate that construction costs will amount to approximately $175 million).

[64] FSEIR II-62, BCDC Comment Letter on Draft SEIR (Sept. 29, 2025).

[65] Final SEIR § 3.2.2.3, at 3.2-6.

[66] *Id.*

[67] *Id.*

[68] *Id.* at § 3.2.5, Impact BIO-1 & Impact BIO-3, at 3.2-18 to 3.2-19.

[69] BCDC, Comment Letter on Draft SEIR (Sept. 29, 2025).

[70] Markowitz, T., et al. (2024). New Urban Habitat for Endangered Humpback Whales: San Francisco Bay. *Aquatic Conservation: Marine & Freshwater Ecosystems*, 34, e4107.

[71] https://www.federalregister.gov/documents/2016/09/08/2016-21276/endangered-and-threatened-species-identification-of-14-distinct-population-segments-of-the-humpback.

b. The FSEIR's noise and disturbance analysis remains conclusory given current conditions

The Final SEIR concludes that impacts from increased noise are less than significant, reasoning that noise from dredging is marginally above ambient, detectable over only a small area near operations, and below levels likely to cause physical injury. "[H]ydraulic suction dredge operations can generate underwater noise levels as high as 130 to 140 dB which is lower than the threshold for behavioral disturbance (150 dB) and acute impacts in fishes (183 dB). This may even be lower than ambient sound levels in San Francisco Bay which can range from 120 to 155 dB."[72] That conclusion does not take into account the duration of the noise generated or the combination of sand mining noise disturbances with impacts from ambient noise in San Francisco Bay, which the FSEIR acknowledges is a concern.[73] This lack is in spite of the FSEIR's acknowledgement that "duration of the sound [and] proximity to the source" can be major factors in determining injury.[74] Given the Final SEIR's own recognition that marine mammal use of the Central Bay has increased and diversified since 2012, these impacts should be further analyzed. A less-than-significant determination premised on localized exposure and short-term disturbance warrants a more transparent, current-condition analysis of:

(i) The frequency and duration of operational noise and vessel activity over the authorized season,
(ii) Behavioral disturbance pathways (including displacement from foraging), and
(iii) The reasonable potential for cumulative disturbance where animals repeatedly use the same Central Bay areas contemporaneously with Project operations.

This is particularly important because the Marine Mammal Protection Act defines harassment to include acts that have the potential to disrupt behavioral patterns such as feeding, breeding, and sheltering.[75]

c. The EIR should not rely on future federal capacity to supply missing analysis or protections

Finally, while federal permitting and consultation obligations may apply to aspects of protected species management, CEQA requires that the EIR itself disclose and analyze the Project's potential impacts under existing conditions. The Final SEIR's own discussion of heightened gray whale presence underscores that marine mammals in and near the Project area may be experiencing broader cumulative stressors.[76] Recent reporting regarding NOAA operational constraints and staffing reductions further reinforces that the Commission should not assume future federal capacity

[72] Final SEIR, § 3.2.5.1, Impact BIO-2, at 3.2-22 (noise analysis and rationale).

[73] *Id*. ("Ambient noise in San Francisco Bay from non-project related vessels and other sources is likely more of a concern").

[74] *Id*.

[75] Marine Mammal Protection Act, 16 U.S.C. § 1362(18)(A)(ii).

[76] FSEIR § 3.3.2.3, at 3.2-6 (in 2025 "it was reported that up to nine individuals were observed within San Francisco Bay at one time and that some individuals have remained within San Francisco Bay for up to 20 days").

will substitute for a complete, self-contained CEQA analysis and enforceable mitigation where warranted.[77]

7. The Final SEIR Fails to Adequately Address Cumulative Impacts

a. Legal Standard for Cumulative Impact Analysis

CEQA requires an EIR to evaluate cumulative impacts, defined as "two or more individual effects which, when considered together, are considerable or which compound or increase other environmental impacts."[78] The cumulative analysis must address whether the Project's incremental contribution to a cumulative effect is cumulatively considerable—that is, whether the Project's incremental effects are significant when viewed in connection with the effects of past projects, other current projects, and probable future projects.[79]

When a project's incremental effect is cumulatively considerable, the EIR must provide a meaningful cumulative discussion commensurate with the severity and likelihood of the effect, and must identify reasonable, feasible options for mitigating or avoiding the Project's contribution to any significant cumulative impacts.[80]

b. The Final SEIR Understates the Project's Contribution to a Documented Sediment Deficit

The July 2025 San Francisco Bay sediment budget synthesizes best available data for 2001–2020 and reports that, at the whole-bay scale, total sediment inflow is approximately 2.0 million metric tons per year (Mt/yr) while total sediment outflow is approximately 3.64 Mt/yr, with the difference supplied by losses from the Bay bed (about 1.67 Mt/yr).[81] For sand, the imbalance is more pronounced: sand inflow is approximately 0.45 Mt/yr, while sand outflow is approximately 1.55 Mt/yr, again with the difference supplied by losses from the bed (about 1.11 Mt/yr).[82]

Within this budget framework, sand mining is a dominant outflow term. The sediment budget reports sand mining removals of approximately 1.17 Mt/yr, equivalent to about 32% of total sediment outflows and about 75% of total sand outflows.[83] In a system where bed losses already supply the difference between inflows and outflows, the Project's continuation at substantial extraction volumes necessarily warrants a robust cumulative impact analysis focused on (i) the Project's proportional contribution to sediment and sand outflows, and (ii) the implications of chronic reliance on bed losses to balance the budget.[84]

---

[77] ESA Section 7 Consultations on the West Coast, NOAA Fisheries (last updated Feb. 6, 2025), https://www.fisheries.noaa.gov/west-coast/consultations/esa-section-7-consultations-west-coast.

[78] Cal. Code Regs. tit. 14, § 15355.

[79] Cal. Code Regs. tit. 14, § 15064(h)(1); Cal. Code Regs. tit. 14, § 15065(a)(3).

[80] Cal. Code Regs. tit. 14, § 15130(a)–(b).

[81] McKee, L.J., et al. (2025, July). Sand Budget and Sand Transport in San Francisco Bay (SFEI Contribution #1125).

[82] *Id.* (reporting sand inflow 0.45 Mt/yr; sand outflow 1.55 Mt/yr; losses from bed 1.11 Mt/yr).

[83] *Id.* (reporting sand mining 1.17 Mt/yr; 32% of total sediment outflows; 75% of sand outflows).

[84] *Id.*

BCDC has specifically criticized the Draft SEIR for dismissing the significance of the sand budget findings and for failing to analyze cumulative impacts of continued removal—particularly in combination with other sediment removals such as navigation dredging—over time.[85] These agency concerns underscore the need for the Final SEIR's cumulative analysis to move beyond generalized statements and instead engage directly with the quantitative budget evidence and its cumulative implications for bed lowering, sediment availability, and related ecological stressors.[86]

c. Failure to Integrate Climate-Driven Erosion Stressors into the Cumulative Analysis

The cumulative impacts analysis is further weakened by the FSEIR's failure to integrate climate-driven stressors that increase the significance of sediment availability and bathymetric stability. NASA, for example, synthesized current research indicating that atmospheric rivers are expected to become larger, wetter, and more frequent as the climate warms, with implications for storm-driven coastal erosion and sediment management needs.[87] In parallel, the Ocean Beach Climate Change Adaptation Project reflects the scale of public infrastructure and shoreline protection planning now underway in response to erosion and sea level rise, including design elements that explicitly contemplate ongoing beach nourishment (sand replenishment) as part of long-term coastal management.[88]

The point is not to claim a single, linear, causal pathway from any one lease area to any one shoreline segment. Rather, the sediment budget and agency comments identify an interconnected Bay–ocean sediment system with acknowledged uncertainties but potentially material cumulative implications.[89] In that context, the Final SEIR must provide a clearer cumulative analysis of how continued extraction at the proposed scale affects the Bay's sediment and sand balance under changing climatic conditions that increase the consequences of sediment deficits for shoreline resilience, habitat conditions, and the public's long-term coastal management obligations.

8. Conclusion

The Final SEIR fails to function as the informational document CEQA requires. It arbitrarily reclassifies significant impacts, manipulates the environmental baseline, dismisses substantial new scientific evidence, and inadequately analyzes public trust obligations and cumulative effects.

The post-2012 scientific record demonstrates that:

[85] BCDC, Comment Letter on Draft SEIR (Sept. 29, 2025) (criticizing failure to consider cumulative impacts of sand budget findings and continued removal including navigation dredging).

[86] *Id.*

[87] NASA Jet Propulsion Lab, Study: Atmospheric Rivers Are Getting Larger, Wetter & More Frequent (2025). https://www.jpl.nasa.gov/news/climate-change-may-lead-to-bigger-atmospheric-rivers.

[88] San Francisco Recreation & Park Dep't, Ocean Beach Climate Change Adaptation Project Staff Report 2 (Oct. 19, 2023). Available at: https://sfrecpark.org/DocumentCenter/View/21399/Item-9-Ocean-Beach-OBCCAP-Staff-Report_FINAL-101923.

[89] BCDC, Comment Letter on Draft SEIR (Sept. 29, 2025) (describing Bay–ocean "common pool of sand" and noting uncertainties).

- Sand mining extracts finite geological deposits at rates vastly exceeding the natural replenishment rate;
- Recovery rates in most lease areas are insufficient to restore pre-mining conditions;
- Listed smelt species have declined to record low populations while bottom dwelling Green and White sturgeon are increasingly threatened in San Francisco Bay;
- Marine mammal use of mining areas has increased substantially;
- The Bay operates in chronic sediment deficit, with mining contributing 66% of sand outflows.

These findings constitute substantial new information requiring preparation of a new full EIR, not merely responses to comments in a final supplemental EIR. The Commission cannot approve lease renewals based on this inadequate environmental analysis without violating CEQA's mandate that environmental review informs agency decision-making and public participation.

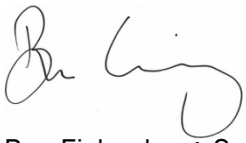
Baykeeper respectfully requests that the Commission:

- Decline to certify the Final SEIR as adequate under CEQA;
- Prepare a subsequent EIR analyzing the Project's impacts in light of post-2012 scientific evidence;
- Conduct a comprehensive public trust analysis evaluating whether continued mining at proposed levels is consistent with the State's fiduciary obligations;
- Consider alternatives that reduce extraction volumes to levels sustainable given natural sand supply rates.

Sincerely,



Ian Wren, Science Consultant
San Francisco Baykeeper



Ben Eichenberg, Senior Staff Attorney
San Francisco Baykeeper

SAN FRANCISCO BAYKEEPER®

Below find a letter and the names of hundreds of San Francisco Baykeeper and Surfrider supporters urging the State Lands Commission to prioritize native wildlife and sea level rise preparedness over the profits of sand mining companies.

Dear Mr. Huitt and Commissioners of the California State Lands Commission:

I'm writing to urge you to follow the science on sand mining in San Francisco Bay.

In its recently released Environmental Impact Report (EIR), the State Lands Commission inadequately analyzed the environmental impacts of sand mining in the Bay, dismissed substantial new scientific evidence, and failed to consider reasonable alternatives that would reduce harm to the Bay's ecosystem.

A growing body of scientific data points to the harmful impact of sand mining on coastal erosion, sea level rise and climate change, and wildlife habitat. Crucially, new studies have shown that much of the Bay's sand is ancient and irreplaceable—and therefore not, as the Commission has claimed, a renewable resource.

As a state agency managing resources on behalf of the public, it is your responsibility to rely on the best available evidence for your recommendations. It is therefore incumbent on the Commission to reject the EIR and revise it with enhanced analysis, strengthened mitigation measures, and meaningful consideration of alternatives that prioritize the protection of San Francisco Bay's natural resources.

I urge you to make more responsible decisions that protect sand as one of San Francisco Bay's most precious nonrenewable resources.

Sincerely,

Lindsay Mugglestone, Berkeley, CA
Elaine Beale, Oakland, CA
Cynthia Cooper, Oakland, CA
Bita Edwards, Woodacre, CA
Dennis Dougherty, Novato, CA
Artemis Schatzkin, Oakland, CA
Marci Spencer, Castro Valley, CA
Lynne Preston, San Francisco, CA
Janet Aguilera, Richmond, CA
Bonnie Thompson, Los Osos, CA
Martin Horwitz, San Francisco, CA
Kathy Silvey, Martinez, CA
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Matthew Reid, Calistoga, CA
Jeanie Scott, San Francisco, CA
Gail Weininger, Oakland, CA
Elina Rubuliak, Alameda, CA
Susan Hampton, El Cerrito, CA
Cathy Holden, Sacramento, CA
Janet Heinle, Santa Monica, CA
Neal Steiner, Los Angeles, CA
Diana Hall, Mountain View, CA
Francesca Suzio, Richmond, CA
Jim DuPont, Alameda, CA
Ann Harvey, Oakland, CA
Christina Ciesla, Simi Valley, CA
Edward Cavasian, Palo Alto, CA
Nora Privitera, Oakland, CA
Carlos Nunez, Reseda, CA
Scott Barlow, Sunnyvale, CA
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Carrie Ousley, Oakland, CA
Jackie Pomies, San Francisco, CA
Carol Schaffer, San Pablo, CA
Leonie Terfort, Mill Valley, CA
Kermit Cuff, Mountain View, CA
Cynthia Alderson, Healdsburg, CA
John Essman, Healdsburg, CA
Betty Kissilove, San Francisco, CA
Jay Rice, Novato, CA
Rachael Denny, Bradley, CA
Joyce Pennell, San Mateo, CA
Jan Buckwald, Oakland, CA
Terri Decker, Redding, CA
Sylvia Seymour, Grass Valley, CA

Mary Spicer, Alameda, CA
Tom Kunhardt, Oakland, CA
Xiang Ding, Oakley, CA
Judy MacLean, Berkeley, CA
Diana Kliche, Long Beach, CA
M. K. Russell, Mill Valley, CA
Sally Marone, Livermore, CA
Maggie Hughes, Berkeley, CA
Jennifer Russell, Walnut Creek, CA
John Everett, Grass Valley, CA
Ana Herold, Pacifica, CA
Barbara Berman, San Francisco, CA
Christina Babst, West Hollywood, CA
Rick DeGolia, Atherton, CA
Yazmin Gonzalez, Bellflower, CA
Maureen Lahiff, Oakland, CA
Judith Anderson, Long Beach, CA
J. Yudell, Santa Monica, CA
Martin Evans, Hercules, CA
Bob Skinner, Novato, CA
Charlene Woodcock, Berkeley, CA
David Zeff, Corte Madera, CA
Valerie Fannin, Chico, CA
Janet Rhodes, Temecula, CA
Deborah Santone, Pleasant Hill, CA
Sylvia Cardella, Hydesville, CA
Anthony Severo, San Francisco, CA
Margot Otway, Menlo Park, CA
Thalia Lubin, Redwood City, CA
Kirsten Holmquist, Sunnyvale, CA
Allen Aronson, Wilmington, CA
Mark Barath, Sacramento, CA
Linda Vanzant, Berkeley, CA
Fred Rinne, San Francisco, CA
Jim Perry, Santa Rosa, CA
Jana Mariposa Niernberger Muhar, Santa Rosa, CA
Tim Barrington, San Jose, CA
Michael Eichenholtz, Richmond, CA
P. Perry, Hidden Valley Lake, CA

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Ruth Burman, San Carlos, CA
Pamela Llewellyn, Berkeley, CA
JL Angell, Rescue, CA
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Andre Thomas, Los Angeles, CA
Rich Moser, Santa Barbara, CA
Brandon Luce, Albany, CA
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Jamie Le, Alameda, CA
Meaveen Connor, Santa Rosa, CA
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Karen Haralson, Felton, CA

Kristin Womack, San Anselmo, CA
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Vanessa Jensen, Berkeley, CA
Pat Brooks, Berkeley, CA
Patricia Roca, El Sobrante, CA
Susanne Herting, Oakland, CA
Kim Russo, San Francisco, CA
Jamie Green, Ventura, CA
Glen Deardorff, Castro Valley, CA
Hillary Ostrow, Encino, CA
Christa Laib, Berkeley, CA
Therese DeBing, Pacific Grove, CA
Blaine Burgstrom, San Jose, CA
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Elizabeth Johnson, Albany, CA
David Coleman, Cobb, CA
Adrianne Micco, Vacaville, CA
Jan Beyer, Stockton, CA
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Dena Bergstrom, Greenbrae, CA
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Sheila Malone, Las Vegas, CA
Elaine Woodriff, Petaluma, CA
Cindy Beckley, Oakland, CA
Martin Marcus, San Diego, CA
Summer Lee, Pacifica, CA
Diann Rose, San Francisco, CA
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Barbara Benzwi, Oakland, CA
Jeff Barnard, Santa Rosa, CA
Susan Von Schmacht, Watsonville, CA
Kevin Jensen, Novato, CA
Sharon Hagen, Pacifica, CA
Holly Hall, Temecula, CA

Allan Campbell, San Jose, CA
Singih Tan, San Jose, CA
Edith Ogella, Santa Barbara, CA
Craig Cook, Santa Rosa, CA
Abbot Foote, Albany, CA
Abbie Bernstein, West Hollywood, CA
Brooke Prather, Kelseyville, CA
Carol Cook, San Mateo, CA
Peter Lee, San Francisco, CA
Denise Vandermeer, Los Angeles, CA
Thomas Zachary, La Crescenta, CA
Russell Stone, San Jose, CA
Stacie Wolny, Campbell, CA
Jim Lieberman, Annapolis, CA
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Kathryn Hyde, San Francisco, CA
Gia Granucci, Healdsburg, CA
Alexa McMahan, Huntington Beach, CA
Kevin Rath, Oakland, CA
John Walton, Gualala, CA
David Burtis, Calistoga, CA
Alison Hill, Lafayette, CA
M. Steere, Petaluma, CA
Bernard Hochendoner, Patterson, CA
Samantha Meza, American Canyon, CA
Mary Vedovi, Hidden Valley Lake, CA
Randy Tom, San Ramon, CA
Judy Schultz, San Francisco, CA
Margaret Mischner, San Francisco, CA
Jonathan Chu, Fremont, CA
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Zora Hocking, Santa Rosa, CA
Rachel Loui, Mountain View, CA
Martha Booz, El Sobrante, CA
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Kathleen Smith, San Jose, CA

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Erin Smith, Monterey, CA
Jerry Lee, Tuscaloosa, AL
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Gary Goetz, Pacific Grove, CA
Judith Kirk, Redwood city, CA
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John Oda, San Francisco, CA
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Jodi Durst, Oakland, CA
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Janet Drew, Santa Rosa, CA
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Erica Stanojevic, Santa Cruz, CA
Roberta Newman, Mill Valley, CA
Candice Schott, Berkeley, CA
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D.G. Sifuentes, Mammoth Lakes, CA
Michele Tusinac, San Francisco, CA
Jan Jones, El Cerrito, CA
Nic Wellington, Kensington, CA
Sue Adler, Emeryville, CA
Flor Montoya-Garcia, Oakland, CA

Leana Zang-Rosetti, Oakland, CA
Glenna Dowling, San Francisco, CA
Evelyn Ibarra, San Leandro, CA
Phyllis Senter, Santa Rosa, CA
Amber Eby, Modesto, CA
Rachel Zawilski, San Mateo, CA
Caephren McKenna, Oakland, CA
Matthew Fladeland, San Francisco, CA
Jessica Cook, Emeryville, CA
Brendan Sheehan, San Francisco, CA
Cynthia Johanson, San Francisco, CA
Richard Finn, South San Francisco, CA
Scott Tipton, Concord, CA
John Hewitt, Berkeley, CA
Greg Lowry, San Francisco, CA
Kyle Frostdale, Oakland, CA
Hannah Dunton, Vallejo, CA
Jim McCabe, La Cañada Flintridge, CA
Djian Salgado, San Francisco, CA
Jim Spooner, Manteca, CA
Ryan Lund, San Francisco, CA
Frederica Davis, Fairfax, CA
Kristian Wittstock, San Francisco, CA
Ryan Enco, Oakland, CA
Anna M, San Francisco, CA
Lindsay Ward, San Francisco, CA
Stephen Dempsey, San Francisco, CA
Bill McLaughlin, San Francisco, CA
Tim Callahan, Oakland, CA
Andy Braden, Sebastopol, CA
Justin Delsignore, Berkeley, CA
Aaron Humpert, Alameda, CA
Michael Archer, Napa, CA
Tamara Voyles, Sebastopol, CA
Simon Read, Santa Clara, CA
Raymond Wu, San Jose, CA
Patrick Kim, El Sobrante, CA
Christopher Lish, San Rafael, CA
Caius Van Nouhuys, San Francisco, CA

Christina Li, San Francisco, CA
Larry Huynh, Oakland, CA
Jackson Rogers, San Francisco, CA
John Devincenzi, San Carlos, CA
Laura Arsenault, Napa, CA
Lily Mejia, Hemet, CA
Ryan Acebo, Oakland, CA
Mahin Charles, San Francisco, CA
Tai Stillwater, San Francisco, CA
Maxine Zylberberg, San Francisco, CA
Suzanne Kunstman, Rio Vista, CA
Nina Atkind, San Francisco, CA
Kay Low, Kensington, CA
Ben Porterfield, San Francisco, CA
Allison Arnold, San Francisco, CA
Katie Lanning, Carmel, CA
Newara Faltas, San Francisco, CA
Ronnie Weisberg, Pacifica, CA
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Southern Distinct Population Segment of North American Green Sturgeon
(Acipenser medirostris)

5-Year Review:
Summary and Evaluation

National Marine Fisheries Service
California Central Valley Office
Sacramento, California

5-YEAR REVIEW

Species reviewed: Southern Distinct Population Segment of the North American Green Sturgeon (*Acipenser medirostris*)

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5-YEAR REVIEW

Species reviewed: Southern Distinct Population Segment of the North American Green Sturgeon (*Acipenser medirostris*)

1.0 GENERAL INFORMATION

1.1 Reviewers

Lead Regional or Headquarters Office: *Page Vick, West Coast Regional Office, California Central Valley Office, Sacramento, CA, (916) 594-4406*

Cooperating Regional Offices: *Scott Anderson, Lacey Office, (360) 753-5828; Lynne Krasnow, Interior Columbia Basin Office, (503) 231-2163; Matt Goldsworthy, Coastal California Office, (707) 825-1621; Brian Meux, California Coastal Office, (707) 575-1253; Susan Wang, Long Beach Office, (562) 980-4199; Thomas Coleman, Long Beach Office, thomas.coleman@noaa.gov*

Cooperating Science Centers: *Peter Dudley, Southwest Fisheries Science Center Santa Cruz, CA, (831) 420-3924*

1.2 Methodology used to complete review

The 5-year review was conducted by the NOAA NMFS West Coast Region's California Central Valley Office, in collaboration with personnel at other NOAA NMFS West Coast Region offices (California Coastal Office, Long Beach Office, Interior Columbia Basin Office, and Lacey Office). The review process included collecting information through the following: (1) a literature search for information published since the last review (2015); (2) publication of a Federal Register (FR) notice soliciting new information about the Southern Distinct Population Segment (DPS) of North American green sturgeon (85 FR 12905; March 5, 2020); and (3) email and phone contact with knowledgeable individuals at universities, tribal agencies, and state and federal government agencies. Overall, the best available scientific information was used to evaluate if the biological status and the Endangered Species Act (ESA) listing factors have changed over the last 5 years.

1.3 Background

1.3.1 FR Notice citation announcing initiation of this review

A Federal Register notice (85 FR 12905, March 5, 2020) announced the initiation of this review.

1.3.2 Listing History

Original Listing
FR notice: 71 FR 17757
Date listed: April 7, 2006
Entity listed: Southern Distinct Population Segment (DPS) of North American green sturgeon
Classification: Threatened

1.3.3 Associated rulemakings

Critical Habitat: On Oct. 9, 2009, NMFS designated critical habitat for the Southern DPS of North American green sturgeon (74 FR 52300).

ESA 4(d) rule: On June 2, 2010, NMFS published final Endangered Species Act protective regulations (ESA 4(d) rule) for the Southern DPS of North American green sturgeon (75 FR 30714).

1.3.4 Review History

Status Review: In 2002, a status review was conducted by a Biological Review Team (BRT) in response to a 2001 petition to list North American green sturgeon under the Endangered Species Act (Adams et al. 2002). The BRT identified the Northern and Southern DPS structure that is currently applied and concluded that green sturgeon in both DPSs should be placed on the Species of Concern list (then the Candidate species list) and their status reviewed within five years (Adams et al. 2002). In 2005, NMFS' Southwest and Northwest Fisheries Science Centers updated the Status Review as a result of a 2004 court ruling remanding to NMFS for further consideration of whether green sturgeon are endangered or threatened in a "significant portion of the species' range" (BRT 2005). The BRT updated the review and concluded that the Northern DPS was not in danger of extinction now or likely to become endangered in the foreseeable future throughout all of its range. All but one member of the BRT concluded that green sturgeon in the Southern DPS were likely to become endangered in the foreseeable future throughout all of its range.

On April 7, 2006, NMFS published notification of the listing of the Southern DPS of North American green sturgeon as Threatened (71 FR 17757). The DPS structure for North American green sturgeon was originally defined as follows: (1) a Northern DPS consisting of populations in coastal watersheds northward of and including the Eel River (''Northern DPS''); and (2) a Southern DPS consisting of coastal and Central Valley populations south of the Eel River, with the only known spawning population in the Sacramento River (''Southern DPS'') (71 FR 17757; April 7, 2006). The definition was slightly revised for accuracy with the announcement of critical habitat as follows: (1) a Northern DPS consisting of populations originating from coastal watersheds northward of and including the Eel River (i.e., the Klamath and Rogue rivers) ("Northern DPS"); and (2) a Southern DPS consisting of populations originating from coastal watersheds south of the Eel River, with the only known spawning population in the Sacramento River ("Southern DPS") (74 FR 52300; Oct. 9, 2009). In the April 7, 2006 listing notification (71 FR 17757), the Northern DPS was identified as a NMFS Species of Concern but was not listed under the ESA. NMFS stated that it would revisit the status of both DPSs' in five years' time. The 2015 5-year review focused only on the status of the Southern DPS. The Northern DPS status was the focus of a separate informal report that was been added to the 2015 record. The 2015 review determined that there was no change in listing and that the Southern DPS will remain Threatened. The 2018 Recovery Plan for the Southern DPS of the North American Green Sturgeon provided demographic criteria for recovery and ranking of threats by listing factor, life

stage, and habitat type (NMFS 2018). This 5-year status review focuses only on the status of the Southern DPS.

1.3.5 Species' Recovery Priority Number at start of 5-year review

The recovery priority number for the Southern DPS green sturgeon is 6C, as reported in the NMFS Recovering Threatened and Endangered Species FY 2017-2018 Report to Congress (available at www.fisheries.noaa.gov/feature-story/recovering-threatened-and-endangered-species-report-congress-2017-2018). A Recovery Priority Number of 6C indicates a moderate magnitude of threat in some regions, a high recovery potential in many regions, and the presence of conflict with economic and resource use interests.

1.3.6 Recovery Plan or Outline

Name of plan or outline: Recovery Plan for the Southern Distinct Population Segment of North American Green Sturgeon (*Acipenser medirostris*)
Date issued: August 8, 2018
Dates of previous revisions, if applicable: N/A

2.0 REVIEW ANALYSIS

2.1 Application of the 1996 Distinct Population Segment (DPS) policy

2.1.1 Is the species under review a vertebrate?

Yes.

2.1.2 Is the species under review listed as a DPS?

Yes.

2.1.3 Was the DPS listed prior to 1996?

No.

2.1.4 Is there relevant new information for this species regarding the application of the DPS policy?

Yes, studies published since 2015 further confirm the DPS structure of North American green sturgeon as defined in Section 1.3.4 of this review. These new studies are covered in Sections 2.3.1.3 and 2.3.1.5 of this review. As discussed in the previous review, Israel et al. (2009) upholds the Northern and Southern DPS determination of spawning rivers. Telemetry studies and unpublished data also confirm the DPS structure (Lindley et al. 2008, Lindley et al. 2011). Spawning of Southern DPS green sturgeon was confirmed in Sacramento River tributaries, the Feather and Yuba rivers during years with higher flow (Seesholtz et al. 2015, Beccio 2018, 2019). Green sturgeon spawning was documented in the Eel River, and all genetic samples represented the Northern DPS (Stillwater Sciences and Wiyot Tribe Natural Resources 2017). Schreier et al. (2016) examined size differences between Northern and Southern DPS holding in the Columbia River estuary, Willapa Bay, and Grays Harbor, with Northern DPS being

significantly smaller than Southern DPS. Green sturgeon concentrating in the Columbia River estuary and Willapa Bay were primarily Southern DPS, while green sturgeon in Grays Harbor were represented equally among the DPSs (Schreier et al. 2016). The population found within the Gulf of Farallones were primarily Southern DPS, while the population found in the plume of the Columbia River were primarily Northern DPS (Anderson et al. 2017). A genetically confirmed green sturgeon was observed holding in a pool in the Stanislaus River near Knights Ferry, California (Anderson et al. 2018). In April 2020, a confirmed green sturgeon was observed in the San Joaquin River near the mouth of the Merced River (Root et al. 2020).

2.2 Recovery Criteria

2.2.1 Does the species have a final, approved recovery plan containing objective, measurable criteria?

Yes.

2.2.2 Adequacy of recovery criteria

2.2.2.1 Do the recovery criteria reflect the best available and most up-to-date information on the biology of the species and its habitat?

Yes.

2.2.2.2 Are all of the 5 listing factors that are relevant to the species addressed in the recovery criteria (and is there no new information to consider regarding existing or new threats)?

Yes.

2.2.3 List the recovery criteria as they appear in the recovery plan, and discuss how each criterion has or has not been met, citing information.

The recovery criteria are organized according to: (1) Demographic Recovery Criteria addressing abundance, distribution, productivity, and diversity; and (2) Threat-Based Recovery Criteria addressing the significant known threats impeding recovery.

Demographic Recovery Criteria:

1. The adult Southern DPS green sturgeon census population remains at or above 3,000 for 3 generations (this equates to a yearly running average of at least 813 spawners for approximately 66 years). In addition, the effective population size must be at least 500 individuals in any given year and each annual spawning run must comprise a combined total, from all spawning locations, of at least 500 adult fish in any given year.
   a. *Criteria Not Met:* The estimated total population of Southern DPS green sturgeon is 17,548 individuals, with an estimated 2,106 adults (Mora et al.

2018). The adult population does not met the criteria of a yearly average 3,000 adults. Reported spawner counts have been less than 500 in the Sacramento River (Mora et al. 2018, Peter Dudley, SWFSC pers. comm. September 28, 2020). Currently, there are no reliable estimates for spawner counts for the Feather and Yuba rivers.

2. Southern DPS green sturgeon spawn successfully in at least two rivers within their historical range. Successful spawning will be determined by the annual presence of larvae for at least 20 years.
   a. *Criteria Not Met:* Although spawning has been reported in the Feather and Yuba rivers, continuous spawning in these rivers has not been observed (Seesholtz et al. 2015, Beccio, 2018, 2019). Continuous spawning has been reported in the Sacramento River since 1995 (Poytress et al. 2015, Poytress 2020).
3. A net positive trend in juvenile and subadult abundance is observed over the course of at least 20 years.
   a. *Criteria Not Met:* Although juvenile monitoring for Southern DPS green sturgeon is occurring, it does not currently inform population indices (Beccio 2020).
4. The population is characterized by a broad distribution of size classes representing multiple cohorts that are stable over the long term (20 years or more).
   a. *Criteria Not Met:* Beamesderfer et al. (2007) estimated that adult, subadult, and juvenile green sturgeon in a hypothetical population at equilibrium would comprise 12%, 63%, and 25% of the population, respectively. These values are the best available information to date and can serve as a guideline for evaluating population equilibrium in the Southern DPS green sturgeon. However, further modeling may identify different benchmarks for measuring population equilibrium, and a larger percentage of younger fish may be present in the Southern DPS in the early stages of potential recovery. Further research needs to be done to inform this criterion.
5. There is no net loss of Southern DPS green sturgeon diversity from current levels.
   a. *Criteria Not Met:* Spawning habitat was used as a proxy for diversity because diversity is closely tied with abundance, distribution, and productivity. Since the previous review, spawning habitat available to Southern DPS has not increased.

Threat-Based Recovery Criteria

A. Present or Threatened Destruction, Modification, or Curtailment of a Species Habitat or Range
   a. Access to spawning habitat is improved through barrier removal or modification in the Sacramento, Feather, and/or Yuba rivers such that successful spawning occurs annually in at least two rivers.

Successful spawning will be determined by the annual presence of larvae for at least 20 years.

i. *Criteria Not Met:* Barriers to migration caused by impoundments continue to be a very high threat to green sturgeon recovery. Although access to spawning habitat was improved with the decommissioning of Red Bluff Diversion Dam (RBDD) in 2012, more needs to be done such as volitional fish passage above spawning barriers at Sunset Pumps rock weir on the Feather River and Daguerre Point Dam on the Yuba River. Threats identified at listing and during recovery plan development include: reduction of spawning areas (i.e., barriers), screened and unscreened water diversions, flow, temperature, and contaminants. Channel control structures, impoundments, and non-native species competitions altering prey base are very high threats.

b. Volitional passage is provided for adult green sturgeon through the Yolo and Sutter bypasses.
    i. *Criteria Not Met*: Volitional passage for adult green sturgeon has been addressed at Yolo Bypass but further passage projects at Yolo and Sutter bypasses need to be completed (e.g., Fremont Weir Adult Fish Passage, Tisdale Weir improvements, Sacramento Bypass Fish Passage). Water diversions remain a medium threat.
c. Water temperature and flows are provided in spawning habitat such that juvenile recruitment is documented annually. Recruitment is determined by the annual presence of age-0 juveniles in the lower Sacramento River or San Francisco Bay Delta Estuary (SFBDE). Flow and temperature guidelines have been derived from analysis of inter-annual spawning and recruitment success and are informing this criterion
    i. *Criteria Not Met:* Water temperature and flow targets have not been developed for green sturgeon. Water temperature and flow remain a medium threat and remain a high priority for research and development.
d. Adult contaminant levels are below levels that are identified as limited population maintenance and growth
    i. *Criteria Not Met*: Additional research focusing on contaminant impacts on green sturgeon life history is needed. The impacts of contaminants remain a medium threat and a high priority for research and development.

B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

a. Take of adults and subadults through poaching and state, federal and tribal fisheries is minimal and does not limit population persistence and growth.
    i. *Criteria Not Met*: Any take of subadult or adult Southern DPS green sturgeon may limit population productivity. Further research is needed to determine the mortality estimates of poaching and fisheries bycatch. Fishery Management and Evaluation Plans (FMEP) are needed by each state to demonstrate the impact of unintentional death related to fisheries. Overutilization for scientific research and poaching remain a high and moderate threat, respectively. Retention of North American green sturgeon is not currently permitted in any state fishery. The California Department of Fish and Wildlife (CDFW), Washington Department of Fish and Wildlife (WDFW), and Oregon Department of Fish and Wildlife (ODFW) all prohibit the commercial retention and sale of green sturgeon.

C. Disease and Predation
    a. No threat-based criteria were developed for this category.
        i. Disease and predation associated with water quality, native and non-native species, and marine mammals remain a high threat. The extent of the potential threat from disease and predation in terms of limiting population growth and recovery remain an area where further research is needed.

D. Inadequacy of Existing Regulatory Mechanisms
    a. No threat-based criteria were developed for this category.
        i. Fishing regulations were considered a high threat at listing; however, regulations were put in place (Recreational: 2006 in California, 2007 in Columbia River and Washington, and 2010 in Oregon; Commercial: 1917 in California, 2006 in Columbia River and Washington, and 2010 in Oregon) prohibiting the take of North American green sturgeon. Derelict fishing gear may pose a risk to green sturgeon. Continued improvements are needed to regulatory mechanisms associated with the following factors: (1) sturgeon passage improvements at critical locations (e.g., rock weir at Sunset Pumps, Daguerre Point Dam); (2) modification of impoundment operations or facilities to address flow, water temperature, and sediment impacts (e.g., Oroville-Thermalito Complex); (3) improvements of lock and gate operations at the Port of Sacramento and Delta Cross Channel; (4) enforcement of poaching and other fishery regulations such as bycatch in state fisheries; (5)

screening criteria and regulation for agriculture, municipalities, and industrial water diversions; (6) land use regulations for non-point and point source contaminants; (7) control of invasive species (e.g., overbite clams); (8) response plans for oil and chemical spills; and (9) permitting of offshore and nearshore kinetic energy projects. Criteria for screening and fish passage for green sturgeon has yet to be developed. These criteria are needed to prevent entrainment of juvenile green sturgeon at water diversions. Once developed, these criteria should be implemented throughout Southern DPS green sturgeon critical habitat.

E. Other Natural or Manmade Factors Affecting Its Continued Existence
    a. Operation guidelines and/or fish screens are applied to water diversions in mainstem Sacramento, Feather, and Yuba rivers and SFBDE such that early life stage entrainment is below a level that limits juvenile recruitment.
        i. *Criteria Not Met*: Current screen criteria (developed for salmonids) may not be useful in preventing Southern DPS green sturgeon from impingement and entrainment. Currently, no screen criteria has been developed for green sturgeon. Although larger water diversions (>250 cfs) in the Sacramento River have been screened, there are still thousands of unscreened diversions (<250 cfs). Water diversions remain a threat medium threat for juvenile green sturgeon and a high priority for research and development.
    b. No threat-based criteria were developed for threats associated with competition for habitat and mortality factors associated with electromagnetic fields, anthropogenic underwater sounds, and entrainment from hydrokinetic projects.
        i. Competition for habitat and mortality factors associated with electromagnetic fields, anthropogenic underwater sounds, and entrainment from hydrokinetic projects continue to be a high or medium threat. Global climate change continues to be a very high threat.

2.3 Updated Information and Current Species Status

2.3.1 Biology and Habitat

2.3.1.1 Background and new information on the species' biology and life history:

Research conducted and published since 2015 confirms and enhances our understanding of the biology and life history of Southern DPS green sturgeon, including reproductive characteristics. The following is a summary of this new information. Where reference is made to North American green sturgeon, the

information is relevant to both DPSs or the original work did not specify the DPS under study. The DPS is specified where known. Much of the laboratory work conducted to date used Northern DPS broodstock, but the results are relevant to our understanding of green sturgeon biology and are reviewed here.

Spawning Adults: North American green sturgeon reach sexual maturity at about 15 years of age or a total length (TL) of 150-155 cm for Southern DPS individuals (Van Eenennaam et al. 2006). Southern DPS green sturgeon typically spawn every three to four years (range two to six years) and spawning occurs primarily in the Sacramento River (Brown 2007, Poytress et al. 2015, Mora et al. 2018). Since 2015, spawning has also been documented in the Feather and Yuba rivers, which are tributaries to the Sacramento River (Seesholtz et al. 2015, Beccio 2018, 2019). Adult Southern DPS green sturgeon enter San Francisco Bay in late winter through early spring, migrate upstream, and spawn from April through early July, with peaks of activity influenced by factors including water flow and temperature (Heublein et al. 2009, Poytress et al. 2015, Miller et al. 2020). Miller et al. (2020) showed that adult Southern DPS green sturgeon use the mainstem Sacramento River, Miner-Sutter Slough, and Steamboat Slough for upstream migration during spawning season.

Spawning primarily occurs in cool sections of the upper mainstem Sacramento River in deep pools (averaging 8-9 meters (m) in depth; Wyman et al. 2018) containing small to medium sized gravel, cobble or boulder substrate (Klimley et al. 2015, Poytress et al. 2015, Wyman et al. 2018). Water flow is an important cue in spawning migration for both Northern and Southern DPS green sturgeon (Benson et al. 2007, Erickson and Webb 2007, Heublein et al. 2009, Poytress et al. 2015, Steel et al. 2019). White sturgeon spawning has been documented to occur after elevated flows (Schaffter 1997, Jackson et al. 2016), suggesting a connection between flow and spawning. Brown (2007) documented Southern DPS green sturgeon spawning both above and directly below the site of the RBDD on the Sacramento River. Behavioral observations in Thomas et al. (2014) indicate that males may fertilize the eggs of multiple females. Prior to decommissioning in 2011, the gates at RBDD would be lowered for several months of the year from late spring through summer, prohibiting many Southern DPS green sturgeon from ascending upstream to spawn. Following the decommissioning of the dam, Southern DPS green sturgeon are able to access spawning habitat upstream of RBDD (Mora et al. 2018, Steel et al. 2019).

Post-spawn fish may hold for several months in the Sacramento River and out-migrate in the fall or winter or move out of the river quickly during the spring and summer months, with the holding behavior most commonly observed (Heublein et al. 2009, Mora 2016, Miller et al. 2020, Seesholtz 2020). Miller et al. (2020) documented one post-spawn adult Southern DPS green sturgeon holding for over a year in the Sacramento River and then exiting over winter. Post-spawn outmigration through the SFBDE is also variable, with some individuals migrating to the Pacific Ocean rather quickly (2-10 days) and others remaining in the estuary for a number of months after leaving upstream holding habitats

(Heublein et al. 2009, Miller et al. 2020). Post-spawn individual holding times vary and are likely influenced by delayed flow cues, which aid in outmigration (Miller et al. 2020). Miller et al. (2020) observed that although adult Southern DPS green sturgeon take several routes to reach spawning grounds, they generally migrate out of spawning grounds through the mainstem Sacramento River.

Eggs, Larvae, and Young-of-Year Juveniles: North American green sturgeon eggs primarily adhere to gravel or cobble substrates, or settle into crevices (Van Eenennaam et al. 2001, Poytress et al. 2011). See the 2015 review and 2018 Recovery Plan for the full description of egg and larvae development of green sturgeon. Temperature influences egg development and hatching rate according to laboratory studies and is likely a factor in Southern DPS recovery. In laboratory experiments, optimal temperature for egg development ranged from 14-17℃, with temperatures higher than 20℃ being detrimental for embryos (Van Eenennaam et al. 2005). Water temperature also impacts larval growth of green sturgeon. Lab experiments show optimal growth at 15℃, while temperatures less than 11℃ and temperatures greater than 19℃ decreased growth rate (Poletto et al. 2018). Juvenile Northern DPS green sturgeon can endure elevated temperatures in the laboratory (up to 28℃ tested) without showing compromised swimming performance, but temperatures above 19°C were correlated with higher expression of heat shock proteins (Allen et al. 2006, Verhille et al. 2016, Poletto et al. 2018). While much of the laboratory data reviewed above have been generated using Northern DPS broodstock, it is likely applicable to the life history of the Southern DPS. Temperatures in the upper Sacramento River documented during the estimated Southern DPS spawning period had a mean temperature of 13.5°C (Poytress et al. 2015).

Green sturgeon move downstream as they transition from larvae and young-of-year into juveniles. In the laboratory, juvenile Northern DPS green sturgeon were highly tolerant to changes in salinity during the first six months (Allen et al. 2011) and the ability to transition to seawater occurred at 1.5 years of age (Allen and Cech 2007). Based on length of juvenile sturgeon captured in the SFBDE, Southern DPS green sturgeon migrate downstream toward the estuary between six months and two years of age (Radtke 1966). Sardella and Kultz (2014) found in a laboratory setting that Northern DPS green sturgeon are able to tolerate large fluctuations in salinity (from 0 g/mL to 24 g/mL over a 12-hour period); however, they will develop high levels of physiological stress from these large fluctuations (Sardella and Kultz 2014).

Juveniles, Subadults, and Adults: Since the 2015 review, more is known about Southern DPS green sturgeon rearing and foraging in the SFBDE. Several telemetry studies provide insight to juvenile, subadult, and adult habitat use in the SFBDE. As part of a three year study (2009 to 2012) with 106 acoustic receivers deployed from the Benicia Bridge to the Port of Oakland, Chapman et al. (2019) found that nine tagged adult green sturgeon were primarily detected in depths greater than five meters within the study area throughout the year. Adult green

sturgeon (n = 134) were detected year-round throughout the study area, including two periods where small relative increases in overall adult green sturgeon detections were observed, February to March and June to September (Chapman et al. 2019). Chapman et al. (2019) also found that 81% of the detected green sturgeon were observed at least in one dredged area or dredged material placement site. Kelly et al. (2020) described how, during the summer and fall months of 2001 and 2002 in San Pablo Bay, two tagged green sturgeon used selective tidal stream transport to move throughout the SFBDE during daily, non-migratory movements. The tagged green sturgeon swam with the current near the surface in deeper, high-current areas of the SFBDE, and they swam along the bottom in shallow areas with little current (Kelly et al. 2020). This behavior is thought to maximize swimming efficiency and save energy for the fish. When the current was going in what was deemed to be an unfavorable direction, then the fish appeared to rest on the bottom rather than swim against the current (Kelly et al. 2020). It should be noted that this study is based on a small sample size (two green sturgeon), and that more research is needed to explore this behavior and its potential energetic benefits. A telemetry study focusing on green and white sturgeon recorded high juvenile, subadult, and adult Southern DPS green sturgeon presence year-round in the Delta, Suisun Bay, San Pablo Bay, and Central San Francisco Bay (Miller et al. 2020). Currently, a telemetry study tagging and tracking juvenile Southern DPS green and white sturgeon is being conducted in the Delta by CDFW. From 2015 to 2020, 143 juvenile Southern DPS green sturgeon have been captured and the majority of those juveniles have been tagged with acoustic telemetry tags (Beccio 2020, 2021). This study is not designed to provide abundance estimates or annual recruitment indices, but annual catch per unit effort (CPUE) indicates that successful recruitment is likely dependent on water year type. CPUE for age-0+ juveniles recruited during spawning years with high Delta outflows as determined by water year type was markedly higher than for recruitment from spawning years with lower Delta outflows (Beccio 2020).

Subadult and adult North American green sturgeon spend most of their life in the coastal marine environment. Miller et al. (2020) described subadult green sturgeon leaving the SFBDE and primarily swimming north along the Pacific Coast. As described in the 2015 review, green sturgeon typically occupy depths of 20 to 70 m while in marine habitats (Erickson and Hightower 2007, Huff et al. 2011) and make rapid vertical ascents while in marine environments, often at night (Erickson and Hightower 2007). Temperatures occupied in the marine environment ranged from 7.3 to 16°C, with a range of mean temperatures from 10.5 to 12.5°C (Erickson and Hightower 2007, Huff et al. 2011). Southern DPS green sturgeon are found in high concentrations in coastal bays and estuaries along the west coast of North America during the summer and autumn, particularly in Willapa Bay, Grays Harbor, and the Columbia River estuary (Lindley et al. 2008, Moser et al. 2016, Schreier et al. 2016). Moser et al. (2017) described feeding pits occupied by green sturgeon during summer months in Willapa Bay. Green sturgeon were documented feeding during high tide on non-

vegetated littoral mud flats. Green sturgeon presence on the mud flats was negatively correlated with invasive seagrass (*Zostera japonica*) and positively correlated with the abundance of thalassinid shrimp burrows (Moser et al. 2017).

Summary: Overall, the new information on the biology of the species provides insights for protecting Southern DPS green sturgeon habitat in freshwater, estuarine, and marine environments. As noted in the 2015 review, access to spawning habitat has been improved with the decommissioning of RBDD in the Sacramento River and opening of Shanghai Bend in the Feather River Since the 2015 review, there have also been fish passage improvements at Fremont Weir. Raising the gates of RBDD eliminated a passage issue and allowed more Southern DPS green sturgeon to access spawning areas upstream of RBDD (Thomas et al. 2014, Steel et al. 2019). Recruitment data are not presently available to measure the impact of the removal of RBDD on Southern DPS reproduction. Laboratory and field studies of larval and young-of-year juvenile Northern DPS green sturgeon indicate optimal thermal regimes in freshwater environments and resilience to salinity and temperature changes. Field studies explore movement and habitat use of all life stages of green sturgeon in the freshwater, estuarine, and marine environments. Limited studies have been conducted to examine rearing and foraging of juvenile Southern DPS green sturgeon in the spawning and rearing areas of the Sacramento River Basin as well as the SFBDE. Estuaries along the west coast are important habitats for subadult and adult Southern DPS green sturgeon. Although passage improvements have occurred at Fremont Weir and spawning events have been documented in the Feather and Yuba rivers, no changes to the species status or threats are evident since the last review.

2.3.1.2 Abundance, population trends (e.g., increasing, decreasing, stable), demographic features (e.g., age structure, sex ratio, family size, birth rate, age at mortality, mortality rate, etc.), or demographic trends:

Since 2015, modeling, genetic, and field-based studies, many targeting species other than green sturgeon, have provided information on the Southern DPS green sturgeon population. However, these studies have not led to a reliable population estimate, nor do they provide information on long-term trends, and demographic features or trends needed to evaluate the recovery of Southern DPS green sturgeon because we lack data on egg-to-larva survival, juvenile recruitment, information on juveniles and subadult life stages, and mortality estimates for all life stages. We are working to gather this data for the future. For now, we will not discuss trends but will rather focus on population abundance.

A Southern DPS population estimate was developed by Mora et al. (2018) through Dual Frequency Identification Sonar (DIDSON) surveys of aggregation sites conducted from 2010-2015 in the upper Sacramento River. Mora et al. (2018) estimated the total population size to be 17,548 individuals (95% confidence interval [CI] = 12,614-22,482). The SWFSC recently updated the total population estimate to 17,723 (Dudley 2021).These surveys estimate the

abundance of Southern DPS adults at 2,106 individuals (95% CI = 1,246-2,966) (Mora 2016, Mora et al. 2018). A conceptual demographic structure applied to the adult population estimate resulted in a Southern DPS subadult population estimate of 11,055 (95% CI = 6,540-15,571) and juvenile population estimate of 4,387 (95% CI = 2,595-6,179) (Mora et al. 2018). The DIDSON surveys and associated modeling will eventually provide population trend data. Demographic Criterion 4 requires that the population is characterized by a broad distribution of size classes representing multiple cohorts that are stable over at least 20 years. Continued monitoring of all life stages is needed to inform this criterion.

Spawning Adults: The 2015 review used genetic analyses from Israel and May (2010) to estimate the number of Southern DPS green sturgeon spawning individuals in the upper Sacramento River (above RBDD). The study indicated that an estimated 10-28 individual Southern DPS green sturgeon effectively reproduced upstream of RBDD in the upper Sacramento River annually (Israel and May 2010). This spawning population estimate was stable over the five year sampling period (2002-2006). It should be noted that the study was conducted prior to the decommissioning of RBDD (2011) when upriver access by Southern DPS green sturgeon to spawning habitat was limited. The Southwest Fisheries Science Center (SWFSC) continued Mora et al. (2018)'s work and conducted DIDSON surveys at aggregation sites in the upper Sacramento River from 2016-2020. Spawner abundance was estimated for the Sacramento River using the DIDSON surveys and associated modeling (Mora et al. 2018). Spawner counts (with 95% CI): 2010: 244 $\pm$ 36; 2011: 223 $\pm$ 20; 2012: 325 $\pm$ 26; 2013: 341 $\pm$ 29; 2014: 530 $\pm$ 31; 2015: 431 $\pm$ 28; 2016: 375 $\pm$ 19; 2017: 82 $\pm$ 9; 2018: 447 $\pm$ 44; and 2019: 252 $\pm$ 44 (Dudley 2020). The confidence interval is different from Mora et al. (2018) due to an error in previous reports. Actual numbers may be slightly different because of slight modification to the method of calculating the area scanned (Dudley 2020). Mora et al. (2018) also showed that spawning periodicity ranged from two to six years with the majority of repeat spawners returning every four years. Continued DIDSON monitoring of aggregation sites is needed in the upper Sacramento River to assess population abundance and establish trends. Demographic Recovery Criterion 1 requires spawning population size to be at least 500 individuals in any given year, which only occurred in 2014.

The number of holding areas (i.e., specific areas in the river where green sturgeon congregate) occupied by Southern DPS green sturgeon in the Sacramento River has not been updated since the last review. The previous review reported 22 holding areas, while there were 125 surveyed areas that were considered suitable based on depth (Mora 2016). Holding areas with sturgeon were distributed across most (i.e., 75 miles) of the study area. There was also a difference in the holding areas occupied by sturgeon during any given sampling year: some areas were occupied in all years, some in just one year, and some in two, three, or four years. Thus, there is temporal and spatial variation in the holding areas occupied by Southern DPS green sturgeon within the Sacramento River.

Juveniles: Young-of-year presence has been incidentally documented during juvenile salmonid monitoring efforts at the RBDD and near the Glenn Colusa Irrigation District (GCID) pumping facility, both located on the upper Sacramento River. Using rotary screw traps set downstream of RBDD, USFWS captured 21,057 larval Southern DPS green sturgeon from 1994 to 2020, with approximately 4,100 larvae collected in 2019. Sampling was limited in 2020 due to COVID-19 sampling restrictions; only 157 larvae were collected in 2020 (Poytress 2020). From 2015 to 2020, 224 young-of-year juvenile Southern DPS green sturgeon were captured in the Sacramento Trawl conducted by USFWS. Several juveniles were tagged with Juvenile Salmonid Acoustic Tracking (JSAT) or 303Hz tags to monitor entrance to the Delta (Poytress 2020). The 2015 review stated that over 2,000 Southern DPS green sturgeon larvae were also collected in fyke nets and rotary screw traps at GCID from 1986 to 2003. From March 2013 to December 2019, 64 Southern DPS green sturgeon juveniles were collected in the rotary screw traps at GCID (fork length [FL] 23-325 mm). Caution is needed in interpreting these data as reflective of abundance since the surveys were not designed to measure green sturgeon abundance. Annual distributions of larvae have been found to peak during June and July at RBDD (with the exception of 2012 when only a June peak was observed) and July at GCID (Adams et al. 2002, Adams et al. 2007, Poytress et al. 2015). Demographic Recovery Criterion 2 requires successful spawning of Southern DPS green sturgeon in at least two rivers within their historical range. Successful spawning has been documented for over 20 years in the Sacramento River, but successful spawning has not been consistently documented in the Feather or Yuba rivers.

Subadults and Adults: The CDFW conducts annual field sampling for sturgeon in San Pablo and Suisun Bays in the months of August through October. Reports from 2015-2019 describe encounters with relatively small numbers of subadult and, to a lesser extent, adult Southern DPS green sturgeon (2015: 18; 2016: 0; 2017: 8; 2019: 3); annual reports are available at https://wildlife.ca.gov/Conservation/Delta/Sturgeon-Study/Bibliography). Sampling did not occur in 2018 due to boat issues (Kelly 2021). Since the study is primarily designed to study white sturgeon, the results cannot be interpreted for estimates of or trends in Southern DPS abundance. Demographic Recovery Criterion 3 requires a net positive trend in juvenile and subadult abundance observed over the course of at least 20 years. There are currently no studies that address juvenile and subadult abundance of Southern DPS green sturgeon.

Since the 2015 review, ODFW and WDFW have not generated estimates of subadult and adult Northern and Southern DPS green sturgeon in Willapa Bay, Grays Harbor, and the Columbia River based on tagging and recapture studies and subsequent analysis. WDFW tagged green sturgeon in Willapa Bay and Grays Harbor in the fall of 2020 but no data have been published. During fall 2020 sampling, 123 green sturgeon were tagged with Passive Integrative Transponder (PIT) tags and 60 green sturgeon were tagged with acoustic telemetry tags. All

sturgeon were sampled for blood and genetics. WDFW is monitoring sturgeon movement in the coastal estuaries with acoustic arrays (Heironimus 2021a).

Summary: In summary, recent studies provide information on the population abundance of Southern DPS green sturgeon. Future surveys and abundance estimates will provide a basis for understanding the population trajectory of the Southern DPS. Since there are no past survey data or abundance estimates that can be used as a reference point, these data do not provide a basis for changing the status of the Southern DPS. Consistent with the 2015 review, data suggest that the spawning population of the Southern DPS is smaller than the Northern DPS, which is consistent with the fact that Southern DPS is listed under the ESA, and the Northern DPS is not.

The spawning population of the Southern DPS in the Sacramento River congregates in a limited area of the river compared to potentially available habitat. The reason for this is unknown. This is concerning given that a catastrophic or targeted poaching event impacting just a few holding areas could affect a significant portion of the adult population. Removal of the RBDD barrier did allow Southern DPS green sturgeon to freely access a larger area of the river over their entire spawning period (Thomas et al. 2014), so the Southern DPS likely now holds in a larger area of the river compared to when RBDD was operating in 2011 (Mora et al. 2018, Steel et al. 2019). Continued monitoring of the adult population in the Sacramento River and its tributaries will provide valuable trend data and information to enhance spatial protection. Of note is that all of the holding areas where green sturgeon were found in the Sacramento River in the DIDSON survey area (Highway 32 overcrossing to the city of Redding) are currently included in the area where CDFW restrictions prohibit fishing for all sturgeon species (See Section 2.3.2.2). No changes to the species status or threats are evident since the last review based on the reviewed information on abundance and demographic trends.

2.3.1.3 Genetics, genetic variation, or trends in genetic variation (e.g., loss of genetic variation, genetic drift, inbreeding, etc.):

Since the 2015 review, there were no genetic studies that furthered our understanding of genetic variation or trends in genetic variation for green sturgeon. However, new genetic techniques were used to distinguish between Northern and Southern DPS in coastal estuaries and to identify green sturgeon in new locations, which provides new information on the species. In testing a new genetic technique to differentiate Northern DPS and Southern DPS, Anderson et al. (2017) observed two “extremely” different proportions of each population in two separate locations—Columbia Plume region (39 Southern DPS of 86 samples) and the Gulf of the Farallones (103 Southern DPS of 104 samples). Seesholtz et al. (2015) documented Southern DPS green sturgeon spawning in the Feather River, a tributary to the Sacramento River. Anderson et al. (2018) confirmed Southern DPS green sturgeon holding in a pool near Knights Ferry, CA in the Stanislaus River using a GoPro and eDNA techniques to confirm species.

Stillwater Sciences and Wiyot Tribe Natural Resources (2017) documented green sturgeon in the Eel River, and of the five genetic samples taken, they only found Northern DPS. Schreier et al. (2016) examined size differences between Northern DPS and Southern DPS in the Columbia River estuary, Willapa Bay, and Grays Harbor and found Northern DPS were significantly smaller than Southern DPS at 120.0 cm and 138.3 cm, respectively. They also found that the populations congregating in the Columbia River estuary and Willapa Bay were majority Southern DPS, while the populations congregating in Grays Harbor had nearly equal representation of each DPS. CDFW has also documented a single spawning event in the Yuba River in 2018 (estimated spawning date June 13) and captured a single juvenile sturgeon (from estimated spawning on July 19) during the 2019 season (Beccio 2018, 2019). During monitoring activities on April 11, 2020, a green sturgeon was captured within the boundaries of the San Joaquin River Restoration Program Area near the mouth of the Merced River in Hills Ferry, California (Root et al. 2020). The 126 cm FL green sturgeon was captured in a fyke trap that was targeting salmonids. This is the second confirmed adult green sturgeon in the San Joaquin River basin, following the adult green sturgeon in the Stanislaus River in 2017. This encounter extends the reported range of green sturgeon by ~44 river miles (RM) (Root et al. 2020). The information summarized in this section does not change the status of the species or the imminence or magnitude of any threat since the genetic data only confirm the DPS structure and add detail to the DPS composition in different estuaries during the sampling periods.

2.3.1.4 Taxonomic classification or changes in nomenclature:

There were no relevant studies examining taxonomic classification since the last status review.

2.3.1.5 Spatial distribution, trends in spatial distribution (e.g. increasingly fragmented, increased numbers of corridors, etc.), or historic range (e.g. corrections to the historical range, change in distribution of the species' within its historic range, etc.):

Work published after 2015 enhances our knowledge of North American green sturgeon spatial habitat use and distribution. As stated in the previous 5-year review, subadult (from the age of ocean entry to age of first spawning) and adult North American green sturgeon spend most of their lives in oceanic environments where they occupy nearshore coastal waters from the Bering Sea, Alaska (Colway and Stevenson 2007) to Baja California, Mexico (Rosales-Casian and Almeda-Jauregui 2009). The 2015 review indicated that North American green sturgeon are observed infrequently in Alaskan waters.

As stated in the 2015 review, telemetry data and genetic analyses suggest that Southern DPS green sturgeon generally occur from Graves Harbor, Alaska to Monterey Bay, California (Moser and Lindley 2007, Lindley et al. 2008, Lindley et al. 2011) and, within this range, most frequently occur in coastal waters of Washington, Oregon, and Vancouver Island and near San Francisco and Monterey

bays (Huff et al. 2012, Moser et al. 2016). Within the nearshore marine environment, tagging and fisheries data indicate that Northern and Southern DPS green sturgeon prefer marine waters less than 110 m deep (Erickson and Hightower 2007). Further telemetry studies performed by the WDFW and NMFS-Northwest Fisheries Science Center (NWFSC) have shown a great amount of seasonal movement between the coastal bays and estuaries and the nearshore marine environment (Heironimus 2021b).

Since the 2015 review, adult and subadult Southern DPS green sturgeon have been observed in large concentrations in the summer and fall within coastal bays and estuaries along the west coast of the United States, including the Columbia River estuary, Willapa Bay, Grays Harbor, and Humboldt Bay (Moser and Lindley 2007, Lindley et al. 2008, Lindley et al. 2011, Washington Department of Fish and Wildlife (WDFW) and Oregon Department of Fish and Wildlife (ODFW) 2014, Goldsworthy et al. 2016). These areas, particularly Willapa Bay and Humboldt Bay, are likely used for foraging and possibly as thermal refugia (Moser and Lindley 2007). The Umpqua River estuary seems to be a preferred habitat for the Northern DPS (Lindley et al. 2011, Goldsworthy et al. 2016). Recent fieldwork indicates that Southern DPS green sturgeon generally inhabit specific areas of coastal estuaries near or within deep channels or holes, moving into the upper reaches of the estuary, but rarely into freshwater (Washington Department of Fish and Wildlife (WDFW) and Oregon Department of Fish and Wildlife (ODFW) 2014). Green sturgeon in these estuaries may move into tidal flats areas, particularly at night, to feed (Dumbauld et al. 2008, Moser et al. 2017). Miller et al. (2020) recorded adult and subadult Southern DPS green sturgeon presence year-round in the Sacramento-San Joaquin Delta, Suisun Bay, San Pablo Bay, and Central San Francisco Bay. Adult Southern DPS green sturgeon were tracked by ship in the SFBDE (Kelly et al. 2007, Kelly and Klimley 2012). Individual Southern DPS green sturgeon occupied the flats during low flows and moved within the channels during high flows, generally swimming near the bottom. There is some evidence that they display ‘rheotaxis,’ gaining directional information from the flow of the water (Kelly et al. 2020). Two tagged green sturgeon in San Francisco Bay displayed selective tidal stream transport to perform daily movement between tidal regions. Rather than fighting currents, the sturgeon swam into the water column when the current was favorable. The tagged green sturgeon swam along the bottom during opposing tides (Kelly et al. 2020). Southern DPS green sturgeon display within-population level diversity in their spatial and temporal use of coastal estuaries that somewhat corresponds to the individual size of the animal (Lindley et al. 2008, Lindley et al. 2011). Green sturgeon also move extensively within an individual estuary and between different estuaries (e.g., between Willapa Bay and the Columbia River) during the same season (Moser and Lindley 2007, Washington Department of Fish and Wildlife (WDFW) and Oregon Department of Fish and Wildlife (ODFW) 2014).

Lindley et al. (2008) found that most, but not all, green sturgeon annually migrate along the continental shelf, traveling from United States waters to Canadian

waters in the fall and returning in the spring. They found that green sturgeon concentrate in the northwest Vancouver Island area from May through June and October through November. One tagged Southern DPS green sturgeon was detected in southeast Alaska, reinforcing the idea that green sturgeon rarely enter Alaskan waters (Lindley et al. 2008). Lindley et al. (2011) described the movements and summer occurrence of Northern and Southern DPS green sturgeon in estuarine and coastal areas such as, the Columbia River estuary, Willapa Bay, Grays Harbor, and the estuaries of smaller rivers in Oregon, particularly the Umpqua River estuary. Green sturgeon from different natal rivers exhibited different patterns of habitat use, with the SFBDE used only by the Southern DPS and the Umpqua River estuary used mostly by the Northern DPS. The Columbia River was visited by both Southern and Northern DPS green sturgeon. Although adults have been detected throughout the SFBDE year-round, spawning Southern DPS adults often use the SFBDE as a migration corridor, passing through within a few days of entering the SFBDE (Heublein et al. 2009, Miller et al. 2020). Subadults and non-spawning adults utilize the SFBDE in the summer for other reasons, possibly to feed, as residency periods are longer (Lindley et al. 2011, Miller et al. 2020). Recent data from telemetry studies from the NWFSC show large concentrations of tagged green sturgeon entering the coastal bays and estuaries of Oregon and Washington during the fall months with migrations into the nearshore marine environment along the Canadian coast in the winter (Smith and Huff 2020).

Section 2.3.1.1 describes current knowledge regarding spawning behavior and timing of Southern DPS green sturgeon in the Sacramento River below Keswick and Shasta dams. Whether Southern DPS green sturgeon historically spawned above Keswick and Shasta dams has been debated (Beamesderfer et al. 2005), with the original status review indicating spawning in the tributaries upstream of those dams (Adams et al. 2007). An analysis based on the habitat occupied at present versus the habitat available above the dams indicates that Southern DPS green sturgeon likely occupied areas above the dams before their construction (Mora et al. 2009, Mora et al. 2018). Adult green sturgeon have been observed in other rivers such as the lower Yuba River downstream of Daguerre Point Dam (Mora et al. 2018), and spawning was documented in the lower Yuba River by CDFW in 2018 and 2019 (Beccio 2018, 2019). Although sturgeon have been observed in the Russian River, the only known photo is of a white sturgeon. Data from angler self-reporting through the Sturgeon Report Cards distributed by CDFW indicate reports of two green sturgeon in the San Joaquin River upstream of designated critical habitat between 2016 and 2020 (See Table 1). Modeling indicates that spawning could have been supported in the San Joaquin River based on the habitat that existed in this system historically (Mora et al. 2009, Mora et al. 2018). Additionally, two confirmed green sturgeon were documented in the San Joaquin River Basin since the 2015 review (Anderson et al. 2018, Root et al. 2020).

As stated in Section 2.2.3, Demographic Recovery Criterion 2 requires the annual presence of larvae for at least 20 years in at least two rivers within Southern DPS historical range to show that green sturgeon are successfully spawning. The 2015 review noted that Southern DPS green sturgeon spawning was documented the Feather River (Seesholtz et al. 2015). Since 2015, a total of 19 green sturgeon eggs were collected in the lower Feather River using egg mats. Egg mats were placed near the Fish Barrier Dam pool (RM 67, n = 2) in 2017 and in the Thermalito Afterbay Outlet pool (RM 59, n = 17) in 2019. The eggs were found in temperatures that ranged from 13.3-19.5°C. Genetic analysis was not completed for these eggs; however, it is likely that they are genetically Southern DPS (Seesholtz 2020). In 2018, a larval sturgeon was sampled in the Feather River below the confluence of the Yuba River, which suggests that a spawning event occurred in either the Feather or Yuba River. DIDSON studies indicate that Southern DPS green sturgeon adults are present in the Feather River every year. Studies also indicate that spawning habitat in the Feather River extends from the Thermalito Afterbay Outlet pool upstream to the Fish Barrier Dam, which can only be accessed by green sturgeon in years with high basin discharge (Seesholtz 2020). In years of low basin discharge, green sturgeon are unable to access habitat upstream of Sunset Pumps rock weir (RM 38.5). A total of 14 larval green sturgeon were sampled using benthic d-nets in the lower Feather River, with 11 larvae sampled in the Fish Barrier Dam pool in 2017 and three larvae sampled near Boyd's Pump in 2018. Larvae were collected in water temperatures that ranged from 13-18°C. This evidence suggests that the Feather River is able to support periodic reproduction of green sturgeon, primarily in years with high water discharge. Continuous spawning needs to occur for at least 20 years to meet Demographic Recovery Criterion 2.

Since 2015, 402 ARIS/DIDSON sonar surveys have been performed in the lower Feather River from the Fish Barrier Dam downstream to Beer Can Beach (RM 7). The majority of the surveys occurred in four areas: Fish Barrier Dam, Thermalito Afterbay Outlet, Sunset Pumps rock weir, and Oswald (RM 23). A total of 747 images of sturgeon were recorded during these surveys. No detections above the Sunset Pumps rock weir occurred in the drier years of 2015 and 2016. In the wetter years of 2017, 2018, and 2019, green sturgeon were detected upstream of Sunset Pumps rock weir when mainstem discharge was higher, which provided sufficient flows for migration above the weir into spawning grounds. A total of 16 Southern DPS adult green sturgeon have been tagged in the lower Feather River basin since 2015. The majority of these fish were tagged in 2018 following spawning season. Since 2015, 22 previously tagged Southern DPS green sturgeon adults were detected in the lower Feather River; however, only three of the 22 were detected above Sunset Pumps rock weir. No telemetry detections occurred at Sunset Pumps rock weir in years of average or below average basin discharge. Green sturgeon detections in the lower Feather River align with the spawning season, with the average first detection occurring March 30. Following the spawning season, the majority of acoustically-detected green sturgeon leave the lower Feather River; however, a small amount of green sturgeon remain for

several months in holding areas of the river such as Oswald (Seesholtz 2020). The DIDSON and acoustic telemetry data suggest that adult green sturgeon use the Feather River for both spawning and holding areas. In years of low water, green sturgeon are restricted to habitat downstream of the Sunset Pumps rock weir, which acts as a barrier to upstream migration. Providing upstream passage at Sunset Pumps rock weir (Priority 2 Recovery Action 1a) increases access to spawning habitat for Southern DPS green sturgeon.

The 2015 review indicated that green sturgeon were observed in the lower Yuba River below the Daguerre Point Dam (11.5 RM). In order to document green sturgeon spawning in the Yuba River, egg mats were deployed in the pool downstream of Daguerre Point Dam in 2018 and 2019. Approximately 270 green sturgeon eggs were collected on a single egg mat on June 15, 2018; the eggs were vouchered for species verification and developmental staging to determine spawning date, which was June 13, 2018 (Beccio 2018). No green sturgeon eggs were collected in 2019, but an early-stage juvenile green sturgeon (40 mm FL) was observed and captured by hand in edgewater habitat 200 m downstream of Daguerre Point Dam on August 19, 2019. The spawning event likely occurred on July 19, 2019 (Beccio 2019). These spawning events suggest that the Yuba River has the potential to support periodic reproduction of Southern DPS green sturgeon. Continuous spawning needs to occur for at least 20 years to meet Demographic Recovery Criterion 2. Green sturgeon are frequently observed holding below Daguerre Point Dam, which is the current extent of anadromy for green sturgeon in the Yuba River. Greater access to spawning habitat is needed to meet Demographic Recovery Criterion 2. Providing upstream passage at Daguerre Point Dam (Priority 2 Recovery Action 1c) will increase access to spawning habitat. Assessing the water temperature and flow in the Yuba River to provide suitable conditions for spawning and rearing green sturgeon (Priority 2 Recovery Action 2c) will aid juvenile recruitment.

Schreier et al. (2016) describes preliminary evidence of green sturgeon spawning in the Columbia River based on the collection of an age-0 individual near Rooster Rock (RM 130). However, genetic analyses assigned this individual to the non-ESA listed Northern DPS rather than the listed Southern DPS. Schreier and Stevens (2020) captured four young-of-year green sturgeon downstream of Bonneville Dam on the Columbia River in 2017. Genetic analysis revealed that the fish are Northern DPS, confirming that the Northern DPS spawns in at least three rivers. Young-of-year green sturgeon were only detected in the Columbia River in times of high spring flow (e.g., 2011 and 2017)(Schreier and Stevens 2020)

Data generated since 2015 regarding the spatial occupancy of Southern DPS green sturgeon reinforces the DPS structure and the importance of coastal and estuarine habitats along the west coast of the United States. New research documents spawning by the Southern DPS in the Feather and Yuba rivers multiple years (Seesholtz et al. 2015, Beccio 2018, 2019, Seesholtz 2020), but

does not meet Demographic Recovery Criterion 2, continuous spawning for 20 years. Several Recovery Actions from the 2018 Recovery Plan directly call for barrier removal or modification in the Sacramento, Feather, and Yuba rivers so that successful spawning occurs annually in at least two rivers. Water temperature and flows in spawning and rearing areas should also allow for juvenile recruitment. While this research gives greater insight into the geographic areas occupied by the Southern DPS, the research shows that spawning in the Feather and Yuba rivers is periodic, not continuous. Given the limited number of occurrences and lack of consistent successful spawning events in additional spawning locations, this threat remains. Based on this, the new information does not support any change in species status.

2.3.1.6 Habitat or ecosystem conditions (e.g., amount, distribution, and suitability of the habitat or ecosystem):

Altered Water Flow and Temperature in Spawning Habitat: A primary concern for Southern DPS green sturgeon identified in the 2018 Recovery Plan is spawning habitat suitability in terms of water flow and temperature in the Sacramento, Yuba, and Feather rivers. Comparative analyses of historic and contemporary hydrologic and thermal regimes indicate that habitats in all of these rivers are different than they were before dam construction (see Section 2.3.2.1). What is less clear is the impact that this has had on green sturgeon spawning and recruitment. Mora et al. (2009) suggest that flow regulation has had mixed effects on habitat suitability.

Wyman et al. (2018) modeled habitat suitability for known spawning areas within the Sacramento River. Models indicated that adult Southern DPS green sturgeon are found in spawning locations that had velocities between 1.0 and 1.1 m/s, depths of 8-9 m, and gravel and sand substrate (Wyman et al. 2018). Following Wyman et al. (2018)'s model, Klimley et al. (2020) used habitat suitability modeling to better understand the relationship between population decline, habitat suitability, and the remaining available spawning habitat for Southern DPS green sturgeon. Klimley et al. (2020) used the habitat suitability of three known spawning pools in the Sacramento River and compared the amount of spawning habitat by water year. Available spawning habitat was reduced in one of the three pools during the four year drought period, from 2012 to 2015 (Klimley et al. 2020). Modeling habitat suitability of spawning habitat for Southern DPS green sturgeon should continue to be studied. It is likely that climate change and other factors will impact the habitat suitability of spawning areas for Southern DPS green sturgeon.

Barriers to Migration: The 2018 Recovery Plan listed barriers to migration as a threat to Southern DPS green sturgeon. Flood bypass systems along the Sacramento River pose a potential entrainment threat to Southern DPS green sturgeon during spawning migrations. Green sturgeon are particularly affected by Tisdale (Sutter Bypass) and Fremont (Yolo Bypass) weirs (Thomas et al. 2013, Beccio 2020, DWR 2020). Fish rescues were conducted by CDFW after

overtopping events at Fremont and Tisdale weirs during water years 2016, 2017, and 2019. Overtopping events occurred during the Southern DPS spawning migrations. CDFW rescued one adult Southern DPS green sturgeon in 2016 from the Fremont Weir stilling basin. Three adult Southern DPS green sturgeon were rescued from the Tisdale Weir stilling basin and two adult Southern DPS green sturgeon were rescued from the Fremont Weir stilling basin during water year 2017 (Beccio 2020). In water year 2019, 25 adult Southern DPS green sturgeon were rescued at Tisdale and Fremont weirs, which represents about 7.2% of the average annual spawning population, given the current average estimate of 348 individuals (Mora et al. 2018, Beccio 2020).

Although rescuing entrained sturgeon from flooded bypasses is important for population viability, fish passage improvement, rather than continued fish rescue, is a more appropriate long-term goal for mitigating this threat. Improvements to Yolo Bypass were part of required actions in the Reasonable and Prudent Alternative (RPA) within the biological and conference opinion on the long-term operations of the Central Valley Project and State Water Project (NMFS 2009a, 2011). Since the 2015 review, passage has been improved, but not completely remediated, at Fremont Weir for sturgeon and salmonids. The Fremont Weir Adult Fish Passage Project was completed in late 2018. The weir overtopped 4 times during water year 2019. This fish passage facility documented at least 76 sturgeon (80 to 200 cm TL) via Adaptive Resolution Imaging Sonar (ARIS) footage using the passage structure to access the Yolo Bypass (DWR 2020). Stranding of adult Southern DPS green sturgeon still occurred in a section of the Fremont Weir stilling basin and a deep scour pond approximately 140 m downstream of the weir (Beccio 2020). Continued improvements to Fremont Weir will occur with the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project (Big Notch Project), which is scheduled for completion in fall 2023 (DWR 2020). The Tisdale Weir Rehabilitation and Fish Passage Project (located at the Sutter Bypass) is scheduled for completion in 2023/2024 (DWR 2020). CDFW will continue to perform stranding surveys and rescue operations for adult sturgeon at Tisdale and Fremont weirs (Beccio 2020).

The decommissioning of RBDD, a barrier to migration in the Sacramento River, has increased the availability and use of upstream spawning habitat of Southern DPS green sturgeon (Thomas et al. 2014, Steel et al. 2019). Habitat modeling studies display that in the absence of impassable dams, Southern DPS green sturgeon would likely spawn in additional areas within the Sacramento River (Mora et al. 2009). This modeling work also found that suitable spawning habitat historically existed in portions of the San Joaquin, lower Feather, American, and Yuba rivers, much of which is currently inaccessible to green sturgeon due to the presence of barriers.

San Joaquin River Basin Habitat Suitability: Historic spawning habitat likely existed for Southern DPS green sturgeon within the San Joaquin River basin (Mora et al. 2009). Designated critical habitat for Southern DPS green sturgeon

does not extend past the Delta in the San Joaquin River basin (74 FR 52300). However, since the last review, green sturgeon were documented in the Stanislaus River (Anderson et al. 2018) and San Joaquin River (Root et al. 2020). In late September and early October 2017, a genetically confirmed green sturgeon was documented in the Stanislaus River holding in a pool near Knights Ferry, California, which is approximately 53 RM upstream of designated critical habitat (Anderson et al. 2018). Self-reporting angler Sturgeon Report cards from CDFW also indicate that green sturgeon have been observed within the San Joaquin River basin (DuBois and Danos 2017). In April 2020, a green sturgeon was observed within the bounds of the San Joaquin River Restoration Program area of the San Joaquin River, which is approximately 44 RM upstream of designated critical habitat (Root et al. 2020). The fish was captured in a fyke trap 300 m upstream of the confluence of the Merced and San Joaquin rivers near Hills Ferry, California (Root et al. 2020). The green sturgeon was confirmed by trained fish biologists who evaluated external meristic characteristics. It should be noted that the 141.8 cm TL fish was documented in the San Joaquin River basin during the spawning period for Southern DPS green sturgeon (Root et al. 2020). Monitoring efforts for green sturgeon within the San Joaquin River basin should be increased to further understand their habitat extent.

Altered Prey Base: The 2018 Recovery Plan listed altered prey base as threat to Southern DPS green sturgeon throughout its critical habitat. The 2015 review stated that two issues may affect prey resources for Southern DPS green sturgeon in coastal bays and estuaries (WDFW and ODFW 2014). The first issue was the increasing presence of Japanese eelgrass (*Zostera japonica*) in tidal areas including sturgeon feeding pits, which would negatively impact the habitat of burrowing shrimp, a common prey item of green sturgeon in estuaries. Moser et al. (2017) found a negative correlation between green sturgeon presence in feeding pits and the presence of Japanese eelgrass, indicating that sturgeon may have difficulty feeding in substrate that has been invaded by Japanese eelgrass. Since 2015, the presence of Japanese eelgrass has been decreasing in the upper intertidal mudflats in coastal estuaries of Northern California, Oregon, and Washington (Goldsworthy 2020). Information is not yet available regarding the impacts of these changes on green sturgeon. An invasive, parasitic isopod that castrates the native blue mud shrimp (*Upogebia pugettensis*) (Dumbauld et al. 2008, Dumbauld et al. 2011, Chapman et al. 2012) has affected the availability of this green sturgeon prey resource in west coast estuaries.

Summary: New information on Southern DPS habitat indicates that the Southern DPS still face threats posed by impassable barriers and flood bypass systems. The decommissioning of RBDD has, however, resulted in additional spawning habitat availability and utilization. Fish passage improvement at Fremont Weir has reduced entrainment into the Yolo Bypass but fish rescues still need to occur. Continued fish passage improvement at flood bypasses will help reduce entrainment of Southern DPS green sturgeon. Hydrological and thermal regimes in spawning habitats are altered as compared to historic profiles, which could

impact recruitment and recovery (see Section 2.3.2.1). New documentation of green sturgeon in the San Joaquin River basin has increased the need for monitoring green sturgeon outside of its designated critical habitat. Invasive species may impact Southern DPS prey resources in coastal estuaries. Overall, the new information does not provide conclusive data indicating that habitat conditions and factors have changed in severity or degree of threat since 2006, since additional research is needed.

2.3.1.7 Other:

None.

2.3.2 Five-Factor Analysis (threats, conservation measures, and regulatory mechanisms)

2.3.2.1 Present or threatened destruction, modification or curtailment of its habitat or range:

Barriers to Migration and Spawning Habitat: The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline in the DPS is the reduction of spawning to a limited area in the Sacramento River (71 FR 17757; April 7, 2006). Keswick Dam on the Sacramento River and Oroville Dam on the Feather River were noted as impassable barriers (71 FR 17757). No change in the status of these dams has occurred since 2006. Potential barriers to adult migration also included RBDD, Sacramento Deep Water Ship Channel locks, Fremont Weir (Yolo Bypass), Tisdale Weir (Sutter Bypass), the Anderson Cottonwood Irrigation District (ACID) dam and the Delta Cross Channel Gates on the Sacramento River, and Shanghai Bend and Sunset Pumps rock weir on the Feather River (BRT 2005, 71 FR 17757). The Fish Barrier Dam on the Feather River and the Daguerre Point Dam on the lower Yuba River are also recognized as limiting the distribution of the Southern DPS (74 FR 52300; October 9, 2009).

Since the listing decision, some improvements to impassable barriers have occurred. Two barriers originally cited in the listing decision as posing a limit to distribution have improved since the listing: RBDD (RM 243) on the Sacramento River and Shanghai Bend on the Feather River. The decommissioning of RBDD in 2011 now permits passage of the Southern DPS during all months that they are present in the river. The breach of Shanghai Bend on the Feather River in early 2012 likely eliminated this naturally-formed passage barrier (flow-dependent) in the lower Feather River (NMFS 2015). Structural modifications at Fremont Weir (Yolo Bypass) improved passage for salmonids and sturgeon to move from the Yolo Bypass back into the Sacramento River with the first facility opening in 2019. The project permitted the passage of over 70 sturgeon during overtopping events (DWR 2020). However, three more fish passage facilities are planned at the Fremont Weir to prevent the need for fish rescue at the site. There are also plans to improve fish passage at Sutter Bypass (Tisdale Weir) and Sacramento Bypass (Sacramento Weir). On the Feather River, Sunset Pumps rock weir removal was funded by the Central Valley Project Improvement Act in 2018; however, as of this review, the project has not been initiated. Although some

barriers to migration have been improved or removed, there are still many migration barriers within the Sacramento River basin. Through the development of fish passage criteria for sturgeon, many more of these barriers can be improved or removed from Southern DPS green sturgeon critical habitat.

Altered Water Flow and Temperature in Spawning Habitat: Temperature and flow have been shown to be relevant parameters with respect to spawning, survival, and growth of North American green sturgeon (see Section 2.3.1). In the Sacramento River, the Proposed Action for the long-term operations of the Central Valley Project and State Water Project (USBR 2019) includes maintenance of 11.9°C (53.5°F) water temperature from May 15 to October 31 at a compliance point above the confluence of the Sacramento River and Clear Creek in years when the U.S. Bureau of Reclamation determines that the Shasta Reservoir cold water pool is sufficient. In years when the cold water is insufficient, a tiered system attempt to optimize the use of cold water for Sacramento River winter-run Chinook salmon will be used. However, during 2014 and 2015, the lack of cold water stored behind Shasta Dam, in combination with water release decisions, led to a high water temperatures below Shasta Dam. Additional temperature requirements are mandated by the California State Water Resource Control Board Water Rights Orders 90-05 and 91-01. The CALFED Science Review Panel determined that temperatures associated with the Clear Creek compliance point may reduce the growth rate of larvae and post-larvae sturgeon relative to warmer temperatures (CALFED 2009). Balanced water temperature management may still be possible in all but the most severe drought years since the water warms as it flows downstream from the release point to the Southern DPS green sturgeon spawning sites, and given the presence of the larval green sturgeon primarily in May, before the peak agricultural water demand in June and July (Zarri et al. 2019).

As stated in the 2015 review, Mayfield and Cech (2004) reported optimal bio-energetic performance of age 0 and age 1 Northern DPS green sturgeon from 15 to 19°C in laboratory conditions. Summer water temperatures in the upper Sacramento River have typically been below this range, generally below 16℃ (Goto et al. 2015). The2015 review noted that the water temperature target of 13.3℃ was not maintained in the Sacramento River during periods of 2014 and 2015 due to the historic drought. Water temperatures increased throughout the Southern DPS green sturgeon spawning range in the Sacramento River. Summer flows also decreased as a result of the drought conditions. The effects of these water temperature and summer flow changes in the Sacramento River on survival and recruitment of green sturgeon remains unclear.

Goto et al. (2015) suggested that the managed temperature may inhibit growth rates of green sturgeon making juvenile green sturgeon more susceptible to size-dependent predation. Larval green sturgeon reared in the laboratory exhibited reduced growth at lower temperatures (i.e., 11℃ vs. 19℃) even with optimal feeding (Poletto et al. 2018). A diet study of wild larval Southern DPS green

sturgeon observed that there were more empty stomachs at colder temperatures, which may indicate that foraging activities or greater food availability occur during warmer temperatures (Zarri and Palkovacs 2019). Hamda et al. (2019) developed a model to analyze the impact of different conditions in the Sacramento River (i.e., water temperature, flow, food availability) on the growth of juvenile green sturgeon. Since water temperature and flow are managed in the Sacramento River to allow for egg-to-fry survival for winter-run Chinook salmon during the same time of Southern DPS green sturgeon spawning, egg incubation, and juvenile rearing, the impact of cooler temperatures on juvenile green sturgeon growth rate was analyzed (Hamda et al. 2019). The model showed that although there is a trend for cooler temperatures to negatively impact growth rates of juvenile green sturgeon, the impact is relatively small. However, in years of warmer water temperatures, the trade-off is much greater for winter-run Chinook salmon with a large reduction in egg-to-fry survival (Hamda et al. 2019). The impact of climate change will be discussed more in Section 2.3.2.5. Further green sturgeon monitoring may produce juvenile year class indices to compare effects of water temperature and flow on recruitment in the future.

Summary: In summary, the available information generated since the 2015 review indicates that impassable barriers still pose high threat to Southern DPS green sturgeon, although the threat is reduced with the decommissioning of RBDD and passage improvement at Fremont Weir. With the decommissioning of RBDD, Southern DPS green sturgeon are spawning in greater numbers in higher reaches and larvae are now rearing in the area influenced by the cooler water temperatures closer to the temperature compliance point. That said, the compliance point has not been consistently maintained and summer flows were reduced during the 2014 and 2015 drought. The effect of the temperature on Southern DPS production during the drought remains unclear. Although the first successful season of direct juvenile green sturgeon sampling by trawling near RBDD occurred during elevated temperatures in 2015, juveniles were subsequently collected from sampling efforts from 2016 to 2020 (NMFS 2018, Poytress 2020). Furthermore, high larval Southern DPS green sturgeon catch at RBDD has occurred in years with relatively low water temperatures (1995, 2011, 2016, 2017, and 2019; NMFS 2018, Poytress 2020). The effect of cold water releases from Keswick Dam may have a greater impact on Southern DPS green sturgeon spawning and incubation in the uppermost accessible reach of the Sacramento River below the ACID. If spawning occurs immediately downstream of the ACID, then the water will not warm to more suitable temperatures for eggs and larvae as it will at spawning locations further downstream near RBDD. The ACID is a migration barrier for Southern DPS green sturgeon, but low water temperature could deter Southern DPS spawning even if passage was restored to this reach (NMFS 2019). Klimley et al. (2020) noted that over the drought (2012 to 2015), spawning habitat was reduced in one of three sites in the Sacramento River. The reduction in available spawning habitat may impact Southern DPS green sturgeon occupation in these areas (Klimley et al. 2020). Laboratory, modeling, and field studies continue to look at the impact of flow and temperature regimes on spawning and recruitment

of the Southern DPS. Given the present data, there is no evidence that the threat posed by modification of habitat has increased in severity since 2015.

2.3.2.2 Overutilization for commercial, recreational, scientific, or educational purposes:

In the final rule, past and present commercial and recreational fishing as well as poaching, were recognized as factors that pose a threat to the Southern DPS (71 FR 17757; April 7, 2006). No estimate of an annual rate of mortality due to poaching has become available since the last review. As stated in the previous review, the threat posed by commercial and recreational fishing has decreased since 2006 given that intentional lethal take of green sturgeon has been prohibited through fishing regulations. Regulations prohibit retention of green sturgeon in California, Oregon, and Washington state fisheries and in federal fisheries in the United States and Canada. These regulations pertain to the range of both Southern and Northern DPS green sturgeon to address the possibility of capture of the threatened Southern DPS throughout the coast.

Retention of North American green sturgeon is not currently permitted in any state fishery. The CDFW, WDFW, and ODFW all prohibit the commercial retention and sale of green sturgeon. The CDFW further prohibits take of any sturgeon (white or green) in the Sacramento River between Keswick Dam to the Highway 162 Bridge in order to protect spawning green sturgeon (CCR, Title 14, Sec. 5.80, 5.81).

State officials performed observations of commercial fisheries in the lower Columbia River and Grays Harbor and Willapa Bay estuaries to detect rates of encounters with green sturgeon. Encounters occurred mostly in the summer/fall period. From 2014 to 2019, observers encountered 5 to 24 green sturgeon per year in the Willapa Bay commercial salmon fishery (Observer coverage ranged from 15-29% per year; (Heironimus 2020b)). In Grays Harbor, observers have not recorded any catch of green sturgeon in commercial fisheries since 2014(Heironimus 2020a). For the lower Columbia River, commercial fisheries encounter up to an estimated 252 Southern DPS green sturgeon per year (NMFS 2018). No error range was provided with this point estimate. These estimates are likely to be overestimates because they are based on historical catch numbers; additional catch data are needed to refine these estimates.

Agency statistics from self-reporting and observation provide additional information about North American green sturgeon encounters in recreational fisheries in Washington and Oregon. Based on ODFW creel surveys from 2007 to 2019, the number of green sturgeon caught and released in the lower Columbia River ranged from 13 to 255 per year; no green sturgeon were reported as caught and released in 2014 (Stevens 2020). A small number of green sturgeon (≤10) are still annually retained in this fishery due to misidentification (WDFW and ODFW 2014).This number is far fewer than the number of animals that were retained before retention was prohibited in 2007 (up to 533 individuals in 1985). Of the

green sturgeon encountered, the majority would be expected to be Southern DPS green sturgeon based on Israel et al. (2009).

Outside of the Columbia River, anglers in Oregon reported catch and release of 6 to 120 green sturgeon per year in 2007 through 2018 (ODFW Sport Catch Statistics 2007-2018 accessed at www.dfw.state.or.us/resources/fishing/sportcatch.asp). No green sturgeon catch was reported in years 2011, 2012, 2013, and 2017 in areas outside of the lower Columbia River. The previous review stated that recreational fisheries outside of the Columbia River in Washington may encounter up to 64 Southern DPS green sturgeon annually (NMFS 2015). No green sturgeon have been reported in Washington coastal and Puget Sound recreational fisheries (outside of Willapa Bay and Grays Harbor) since the 2007 closure to retention, although anglers are only required to report fish kept, not those released.

Southern DPS green sturgeon are encountered annually by California recreational fishers based on self-reporting and creel data. Table 1 summarizes data from sturgeon report cards submitted annually by anglers. From 2007 to 2017, the number of green sturgeon caught and released ranged from 89 to 311 per year, the average fork length ranged from 29 to 40 inches, and the main areas where green sturgeon were encountered were the Sacramento River from Rio Vista to Chipps Island, as well as Suisun Bay (Table 1). Creel surveys conducted in recreational fisheries also report green sturgeon encounters. California commercial passenger fishing vessels (CPFV) report encounters with sturgeons, but have not recorded sturgeon to the species level in the past. CPFV operators were instructed to record sturgeon to the species level in 2012. As of this review, no data was available from CPFV reports.

Table 1. Information collected through CDFW sturgeon report cards. ND = no data available. Data sources: Gleason *et al.* 2008; Dubois *et al.* 2009-2012, 2014; Dubois 2013; DuBois and Harris 2015 and 2016; DuBois and Danos 2017, 2018.

Year	# Cards Issued[1]	# Cards Returned[1]	# Cards with Sturgeon Recorded[1]	# Green Sturgeon Released	Average Fork Length of Green Sturgeon Measured at Release	Main Areas Encountered
2007[2]	37,680	6,919	1,855	311	37 inches	Sac. River Red Bluff to Colusa, Rio Vista to Chipps Island
2008	53,777	7,329	2,048	240	31.6 inches	Sac. River Red Bluff to Colusa, Rio Vista to Chipps Island

2009	72,499	8,558	2,208	215	29 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2010	66,357	7,515	1,758	151	40 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2011	112,000	12,413	2,274	89	31.3 inches	San Pablo Bay, Suisun Bay
2012	112,800	12,637	2,052	175	36 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2013	50,915	10,642	2,290	168	32 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2014	49,260	12,076	2,645	154	32 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2015	48,337	14,382	2,870	192	30 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2016	47,617	15,674	2,997	220	33 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2017	44,374	14,872	2,814	230	33 inches	Sac. River Rio Vista to Chipps Island, Suisun Bay
2018[3]	44,146	14,368	2,398	ND	ND	ND
2019[3]	40,844	12,596	1,742	ND	ND	ND

[1] Updated data obtained from DuBois et al. 2020 for the number of cards issued, returned, and with sturgeon recorded.
[2] Note that 2007 data are not from the entire year since the report card program started that year and cards were first issued in February 2007.
[3] For 2018 and 2019, data are not available on the number of green sturgeon released and the average fork length of measured green sturgeon. Data on the number of sturgeon report cards issued, released, and with sturgeon recorded were obtained from DuBois et al. 2020.

Both Southern and Northern DPS green sturgeon are encountered in the state-regulated California halibut bottom trawl fishery in coastal marine waters. From 2002 through 2017, a range of 29 to 786 green sturgeon encounters occurred per year in the fishery (Richerson et al. 2019). The majority of the green sturgeon

encountered likely belonged to the Southern DPS, based on the location of the encounters (primarily in coastal marine waters adjacent to San Francisco Bay) (Richerson et al. 2019) and genetic data (Anderson et al. 2017). In 2017, one green sturgeon was caught in the California nearshore fixed-gear fishery near the mouth of San Francisco Bay, using hook and line gear (Richerson et al. 2019). The fish dropped off the line before it could be brought on board, so no data were collected on the fish size, weight, and condition, but the fish was at the surface long enough for visual identification. This was the first time green sturgeon bycatch has been observed in the California nearshore fixed-gear fishery since observer coverage began in 2002.

The 2015 review stated that in Alaska, North American green sturgeon is listed as a "nominee" species in the State of Alaska Wildlife Action Plan and designated as a "Species of Greatest Conservation Need" under the Aquatic Habitat Implementation Plan, which is part of the Comprehensive Wildlife Conservation Strategy. The Alaska Department of Fish and Game (ADFG) indicates that information about green sturgeon is limited to a few anecdotal reports of sightings and captures in Alaska waters, mostly in Alaska District 8 and District 11 (encompassing the mouths of the Stikine and Taku, respectively) driftnet fisheries. ADFG has received no reports of regular sightings of sturgeon. The North Pacific Groundfish Observer Program, which observes Federal groundfish fisheries off Alaska, has recorded rare encounters with green sturgeon in trawl fisheries in the Bering Sea (1982:1; 1984:2; 2005:1; 2006:3; 2009:1; 2012:1; 2013:1; 2015:1; NPGOP data received April 2015). It is unknown whether the green sturgeon encountered belonged to the Northern DPS or the Southern DPS.

In Canada, North American green sturgeon are occasionally encountered by commercial bottom trawlers, with most catches off the north or southwest ends of Vancouver Island. The species is also encountered at low rates in commercial hook-and-line fisheries, in the recreational white sturgeon fishery in the lower Fraser River, and in commercial salmon gillnet and seine fisheries in the Fraser River as well as at the mouths of other rivers (Fisheries and Oceans Canada 2017). Green sturgeon is listed as a species of Special Concern under Canada's Species at Risk Act (SARA) and is protected by the federal Fisheries Act, which prohibits destruction of fish habitat. A Management Plan for the species is required under the SARA and was finalized in 2017 (Fisheries and Oceans Canada 2017).

As summarized in the 2015 review, Canada prohibits retention of North American green sturgeon in recreational and commercial fisheries, and all commercial fisheries are required to release bycatch at sea with the least possible harm. The commercial groundfish bottom trawl fishery has 100% at-sea observer coverage, while the commercial hook and line/trap groundfish fisheries have 100% at-sea monitoring as either observers or electronic monitoring. Dockside monitoring is also in place for groundfish (i.e., groundfish trawl, rockfish hook and line, sablefish, halibut, lingcod and dogfish). This monitoring, in addition to logbooks,

enables more accurate accounting of green sturgeon bycatch in these fisheries. Food, social and ceremonial First Nations fisheries may retain green sturgeon if they are encountered. No capture statistics are available for these fisheries.

Canadian fisheries closures established to protect large areas of significant bottom habitat (e.g., rockfish conservation areas and groundfish bottom trawl closures) also serve to protect some North American green sturgeon habitat. Additionally, standard operating practices for industries and regulatory agencies with authority in the Fraser River have been developed to mitigate impacts to freshwater habitat for green sturgeon.

Take of Southern DPS green sturgeon in Federal fisheries was prohibited as a result of the ESA 4(d) protective regulations (ESA 4(d) Rule) issued in 2010 (75 FR 30714; June 2, 2010). Northern and Southern DPS green sturgeon are, however, incidentally encountered in the west coast Pacific Groundfish fisheries, including the Limited Entry (LE) groundfish bottom trawl sector and the at-sea Pacific hake/whiting sector (at-sea hake sector) (Richerson et al. 2019). Incidental catch of green sturgeon in these fisheries has varied over the years. The LE groundfish bottom trawl sector encountered an estimated 0 to 43 green sturgeon per year from 2002 through 2017 (Richerson et al. 2019). Based on the location of the encounters and data on green sturgeon stock composition in marine and coastal estuarine waters, the majority of the green sturgeon encountered likely belonged to the Southern DPS (Anderson et al. 2017). Most of the fish were released alive. In the at-sea hake sector, three green sturgeon were encountered from 1991 through 2017 and all had died (Richerson et al. 2019). Data are not available on whether the fish belonged to the Southern DPS or Northern DPS.

Assessing the potential effects of fisheries bycatch on Southern DPS green sturgeon in commercial and recreational fisheries requires an understanding of bycatch mortality using different gear types. While immediate mortality can be more directly measured and detected and is expected to be low, some delayed mortality may occur. The issue of delayed, post-release mortality requires further study. Robichaud et al. (2006) estimates post-release mortality at 5.2% in commercial gillnet fisheries and 2.6% in recreational hook-and-line fisheries. A satellite tagging study in collaboration with the California halibut fishery estimated post-release mortality for green sturgeon to be 18% (95% CI: 6-28% mortality) (Doukakis et al. 2020). Green sturgeon are also episodically encountered and potentially injured in traps used for the Dungeness Crab fishery along the California Coast (Goldsworthy 2017). Efforts made by state and federal agencies to monitor, minimize, and evaluate the effects of fisheries capture of green sturgeon are ongoing. Studies to better understand the circumstances under which bycatch mortality increases are needed to guide fishery management efforts.

Outreach by CDFW, ODFW, and WDFW has been undertaken regarding North American green sturgeon catch and handling regulations. State commercial and

sport fishing rules pamphlets indicate prohibitions on green sturgeon retention. These regulations, as well as posters at boat launch and bank fishing sites also offer information on distinguishing between green and white sturgeon. WDFW requires commercial gillnet fishers in Willapa Bay and Grays Harbor to report all green sturgeon encounters. In 2012, WDFW also deployed onboard commercial fishing vessel monitoring. All fishermen in the Willapa Bay and Grays Harbor region must attend a Fish Friendly Best Fishing Practices class. Monitoring of commercial fisheries in the Columbia River has occurred annually since 2002 and has increased in scope in recent years. Since January 2004, the California Halibut trawl fishery has carried federal observers who record all green sturgeon encounters, although coverage rates have been fairly limited (Richerson et al. 2019). The Pacific groundfish fisheries are observed at higher rates and data indicate fewer encounters with green sturgeon as compared to the California Halibut fishery (Richerson et al. 2019).

The ESA 4(d) Rule provides an exemption from take prohibitions (not retention) for Southern DPS green sturgeon for commercial and recreational fisheries if those fisheries activities are conducted in accordance with a NMFS-approved Fishery Management and Evaluation Plan (FMEP) (75 FR 30714; June 2, 2010). The FMEP has nine required elements, including setting maximum incidental take levels that will not reduce survival or recovery of the Southern DPS, effective monitoring and evaluation planning, enforcement and education, and reporting of the amount of incidental take on a biannual basis (75 FR 30714; June 2, 2010). Washington has submitted a draft FMEP, which has undergone NMFS review and comment. We will continue to work with Washington to address our comments and finalize the FMEP. We have also coordinated with Oregon and California on development of FMEPs. California is drafting an FMEP and Oregon may submit a plan in the future. As the states develop their FEMPs, we continue to work with them to assess and address green sturgeon bycatch. For example, we have worked with WDFW to monitor green sturgeon habitat use in coastal bays and estuaries to inform fisheries management decisions to minimize bycatch of green sturgeon. We are also working closely with CDFW to evaluate bycatch effects on green sturgeon and develop new ways to reduce bycatch in the California halibut bottom trawl fishery. These efforts will inform the development of the FMEPs.

Since the ESA 4(d) Rule was promulgated in 2010 (75 FR 30714; June 2, 2010), take for scientific purposes has been managed by NMFS under the ESA 4(d) research program and ESA Section 10(a)(1)(A) permits. Authorized take (not retention) of Southern DPS green sturgeon for scientific purposes has been tracked since 2006. In reviewing projects involving Southern DPS green sturgeon, NMFS seeks to minimize the impact of scientific research and maximize the benefits to the species. A protocol for sturgeon research developed by NMFS provides guidelines for all scientific research that involves Southern DPS green sturgeon (Kahn and Mohead 2010). The protocol's recommendations are designed to minimize stress and potential mortality to sturgeon due to research activities.

In summary, the level of lethal take of Southern DPS green sturgeon has decreased because of state and federal regulations that prohibit their retention in almost all fisheries. Lethal take still occurs as a result of bycatch mortality and a limited number of permitted activities. The impact of lethal take on the overall population abundance of Southern DPS is still being assessed. No estimate of an annual rate of mortality due to poaching has become available since the 2015 review, and therefore, it is still considered a threat.

2.3.2.3 Disease or predation:

Disease was not recognized as a principle factor in listing the Southern DPS due to a lack of sufficient information. Recently, *Chryseobacterium* spp. was detected in one green sturgeon in California waters; however, no potential negative effects of the bacteria were observed (Sebastiao et al. 2019). No new information has become available that changes the conclusion that disease is a principal factor since the listing.

Predation by introduced species was recognized as a possible threat to long-term survival of the Southern DPS. Baird et al. (2020) described the predation rate of juvenile Northern DPS green sturgeon in a laboratory setting using two species introduced to the SFBDE: striped bass and channel catfish. Compared to alternate prey, the predators showed much lower rates of predation on juvenile green sturgeon. Predation decreased with green sturgeon size, and reached zero when green sturgeon were 20-22 cm TL. The predators exhibited low motivation to feed on juvenile green sturgeon (Baird et al. 2020).

Steller sea lions (*Eumetopias jubatus*) are known to feed on sturgeon in the Columbia River. Observations by the U.S. Army Corps of Engineers have recorded only white sturgeon being consumed (Tidwell et al. 2019, Tidwell et al. 2020). In 2009, however, a photograph of a sea lion eating a green sturgeon was taken in the Rogue River. Researchers in Washington and Oregon have also reported puncture wounds and scrapes on North American green sturgeon consistent with pinniped attacks. The previous review stated that CDFW noted predation on Southern DPS green sturgeon by California sea lions (*Zalophus californianus*) in the Sacramento River, bays and Delta. Steller and California sea lion abundance has increased in recent decades (NMFS 2013, Carretta et al. 2018). Southern DPS green sturgeon was collected from the State Water Project's Skinner Delta Fish Protection Facility after a California sea lion attack (Miranda 2018). WDFW has also observed markings on North American green sturgeon that could be consistent with shark attack. A North American green sturgeon was identified in the stomach contents of a white shark captured off Central California (Klimley 1985). The impact of predation on adult and subadult North American green sturgeon is unknown. Although sea lion abundance has increased, there is no new information to support that the threat of predation by sea lions or sharks has changed in severity since the last review.

2.3.2.4 Inadequacy of existing regulatory mechanisms:

The final rule concluded that inadequacy of existing regulatory mechanisms has significantly contributed to the decline of the Southern DPS and to the severity of threats that the species currently faces (71 FR 17757; April 7, 2006). Although there have been improvements to fishing regulations to eliminate harvest and reduce bycatch mortality, and some passage barriers have been removed, less has been accomplished through regulatory mechanisms to reduce other threats (i.e., those posed by still existing migration barriers, water diversions and management). As such, inadequacy of existing regulatory mechanisms regarding Southern DPS green sturgeon habitat remains a high threat.

The 2018 Recovery Plan described regulatory mechanisms that need to be established or improved for Listing Factors A through C and E. For Listing Factor A, regulatory mechanisms need to address sturgeon passage improvement at outstanding barriers to migration (e.g., Sunset Pumps rock weir on the Feather River, Daguerre Point Dam on the Yuba River). Additionally, there should be modifications at impoundment operations and facilities to address flow, water temperature, and sediment impacts for Southern DPS green sturgeon (e.g., Oroville-Thermalito Complex, Keswick Reservoir, Shasta Lake). Lock and gate operations at the Port of Sacramento and Delta Cross Channel also need to be improved. For Listing Factor B, poaching and fishing regulations (e.g., bycatch in state fisheries) need to be enforced. For Listing Factor C, invasive species (e.g., overbite clam) in the SFBDE and Coastal Bays and Estuaries (CBE) need to be controlled. For Listing Factor E, land use regulations for non-point and point source contaminants in the Sacramento River basin and SFBDE need to be enforced. Additionally, response plans for oil and chemical spill in the SFBDE and CBE need to be developed, and offshore and nearshore kinetic energy projects in the CBE and nearshore marine habitat need to be permitted.

Another regulatory mechanism that needs to be established is the development of screen criteria and operations guidelines for agricultural, municipal, and industrial water diversions in the Sacramento River basin and SFBDE. In 2019 and 2020, an interagency group was formed to create a study that would inform screening and passage criteria for sturgeon on the west coast. The group consists of NMFS engineers and biologists, CDFW engineers and biologists, DWR biologists, USACE biologists, and University of California at Davis (UCD) researchers. The Sturgeon Passage Study Plan consists of several modules that will inform criteria for all life stages of green and white sturgeon (Fangue Fish Ecophysiology Laboratory and J. Amorocho Hydraulic Research Laboratory 2020). Modules from this study plan need to be funded to develop screening and passage criteria for Southern DPS green sturgeon.

The Green Sturgeon Habitat, Mitigation, and Monitoring Plan (GS HMMP) team formed to help the USACE implement Southern DPS green sturgeon commitments in the American River Watershed Common Features, West Sacramento, and Sacramento Bank Protection Project biological opinions. These

levee projects will likely impact the migratory corridor for several life stages of Southern DPS green sturgeon in the Sacramento River. The GS HMMP team is composed of the USACE, NMFS, and other cooperative agencies as well as green sturgeon experts. The USACE will fund, plan, and implement actions that benefit Southern DPS green sturgeon. This includes a revision of the Standard Assessment Methodology model for green sturgeon, monitoring activities, and mitigation site selection. This strategic implementation plan seeks to ensure that adverse impacts of future bank protection projects on Southern DPS green sturgeon are sufficiently mitigated to allow for the growth, survival, and recovery of the species in areas impacted by USACE projects.

Conservation banks can provide an efficient and effective mechanism to offset adverse effects on listed species and critical habitat and contribute to their conservation. A conservation bank is typically a site that provides ecological functions and services and is conserved and managed in perpetuity for a particular species. Once established, they are later used to compensate for impacts occurring elsewhere to the same species. The value of a bank is defined in compensatory mitigation credits. Credits generated by the bank site are sold by the bank sponsor to parties that need to compensate for the adverse effects of their activities and/or to contribute to the conservation of protected species and their habitats. Compensatory mitigation credits for Southern DPS green sturgeon were not available during the 2015 review. As of this review, a limited amount of Southern DPS green sturgeon credits are available at several conservation banks in the Sacramento River and Delta. Existing conservation banks which provide ecological functional uplift to green sturgeon should develop green sturgeon compensatory mitigation credits to allow project proponents to offset adverse effects to the species. Additionally, new conservation banks should be developed to offer compensatory mitigation credits for habitat functions and services specific to the recovery priorities of green sturgeon.

As stated above in Section 2.3.2.2, the states of California, Oregon, and Washington have enacted regulations to prohibit retention of North American green sturgeon in all commercial and recreational fisheries. Canada has similar regulations in place.

2.3.2.5 Other natural or manmade factors affecting its continued existence:

Non-native species: The final rule did not recognize this as a primary factor in the decline of the Southern DPS. No new data are available on risks posed by non-native species.

Ship Strikes: In early 2020, an interagency team was formed to better understand sturgeon mortality associated with ship strikes. The team is composed of representatives from NMFS, SWFSC, USBR, CDFW, DWR, Delta Stewardship Council, and Cramer Fish Sciences. In April 2018, a white sturgeon mortality from a ship strike was documented in Carquinez Strait (Demetras et al. 2020). Since this reported mortality, other mortality events reported by concerned

citizens and local fisheries biologists have occurred. The reports are primarily concentrated in Carquinez Strait, which is a known sturgeon feeding ground and vital migratory corridor to access spawning habitat. The three objectives of the team are: (1) to estimate sturgeon mortality rates in the SFBDE; (2) determine pre-mortality population characteristics, migratory behavior, and habitat use from sampled carcasses; and (3) investigate public perception of sturgeon mortality in the SFBDE. A formal reporting system was introduced in July 2020 with the placement of fliers around the Carquinez Strait asking the public to report any observed sturgeon carcasses to CASturgeonResearch@gmail.com, later changed to CASturgeon.Research@noaa.gov. Along with the report, the public was asked to include photos with a known object for size reference, exact latitude and longitude. As of February 2021, the group has received reports of 23 sturgeon carcasses in the Carquinez Strait from members of the public. Following the reports, scientist collected samples from the reported carcasses. Samples included tissue for genetic confirmation and fin rays to reconstruct age, growth, and migratory history of sturgeon in the SFBDE. These results are not yet available.

Juvenile Entrainment at Unscreened Diversions: In the final rule, the threat posed by juvenile entrainment to the continued existence of the Southern DPS was considered to be uncertain. As stated in the 2015 review, thousands of diversions exist in the Sacramento River and Delta that could potentially entrain Southern DPS green sturgeon (Mussen et al. 2014). Data on entrainment of Southern DPS green sturgeon is limited. Many large diversions (250 cfs and higher) have been screened and projects are planned for screening some smaller diversions (up to 250 cfs) (Vogel 2013). The effectiveness and impact of screening for green sturgeon requires further study given that screen criteria are currently designed to reduce salmon entrainment and impingement (NMFS 2015). For example, Southern DPS green sturgeon spawn upstream and downstream of the Red Bluff Pumping Plant (Poytress et al. 2015), which operates utilizing Chinook salmon screening criteria. Though the diversion facility meets NMFS's screening criteria for salmon, the impact on larval or juvenile Southern DPS green sturgeon that pass this site during some of the highest diversion rates is unknown, and evaluation of screening criteria in regard to green sturgeon is needed. Experimental modules developed by UCD (see Section 2.3.2.4) will inform screening criteria for larval and juvenile green sturgeon. These studies need to be funded to provide screening and operations criteria for sturgeon on the west coast.

As noted in the 2015 review, laboratory experiments indicate that young-of-year juvenile green sturgeon contact screens and become impinged upon them more frequently than similarly-sized white sturgeon (Poletto et al. 2014a). Deterrent treatments (e.g., acoustic vibrations, strobe lights) did not reduce the number of impingements for either species (Poletto et al. 2014a). The long-term impact of repeated impingement has not been studied. Young-of-year juvenile green sturgeon are highly vulnerable to entrainment through unscreened diversion pipes, and water diversion rates impact entrainment with lower diversion rates resulting in lower entrainment rates (Mussen et al. 2014). Poletto et al. (2014b) found that

when sturgeon were exposed to sweeping velocities and diversion rates similar to operation flows that entrainment rates were significantly reduced with the use of terminal pipe plates and upturned pipe plates. Strobe lights did not significantly reduced entrainment rates for sturgeon in these scenarios. Poletto et al. (2014b) recommended installation of terminal pipe plates as the more feasible way to reduce entrainment. Further study is needed to understand changes associated with ontogeny and to define conditions where fish are most susceptible, so as to better apply the findings to conservation of the Southern DPS within the river and estuary environment.

A better understanding of the threat posed by unscreened diversions could be gathered by comparing when and where vulnerable stages of Southern DPS green sturgeon (e.g., eggs or newly emerged) occur in the river with the location and operation of unscreened diversions that may be diverting at critical locations during critical periods. Limited field data exist on entrainment of the Southern DPS green sturgeon in unscreened diversions. See the 2015 review for further information regarding field data. A laboratory flow study recommends that water diversion flows should be limited to 29 cm/s at water diversion structures in the upper and middle reaches of the Sacramento River from May through the summer (Verhille et al. 2014). In the middle reaches of the Sacramento River, the maximal velocity should be 54 cm/s during the night from July until the following May. During October and November, maximal diversion velocities should not exceed 40 cm/s in the middle and lower reaches of the Sacramento River and the Delta and Bays (Verhille et al. 2014).

As stated in the 2015 review, entrainment rates of green sturgeon at Tracy Fish Collection Facility and the Skinner Delta Fish Protective Facility in the South Delta have decreased since 1986 (Adams et al. 2002, BRT 2005, Adams et al. 2007). Salvage data from the Tracy Fish Collection Facility and Skinner Delta Fish Protective Facility for the period from January 2016 to July 2020 indicates that few Southern DPS green sturgeon are encountered at the facilities. Southern DPS green sturgeon encounters by year are as follows (observed number, estimated number salvaged): 2016: 1, 4; 2017: 1, 4; 2020: 2, 8 (DWR 2020 database https://apps.wildlife.ca.gov/Salvage/). In comparison to the 2015 review, salvage was much lower at the facilities from 2016-2020. The USFWS Delta Juvenile Fish Monitoring Program did not encounter any green sturgeon during beach seines and trawl surveys from 2012 to 2020. However, CDFW has captured over 140 juvenile green sturgeon in the Delta from 2016-2020 (Beccio 2020). Surveys in Grays Harbor and Willapa Bay (2020) report captures of over 120 green sturgeon (Heironimus 2021b). The reviewed information suggests that the number of green sturgeon entrained remains low.

Contaminants: Since the 2015 review, the application of the chemical pesticides carbaryl and imidacloprid to control burrowing shrimp (i.e., ghost shrimp (*Neotrypaea californiensis*) and mud shrimp (*Upogebia pugettensis*)) populations in Washington estuaries has been banned. The use of carbaryl, and its control of

burrowing shrimp and mud shrimp, poses a threat to North American green sturgeon. Since green sturgeon feed on burrowing shrimp, a potential negative impact from carbaryl application may occur, but little is known about the nature of this impact (Dumbauld et al. 2008). Exposure to carbaryl also may make green sturgeon more vulnerable to predation (NMFS 2009b). The chemical carbaryl was used to remove the threat of burrowing shrimp to oyster aquaculture in Willapa Bay and Grays Harbor. The chemical imidacloprid, a proposed alternative to carbaryl, was slated to come into use in 2015, but the use of imidacloprid was prohibited by the Ecology Department of Washington state in 2019 (Willapa-Grays Harbor Oyster Growers Assocation v. State of Washington, Department of Ecology and Ad Hoc Coalition for Willapa Bay, Center for Food Safety, Center for Biological Diversity, and Coaltion to Protect Puget Sound; October 15, 2019). Imidacloprid is less toxic than carbaryl to fish (Frew et al. 2015, Troiano and Grue 2016). Studies show that green sturgeon will opportunistically feed on burrowing shrimp that have been exposed to imidacloprid, which would result in exposure to imidacloprid through consumption of contaminated shrimp as well as the sediment porewater (Frew et al. 2015). No sublethal effects of imidacloprid were observed during exposure studies on surrogate white sturgeon (Frew and Grue 2015). Carbaryl is also used in Central Valley agriculture, which may make green sturgeon more vulnerable to predation, but its full effects on green sturgeon have not been studied.

Selenium contamination in San Francisco Bay, San Pablo Bay, and Suisun Bay continues to pose a potential threat to Southern DPS green sturgeon because green sturgeon feed on benthic invertebrates, including the Asian clam (*Corbula amurensis)*, which is an effective bioaccumulator. As stated in the 2015 review, yolk sac larvae of green sturgeon are more sensitive to selenium than those of white sturgeon (USFWS 2012). Although larval green sturgeon are not found in the SFBDE, the impact of selenium on larval green sturgeon informs current knowledge of how selenium affects the species. Using a regression approach and data from white sturgeon as a proxy, USFWS (2012) calculated selenium concentrations in the tissue and diet of green sturgeon and offered benchmark selenium concentrations in different life stages. Adverse effects on green sturgeon including significant mortality and reduced growth rate was observed when exposed to L-Selenomethionine (Se-Met), a common natural food source of selenium, at levels in the range of selenium levels reported in the benthic macro-vertebrate community of San Francisco Bay (De Riu et al. 2014). Exposure had a more severe pathological effect on green sturgeon as compared to white sturgeon (De Riu et al. 2014). De Riu et al. (2014) concluded that white sturgeon is a poor surrogate model for green sturgeon dietary Se-Met toxicity.

Other contaminants impacting green sturgeon were observed in laboratory and field studies. Laboratory experiments in which green sturgeon were exposed to dietary methylmercury indicate that green sturgeon are more susceptible to being adversely affected by dietary methylmercury as compared to white sturgeon as evidenced by higher mortality and lower growth rates (Lee et al. 2011). A blood

plasma study of wild green sturgeon in Washington estuaries observed that most frequently detected contaminants included Aldrin, 4,4,-DDE, a-BHC, copper and selenium (Layshock et al. In press.). Layshock et al. (In press.) found that fish caught in the most urbanized sites had the highest contaminant loads. The study did not find any significant difference in contaminant load based on sex, age, or maturity, or population. No additional information is available on the impacts of other chemicals, pesticides, or heavy metals on Southern DPS green sturgeon.

Climate Change: Climate change has the potential to impact Southern DPS green sturgeon in the future, but it is unclear how changing oceanic, nearshore, and river conditions will affect the Southern DPS overall. In freshwater environments (e.g., Sacramento River system), water flow and temperature are important factors influencing Southern DPS green sturgeon spawning and recruitment success (see Section 2.3.1.1). Climate change models predict increased runoff in the winter with reduced spring flows over the course of the 21st century (CH2M HILL 2014, NMFS 2019). Reservoir operations will also be impacted by climate change, with reservoirs filling up earlier and excess water being released to ensure flood control capacity. These changes in water temperature and flow in the Sacramento, Feather, and Yuba rivers may impact the timing and success of Southern DPS green sturgeon spawning.

It is difficult to predict how the Southern DPS may respond to these changing conditions and how climate change impacts in the nearshore and estuarine environment will also impact spawning timing and success. For example, ambient water temperature modeled under the Proposed Action of the Long-Term Operations of the Central Valley Project and State Water Project may exceed suitable levels (>17°C) during the critical egg fertilization and incubation period in the majority of years at the downstream extent of the spawning reach near Hamilton City, California (NMFS 2019). Klimley et al. (2020) modeled the impact of flows on habitat suitability within three known spawning areas in the Sacramento River. In years of low discharge (2012 to 2015 drought conditions), the amount of suitable spawning habitat was reduced in one of three locations (Klimley et al. 2020). Diminished flows in dry years may impact Southern DPS spawning habitat. As stated in the 2015 review, the salinity in the Sacramento River is projected to increase by 33%, on average, by the end of the 21st century (CH2M HILL 2014). This will result in declining habitat quality and food web productivity, which will likely impact the health of green sturgeon subadults. Laboratory experiments confirm the potential negative impacts of salinity and prey base changes predicted for the SFBDE on green sturgeon (Sardella and Kultz 2014, Haller et al. 2015, Vaz et al. 2015). Similar climate change induced habitat quality impacts in estuaries in Washington and Oregon could affect the health of subadult and non-spawning adult Southern DPS green sturgeon. The prey base for the Southern DPS could be further impacted by ocean acidification. Changing ocean conditions could also impact Southern DPS green sturgeon since subadults and adults use ocean habitats for migration and potentially for feeding. Based on their use of coastal bay and estuarine habitats, subadults and adults can occupy

habitats with a wide range of temperature, salinity, and dissolved oxygen levels, so predicting the impact of climate change in these environments is difficult (Kelly et al. 2007, Moser and Lindley 2007).

Rodgers et al. (2019) analyzed the mean effects of elevated temperatures, salinity, low food availability, and contaminants on growth as well as thermal tolerance, swimming performance, and heat shock protein expression. All variables significantly impaired green sturgeon growth. Elevated water temperature increased heat shock protein expressions and deformities while decreasing hatchling success. Salinity increased plasma osmolality and muscle moisture. Food restriction only affected growth. Contaminants increased mortality and body burden. Rodgers et al. (2019) suggested conservation measures such as mitigating salt intrusion in nursery habitats and maintaining water temperatures within optimal ranges during peak spawning periods. Overall, our knowledge of the environmental impact of climate change is increasing, but the direction of the impact on the Southern DPS is unknown at this point in time. Further research on the impact of climate change (multiple stressors) on embryonic, yolk-sac larvae, and post yolk-sac larvae needs to occur. Monitoring potential impacts into the future is important for population recovery.

Kinetic Energy Projects: An emerging threat is the development and operation of offshore and near shore kinetic energy projects. Impacts of such projects on green sturgeon could involve direct mortality or habitat loss and sensitivity to low levels of electromagnetic fields associated with the operations that could impact migration and habitat use (Nelson et al. 2008, Normandeau et al. 2011, NMFS 2018). Another concern is the potential effect on green sturgeon from the use of turbines at the mouths of large rivers (e.g., just upstream of the Golden Gate Bridge in San Francisco Bay). The effect of electromagnetic fields from a high voltage, direct current cable leading from Pittsburg, California to San Francisco, California has been studied based on detections of acoustically tagged green sturgeon before and after the cable was installed in 2010 (Klimley et al. 2017). Cable activity did not impact overall successful movement of green sturgeon through the area. However, additional research is needed regarding this threat, including examining the response of green sturgeon to different levels of electromagnetic fields. It should be noted that the permitting process for these facilities considers potential Southern DPS effects and monitoring may be a requirement for any facility receiving a permit.

Summary: In summary, no new information is available regarding the threats posed by non-native species. While efforts have been made to screen large diversions (>250 cfs), entrainment into smaller diversions still poses a threat to Southern DPS green sturgeon. No changes in NMFS or CDFW screen criteria have been made since the last review. Screening and facility operation criteria needs to be developed for green sturgeon. Carbaryl and imidacloprid have been phased out, but a new chemical may be used in its place in the future, which could impact the Southern DPS. Selenium is still likely a threat to the Southern DPS.

The threat of climate change and ocean acidification to Southern DPS green sturgeon cannot be measured using the available information, but changing freshwater and nearshore environments could impact Southern DPS green sturgeon health, spawning, and recruitment. The emerging threat posed by nearshore and offshore energy development requires continued attention into the future. The threats covered in this section are numerous. Overall, the new information does not support a conclusion that the threats have increased in severity since the last review, but many of the threats require close attention into the future.

2.4 Synthesis

The DPS structure of the North American green sturgeon has not changed since the last review. The Southern DPS occupies the same range as originally defined. Southern DPS spawning primarily occurs in the mainstem Sacramento River. Spawning has also been confirmed in the Feather and Yuba rivers in some years at low levels, relative to the spawning that occurs in the Sacramento River. The spatial structure of Southern DPS green sturgeon within the Sacramento River and in coastal environments is now better defined. Limited occupancy within the Sacramento River is concerning, and trends in this pattern and the number of individual green sturgeon present in the river should be monitored into the future. Protective measures instituted by CDFW to prohibit any sturgeon fishing where Southern DPS green sturgeon reproduce are important and should be maintained.

Many of the principal factors considered when listing Southern DPS green sturgeon as threatened are relatively unchanged. Recent studies confirm that the spawning area utilized by Southern DPS green sturgeon is small. Confirmation of spawning in the Feather and Yuba rivers is encouraging and the decommissioning of RBDD and the breach of Shanghai Bend makes spawning conditions more favorable, although Southern DPS green sturgeon still encounter impassable barriers in the Sacramento, Feather, and Yuba rivers that limit their spawning range. Improving fish passage at Fremont Weir is also encouraging for subadult and adult survival by reducing entrainment and potential poaching due to entrainment on floodplains (Seesholtz 2020). The relationship between altered flows and temperatures in spawning and rearing habitat and Southern DPS green sturgeon population productivity is uncertain. Entrainment, as well as stranding in flood diversions during high water events, also negatively impact Southern DPS green sturgeon. The prohibition of retention in commercial and recreational fisheries has eliminated a known threat and likely had a very positive effect on the overall population, although recruitment indices are not presently available.

The 2018 Recovery Plan developed five demographic criteria for recovery of Southern DPS green sturgeon. Since there were no reliable estimates of historical or current Southern DPS green sturgeon abundance, adult abundance criteria was developed using the best available information from general principles in conservation biology relating population viability to abundance (NMFS 2018). Continued annual DIDSON surveys are needed to track Southern DPS green sturgeon spawning populations into the future.

Additional future work utilizing this and other data sources (Beamesderfer et al. 2007) to look at abundance within a modeling framework would be useful and could provide a baseline for understanding the population-level impact of various sources of Southern DPS green sturgeon "take" authorized under the ESA 4(d). Studies measuring fisheries bycatch mortality by gear type would assist in measuring the impact of bycatch of Southern DPS green sturgeon in state and federal fisheries. Information gathered through the FMEP process will assist in understanding and limiting fisheries impacts.

Evaluation of new information since the 2015 review does not suggest a significant change in the status of Southern DPS green sturgeon. With respect to threats, the available information indicates that some threats, such as those posed by fisheries and impassable barriers, have been reduced. The threat of climate change on all life stages of green sturgeon persists, but requires continued research. The emerging threat posed by nearshore and offshore energy development requires continued attention into the future. Since the majority of the threats cited in the original listing still exist, the Threatened status is still applicable.

3.0 RESULTS

3.1 Recommended Classification

Given your responses to previous sections, particularly section 2.4. Synthesis, make a recommendation with regard to the listing classification of the species

_____**Downlist to Threatened**
_____**Uplist to Endangered**
_____**Delist** *(Indicate reason for delisting per 50 CFR 424.11):*
_____*Extinction*
_____*Recovery*
_____*Original data for classification in error*
X____**No change is needed**

3.2 New Recovery Priority Number

No change

Brief Rationale:

3.3 Listing and Reclassification Priority Number

Reclassification (from Threatened to Endangered) Priority Number: ____
Reclassification (from Endangered to Threatened) Priority Number: ____
Delisting (Removal from list regardless of current classification) Priority Number: ____

Brief Rationale:

4.0 RECOMMENDATIONS FOR FUTURE ACTIONS

The Recovery Plan for the Southern DPS of the North American Green Sturgeon was completed in August 2018. The Recovery Plan outlines specific recovery actions, research priorities, monitoring priorities, and education and outreach priorities for Southern DPS Green Sturgeon. Actions and Priorities within the plan are ranked. Actions stemming from this review that would assist in improving the status of and available information about Southern DPS green sturgeon are as follows:

1. Continue monitoring and studying key life history stages and modeling population abundance. Long-term monitoring is needed to develop temperature and flow targets in accessible spawning, incubation, and rearing habitats. Research is needed on the impact of habitat modification and/or restoration (e.g., levee alteration, floodplain connectivity) on green sturgeon recruitment and growth and water management on green sturgeon habitat, as well as growth and survival. Increased acoustic tagging and monitoring for all life stages of green sturgeon is needed throughout its range but especially within the Sacramento River basin and SFBDE. Continued funding for a core acoustic telemetry array is highly recommended. Acoustic monitoring for green sturgeon needs to be expanded to South San Francisco Bay to better understand habitat use and foraging of subadults. Also, greater monitoring for juvenile green sturgeon is needed in the lower estuary to better understand juvenile green sturgeon habitat use of the SFBDE. Monitoring for green sturgeon in the San Joaquin River basin using eDNA, telemetry, and other sampling methods (i.e., fyke trap, hook-n-line) should be increased.
2. Continue efforts to remove barriers to migration and improve fish passage so that Southern DPS green sturgeon can reach spawning grounds and to reduce entrainment on floodplains. Upstream passage above the boulder weir located at Sunset Pumps in the Feather River is needed so that Southern DPS green sturgeon will be able to access spawning grounds. Structural improvements to Fremont Weir (Yolo Bypass) and Tisdale Weir (Sutter Bypass) need to be made to reduce stranding and provide passage. In addition to fish passage improvements at Yolo and Sutter bypasses, adaptive management plans should be developed with green sturgeon experts to reduce any further entrainment of sturgeon. Fish passage for Southern DPS green sturgeon needs to be developed at Daguerre Point Dam on the Yuba River. Removing this barrier will likely provide greater access to suitable spawning habitat for Southern DPS green sturgeon. Further research is needed to determine the impact of the Sacramento Deep Water Ship Channel and Port of Sacramento on upstream migration of green sturgeon. Research is also needed to determine the impact on the operations of the Delta Cross Channel gates on green sturgeon migration.
3. Continue efforts to reduce entrainment of juvenile green sturgeon at water diversions with the development of screening and fish passage criteria. First, all current and proposed water diversions posing a risk to green sturgeon need to be identified. Second, operation and screening guidelines need to be developed for green sturgeon. The modules outlined in the UCD Sturgeon Study Plan provide experiments that will develop criteria based on swimming performance metrics, attraction flows, and fishway interaction for all

life stages of green sturgeon. Once criteria is developed, it should be applied in the mainstem Sacramento, Feather, and Yuba rivers as well as the SFBDE to reduce early life stage entrainment. Climate change and sea level rise should also be addressed by the screening and passage criteria.

4. Continue efforts to reduce take from poaching and fisheries bycatch. There should be an increased enforcement presence at Yolo and Sutter bypasses when overtopping events occur to prevent poaching events. Relocation efforts of green sturgeon from entrainment on floodplains by CDFW should continue to occur with passage improvement at Fremont and Tisdale weirs. Further research is needed to estimate the annual level of mortality of Southern DPS green sturgeon from poaching. An outreach program for law enforcement personnel, fishing guides, and fishermen should be developed to inform them of the green sturgeon protections under the ESA as well as potential problems of post-release mortality and poaching. The FMEPs for state fisheries encountering Southern DPS green sturgeon also need to be completed. Research on green sturgeon mortality from incidental capture in fisheries (e.g., gillnet, hook-n-line, coastal trawling) should continue to occur. Research on green sturgeon interaction with commercial fishing gear should also continue to occur. Estimating mortality from poaching and fisheries bycatch is needed to inform population estimates for Southern DPS green sturgeon.
5. Continued research of the impact of contaminants, disease, predation, and climate change on all life stages of green sturgeon should occur. More data is needed to inform natural resource managers about these threats to Southern DPS green sturgeon recovery.

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NATIONAL MARINE FISHERIES SERVICE
5-YEAR REVIEW
Southern Distinct Population Segment of the North American Green Sturgeon
(*Acipenser medirostris*)

Current Classification: Threatened

Recommendation resulting from the 5-Year Review

____ Downlist to Threatened
____ Uplist to Endangered
____ Delist
X___ No change is needed

Review Conducted By:

National Marine Fisheries Service, West Coast Region, California Central Valley Office

REGIONAL OFFICE APPROVAL:

Lead Regional Administrator, NOAA Fisheries

Approve _____ Date: October 26, 2021

The Lead Region must ensure that other Regions within the range of the species have been provided adequate opportunity to review and comment prior to the review's completion. Written concurrence from other regions is required.

HEADQUARTERS APPROVAL:

Assistant Administrator, NOAA Fisheries

__X___Concur _____ Do Not Concur

Signature RAUCH.SAMUEL.D.III.1365850948 Digitally signed by RAUCH.SAMUEL.D.III.1365850948 Date: 2021.10.28 10:47:38 -04'00' Date _____

Appendix F, Modeling

Attachment F.4 Delta Smelt Life Cycle Model with Entrainment

F.4.1 Model Overview

Polansky et al. (2021) developed a hierarchical stage-structured state-space life cycle model for Delta Smelt to identify factors with the strongest statistical support for having influence on the species' recruitment and survival. This modeling approach is useful as an ecological modeling tool because it can separate descriptions of state and observation processes and permit the integration of disparate data sets. This Delta Smelt life cycle model was later expanded from four to seven life stages with a component that separately describes the entrainment process at the Delta export facilities (Smith et al. 2021). This model produces expected values for larval recruitment and survival at the subsequent life stages. The most statistically supported model variant in Smith et al. (2021) used means of December-June Old and Middle River (OMR) values and June-August outflow aggregated from monthly values or longer timescales; therefore, CalSim output for the scenarios/alternatives can be directly incorporated into the model framework. The most statistically supported model in Smith et al. (2021) also included food/prey metric term during the months of January to March. By using the relationship between zooplankton density and salinity, CalSim-predicted X2 values were then used to estimate the expected change in the food/prey metric for January-March months across alternatives. Reclamation used this model to calculate expected annual population growth rate (λ; the abundance of current year divided by abundance from previous year) for alternative flow scenarios by using CalSim output and subsequent zooplankton model. The metric of interest will be geometric mean of λ for a specified time frame (e.g., 1995-2015), which will be compared across alternatives. For the purpose of this text, Smith et al.'s (2021) model will be referred to as the Delta Smelt Life Cycle Model with Entrainment (LCME).

F.4.2 Methods, assumptions

The Delta Smelt LCME was run based on flow inputs from CalSim 3. The approach followed the Collaborative Science and Adaptive Management Program (CSAMP) Delta Smelt Structured Decision Making (SDM) process, where historical years (1995-2015) were adjusted according to a CalSim 3 scenario and the geometric mean λ was calculated for each scenario. There is an expectation that zooplankton abundance (i.e., prey item for Delta Smelt) would change based on flow (Kimmerer and Rose 2018), and as such, a zooplankton submodel constructed for the Delta Coordination Group and CSAMP Delta Smelt SDM was applied to the CalSim 3 scenarios. For the zooplankton term, an upper and lower 95% confidence intervals were calculated and applied into the analysis to better understand sensitivity of the model output to variation in zooplankton

abundance. We did not update any other model inputs (turbidity, temperature, and predators) due to the complexity and lack of predictive models associated with the other values. Furthermore, it is unclear whether flow changes at a project operations scale meaningfully affect the functioning of the Bay-Delta food web. What is of interest in this analysis is to determine how much the expected long-term abundance of delta smelt might change based on the proposed changes in water management.

F.4.2.1 Main model

Monthly flow data were pulled from CalSim 3 dss files through R and were summed or averaged depending on the variables for the LCME. Old and Middle river flow variables were either extracted directly from CalSim 3 as monthly average value in cfs, or averaged if the timespan covers two months (Table F.4-1). Sum of Delta outflow from June to August were calculated by multiplying the CalSim 3 predicted monthly Net Delta Outflow Index (NDOI-ADD + NDOI-MIN) by the number of days for each month and then added together. The total values in cfs per day were then converted to acre-feet (1 cfs = 1.983 acre-feet per day). Methods and findings of the original application of LCME can be found in Smith et al. (2021). The list of LCME flow variables that were acquired from the CalSim 3 runs can be found in Table F.4-1. R script and data used for the model can be found at (https://github.com/BDO-Science/DeltaSmelt_LCM).

Table F.4-1. List of covariates used in LCME that were replaced with values from CalSim 3 for each alternative. The Old and Middle River covariates imply entrainment as the mortality mechanism (Smith 2019; Smith et al. 2020). The Delta outflow covariate implies foraging habitat suitability as a suite of mechanisms that align better when outflow is elevated (Smith and Nobriga 2023). The covariates are listed in the order they affect a given cohort in the model.

Life stage	Covariate	Unit	Covariate summary details
Early post-larval (May)	April-May Old and Middle river flow	cfs	Mean of the daily sum of tidally filtered flows in the Old and Middle rivers during April to May
Late post-larval (June)	June-August Outflow	Acre-foot	Sum of the volume of water moving past a point near the confluence of the Sacramento and San Joaquin rivers, near Pittsburg, California, during June to August
Late post-larval (June)	June Old and Middle river flow	cfs	Mean of the daily sum of tidally filtered flows in the Old and Middle rivers during June
Early subadult (October-November)	December-January Old and Middle river flow	cfs	Mean of the daily sum of tidally filtered flows in the Old and Middle rivers during December to January
Late subadult (January-February)	February Old and Middle river flow	cfs	Mean of the daily sum of tidally filtered flows in the Old and Middle rivers during February
Early adult (March)	March Old and Middle river flow	cfs	Mean of the daily sum of tidally filtered flows in the Old and Middle rivers during March

F.4.2.1.1 Assumptions related to the model calibration and new flow inputs

- The LCME was parameterized using Old and Middle River (OMR) flow values derived from the USGS gages and Delta Outflow estimates from DAYFLOW (https://data.cnra.ca.gov/dataset/dayflow), which may differ to some extent with how CalSim 3 calculates these values (OMR and Delta Outflow).

- The LCME separately accounts for the influence of OMR and turbidity on delta smelt entrainment. However, the CalSim 3 runs had assumptions built into them about how frequently turbidity triggers that affect OMR would occur. This confounds the turbidity effect on entrainment with the OMR effect in a way that the LCME cannot account for. This may lead to a negative bias in the predicted effect of entrainment; in other words, it may be underestimated somewhat.

- The only flow data included in the published LCME (Smith et al. 2021) are OMR and June-August Delta Outflow. In essence, the LCME assumes that these are the most influential flow variables associated with Delta Smelt recruitment and survival. This assumption was supported by Polansky et al. (2021), which is why these flow variables were carried forward and re-tested in the Smith et al. (2021) model.

- This analysis consisted of the years 1995 to 2015, so it is unclear how representative model predictions of Delta Smelt population trajectory will be when simulating scenarios that include environmental conditions outside the range of observations the model was fit to. In addition, it is unclear how model parameter estimates and predictions of Delta Smelt population may be affected by climate change impacts and the ongoing and proposed supplementation efforts.

F.4.2.2 Zooplankton model

To calculate zooplankton abundance/density changes related to changes in flow associated with the CalSim 3 scenarios, Reclamation leveraged the zooplankton abundance estimation process used in the CSAMP Delta Smelt SDM group. To replicate the zooplankton abundance calculation used in the CSAMP Delta Smelt SDM process, estimated X2 values for each month were first retrieved from CalSim 3 dss files. These monthly X2 values were then converted into salinity values for each region defined in the Delta Smelt Individual-Based Model (IBM) (Rose et al. 2013) using a generalized linear model developed by Compass (see Attachment 1).

Similar to the CSAMP Delta Smelt SDM process, Generalized Additive Models (GAMs) were constructed to predict the zooplankton density Delta Smelt were expected to spatially overlap with given a salinity level for each IBM region and zooplankton taxon (see Attachment 2). Predictor variables for each GAM were the tensor product smooth of the interaction between salinity and day of year, as well as random effects for year and station (when more than one station exists in the dataset). To produce the monthly model output, the 15th of each month was used as the data input.

Once salinity values were calculated for each Delta Smelt IBM region, month, and scenario, expected zooplankton densities were then estimated for every zooplankton taxon, month, region, and scenario using output from the GAMs. Upper and lower 95% confidence intervals from these predictions were calculated through 1,000 independent draws from the model distribution, similar to a bootstrapping process. Just as was done in the CSAMP Delta Smelt SDM process, for each alternative, the initial output was scalar values of the taxa-specific zooplankton density under the particular management conditions divided by the same prediction under baseline/historical conditions. However, because 0 values were present in the baseline, it resulted in infinite values for the scalar calculations. These infinite values were replaced with the maximum finite scalar calculated from model predictions for a specific alternative, taxon, region, and month (across years). When this step still yielded no finite scalar value, the maximum finite scalar value from a given alternative, taxon, and month was used instead.

Because the Delta Smelt IBM and LCME differ in how regions are defined and how zooplankton taxa are grouped, additional conversions were needed. The Delta will continue to be managed as a freshwater ecosystem (i.e., not expected to vary much in terms of salinity) in the near future, and as such, any IBM regions upstream of the Confluence were ignored, and likewise LCME North and South regions were left as is (Figure F.4-1). The Far West LCME region only overlapped with the SW Suisun IBM region and thus, the SW Suisun IBM region results were used to define zooplankton changes in the Far West LCME region. To calculate zooplankton changes in the West LCME region, the following IBM regions were used: NW Suisun, NE Suisun, SE Suisun, Suisun Marsh, and the Confluence. Results from the five IBM regions within the West LCME region were aggregated by multiplying each IBM region's value with the proportion of the region's water volume relative to the total water volume across all five regions. The calculations were as follows:

- Far West LCME region: SW Suisun IBM region (Figure F.4-1)
- West LCME region: (Confluence IBM region x 0.233) + (Suisun Marsh IBM region x 0.174) + (NE Suisun IBM region x 0.110) + (SE Suisun IBM region x 0.220) + (NW Suisun IBM region x 0.264)

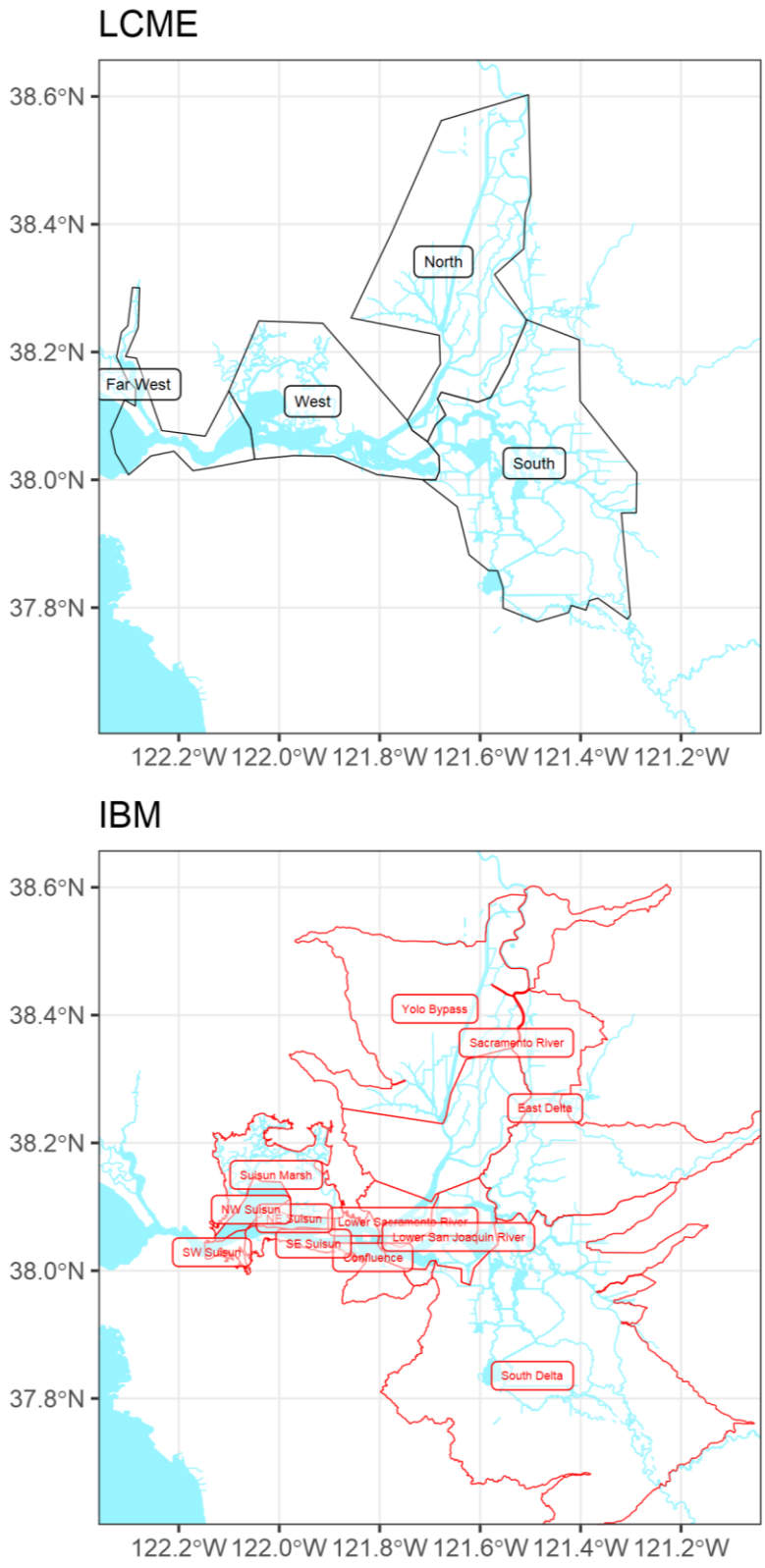


Figure F.4-1. Map of the San Francisco Bay-Delta with LCME regions shown in black (top) and IBM regions shown in red (bottom).

The LCME uses aggregate zooplankton biomass per volume values calculated by summing a number of different zooplankton species and life stages (see Attachment 3), whereas IBM taxa were more specific, often down to species. Therefore, the proportion of each zooplankton taxa that make up the aggregate zooplankton groups in the LCME data input had to be first estimated for each month and LCME region using raw data provided by the primary authors of the LCME ("ZooMysid_74_19_df.csv"). Using these proportions, the final scalar multiplier values were acquired for the Far West and West LCME regions and zooplankton aggregate groups. In other words, the multiplier scalar values were applied based on the proportion of the particular taxon that make up the prey biomass for a given month and LCME region. For example, if *Pseudodiaptomus forbesi* adults are expected to be twice as abundant and *Eurytemora affinis* adults are expected to be three times as abundant under an alternative, and the two species make up 50% of the biomass each, the final multiplier scalar values will be 2.5 (i.e., [2 x 0.5] + [3 x 0.5]).

These final scalar multipliers were then applied to the LCME aggregated zooplankton dataset ("ZooMysid_74_19_df_median.csv") for the median estimate and the lower and upper 95% confidence interval values (Figure F.4-2). These predictions were then capped at the maximum value that was observed in the LCME aggregated zooplankton dataset ("ZooMysid_74_19_df_median.csv") for the region and month using only data from 1995 to 2019. Lastly, the prey covariates (see Table F.4-3) were acquired by calculating the mean across the four LCME regions.

Table F.4-2. List of taxa analyzed using GAM and the equivalent LCME taxa used to calculate the proportion of each taxon that make up the prey biomass at a given month and LCME region.

GAM response variable	**Taxon definition**	**LCME taxon used to calculate proportion of prey biomass for each month and LCME region**
acartela	*Acartiella sinensis* (copepod) adults	*Acartiella sinensis* (copepod) adults
eurytem	*Eurytemora affinis* (copepod) adults	*Eurytemora affinis* (copepod) adults
pdiapfor	*Pseudodiaptomus forbesi* (copepod) adults	*Pseudodiaptomus forbesi* (copepod) adults
othcalad	Other calanoid copepod adults	Other calanoid adults + *Sinocalanus doerrii* (copepod) adults
othcaljuv	Other calanoid copepodites	Calanoid copepodids + Other calanoid copepodids + *Eurytemora affinis* copepodids + *Sinocalanus doerrii* copepodids + *Pseudodiaptomus* spp. Copepodids + *Acartiella sinensis* copepodids + *Acartia* spp. Copepodids + Diaptomidae copepodids + *Tortanus* spp. copepodids
limno	*Limnoithona* spp. copepods (all stages)	*Limnoithona* spp. + *Limnoithona sinensis* + *Limnoithona tetraspina*

GAM response variable	Taxon definition	LCME taxon used to calculate proportion of prey biomass for each month and LCME region
othcyc	Other cyclopoid copepods (all stages)	*Acanthocyclops vernalis*
allcopnaup	Copepod nauplii (all spp.)	Copepod nauplii + Other copepod nauplii + *Eurytemora affinis* nauplii + *Sinocalanus doerrii* nauplii + *Pseudodiaptomus* spp. nauplii
daphnia	*Daphnia* spp. (cladocerans)	*Daphnia* spp. (cladocerans)
othclad	Other cladocerans	*Bosmina longirostris* + *Diaphanosoma* spp. + Other cladocera
other	All other taxa	N/A (model was not used)
mysid	*Hyperacanthomysis longirostris*	*Hyperacanthomysis longirostris* + *Neomysis mercedis*

As zooplankton covariates for natural mortality were only supported for adult life stages (Smith et al. 2021), only zooplankton modeling results from the months of February and March were used as data input for the LCME (Table F.4-3). In other words, a flow effect on delta smelt's food supply is only supported statistically in February-March. The most parsimonious mechanistic explanation is that prey available to adult fish early in the spawning season had a population-scale effect, perhaps by affecting how many eggs could be produced or affecting how many adults survived to spawn a second time. R script and data used for the salinity and zooplankton models can be found at https://github.com/BDO-Science/DeltaSmelt_LCM.

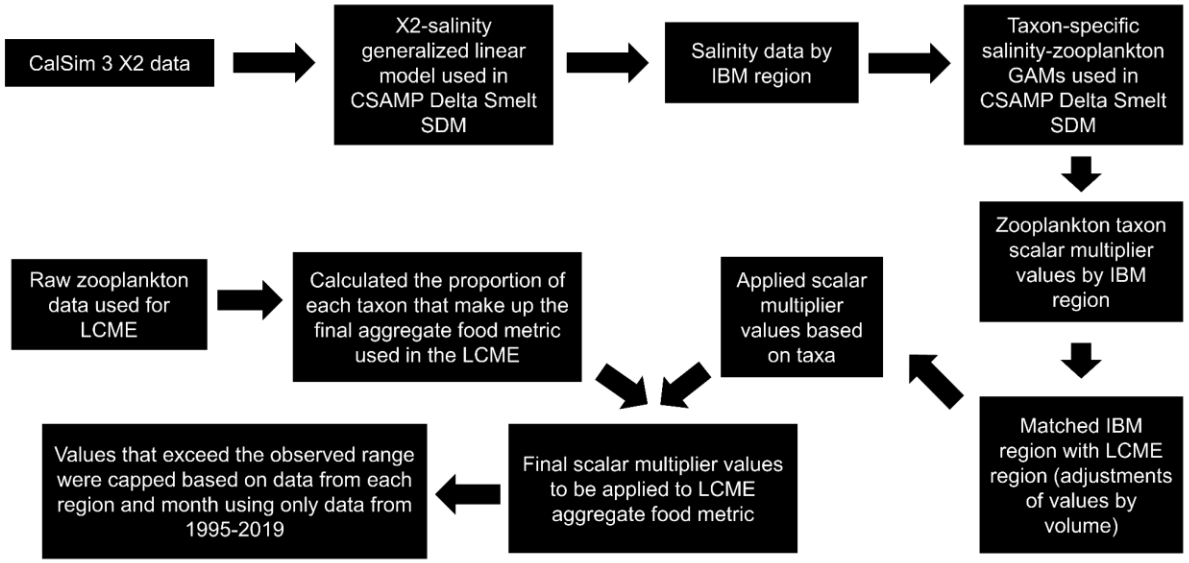


Figure F.4-2. Summary of steps taken to generate estimates of the zooplankton prey density metric for each alternative.

Table F.4-3. List of covariates used in LCME that were replaced with new values based on CalSim 3 and zooplankton model for each alternative. The mechanism implied by these prey density covariates is related to food limitation of adult spawners that may affect the number or quality of eggs produced or the number of repeat spawns the fish are able to complete before dying.

Life stage	Covariate	Unit	Covariate summary details
Late subadult (January-February)	Food metric for February	Microgram carbon per meter^3	Mean carbon-weighted density of adult calanoid copepods, cyclopoid copepods, cladocerans, and mysid shrimp observed during February zooplankton surveys
Early adult (March)	Food metric for March	Microgram carbon per meter^3	Mean carbon-weighted density of adult calanoid copepods, cyclopoid copepods, cladocerans, and mysid shrimp observed during March zooplankton surveys

F.4.2.2.1 Assumptions related to the model calibration and new flow inputs

The zooplankton modeling workflow used salinity to estimate changes in zooplankton biomass related to flow. There are several mechanisms by which a correlation between flow and zooplankton biomass may arise that are not based on salinity per se such as transport from upstream, estuarine circulation, etc.

The use of salinity as a covariate also meant that predicted zooplankton biomass at a particular region is static anywhere and everywhere salinity is ≤0.1 ppt salinity, even with additional Delta outflow.

The original purpose of the salinity and zooplankton modeling was to adjust zooplankton data input for the Delta Smelt IBM (Rose et al. 2013; Kimmerer and Rose 2018). As such, there were limitations when the data were converted for the purpose of Delta Smelt LCME (e.g., some missing species and/or life stages in the aggregate LCME zooplankton groups).

F.4.3 Results

The general statistical prohibition against extrapolation suggests that model predictions are more uncertain when explanatory variables are outside the range of observations to which the model was fit. To visually inspect when the predicted flows and food were outside the observed/empirical range for the LCME, output from CalSim 3 and the zooplankton model were plotted against the empirical data (i.e., data used to estimate parameters in the LCME). See Figure F.4-3 through Figure F.4-5 below. Most CalSim-predicted flows and zooplankton predictions were not outside the range of observations to which LCME was fit, but some alternatives did include out-of-range values. EXP1 included much lower June-August Delta Outflows than observed and higher (more positive) OMR values than observed in some years. EXP3 OMR values were similar to EXP1, but EXP3 June-August Delta Outflows were within the observed range. Alt1, the components of Alt2, and Alt4 contained some April-May OMR values that were more negative than the observed range. Alt1 also contained OMR values more negative than the observed data for the months of December-January and March. Overall, CalSim-predicted June-August Outflow values were generally lower than the DAYFLOW estimates under Wet or Above Normal years (Figure F.4-3, Figure F.4-6). Predicted prey biomass for all alternatives was within the observed range (Figure F.4-5). However, for certain years higher prey biomass than the empirical data were predicted for all alternatives (Figure F.4-3). As a result, mean predicted prey biomass across all alternatives were also higher than the observed data (Figure F.4-5 through Figure F.4-7).

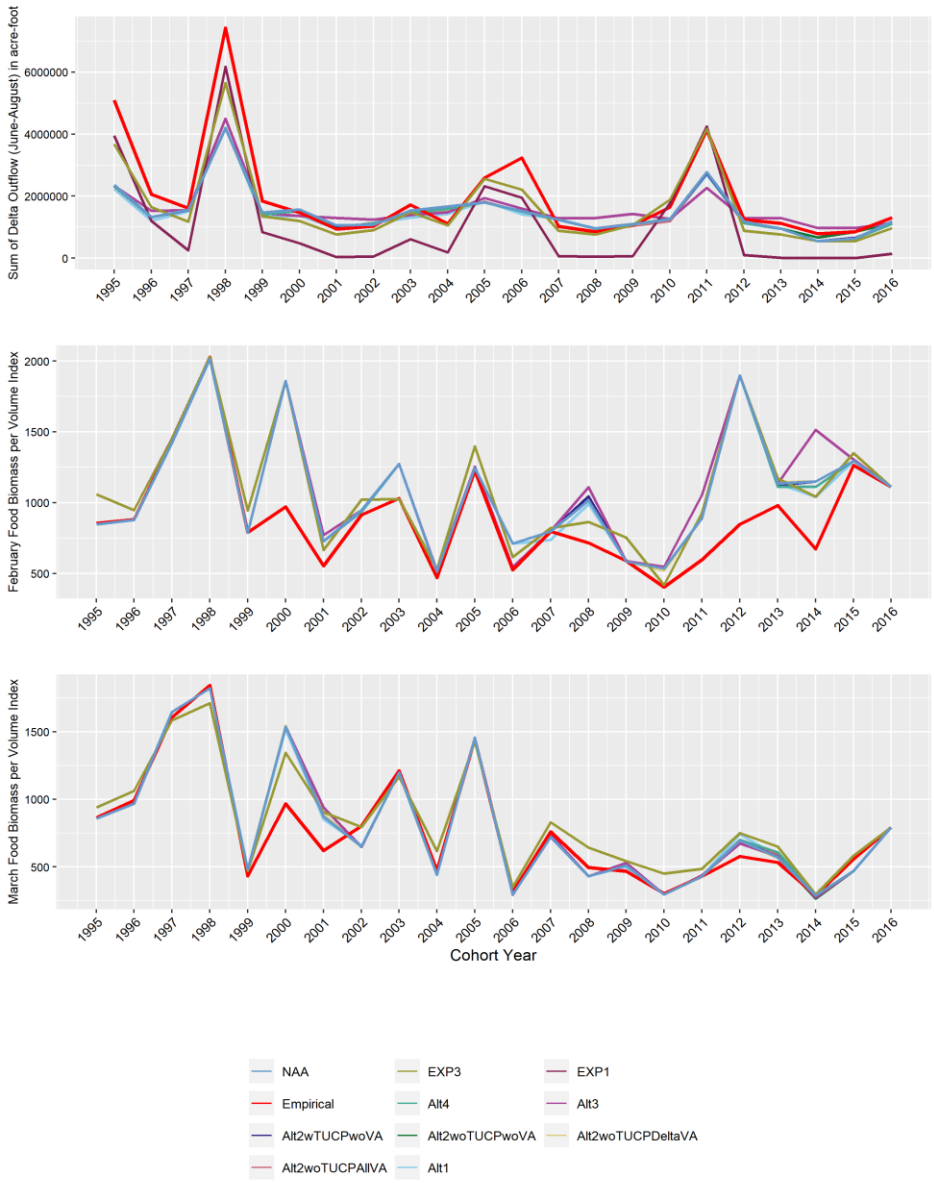


Figure F.4-3. Annual time series of outflow and prey metric data based on CalSim3 data and salinity-zooplankton model relative to the original dataset used to build the Delta Smelt LCME (labeled as "Empirical"). From top to bottom: June-August sum of Delta outflow, February, and March prey metric (biomass per volume) data composed of adult copepods, cladocerans, and mysids. Note that the x-axis represents Delta Smelt cohort year (e.g., February and March prey metric for cohort year 2012 represents data for February and March of 2013).

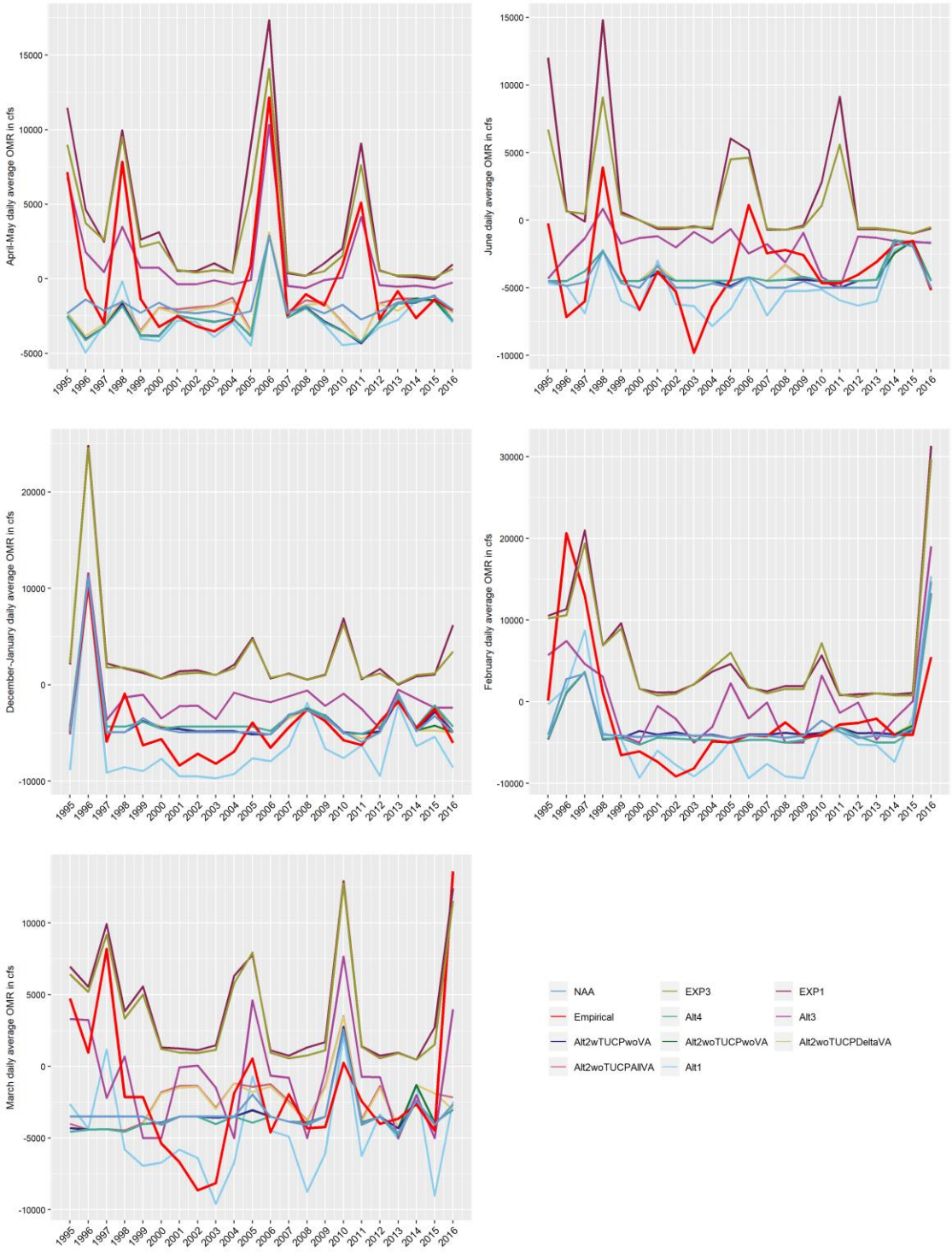


Figure F.4-4. Annual time series of monthly average OMR flow data for input to the LCME produced from CalSim3 relative to the original LCME dataset (labeled as "Empirical"). (e.g., February and March OMR values for cohort year 2012 represents data for February and March of 2013).

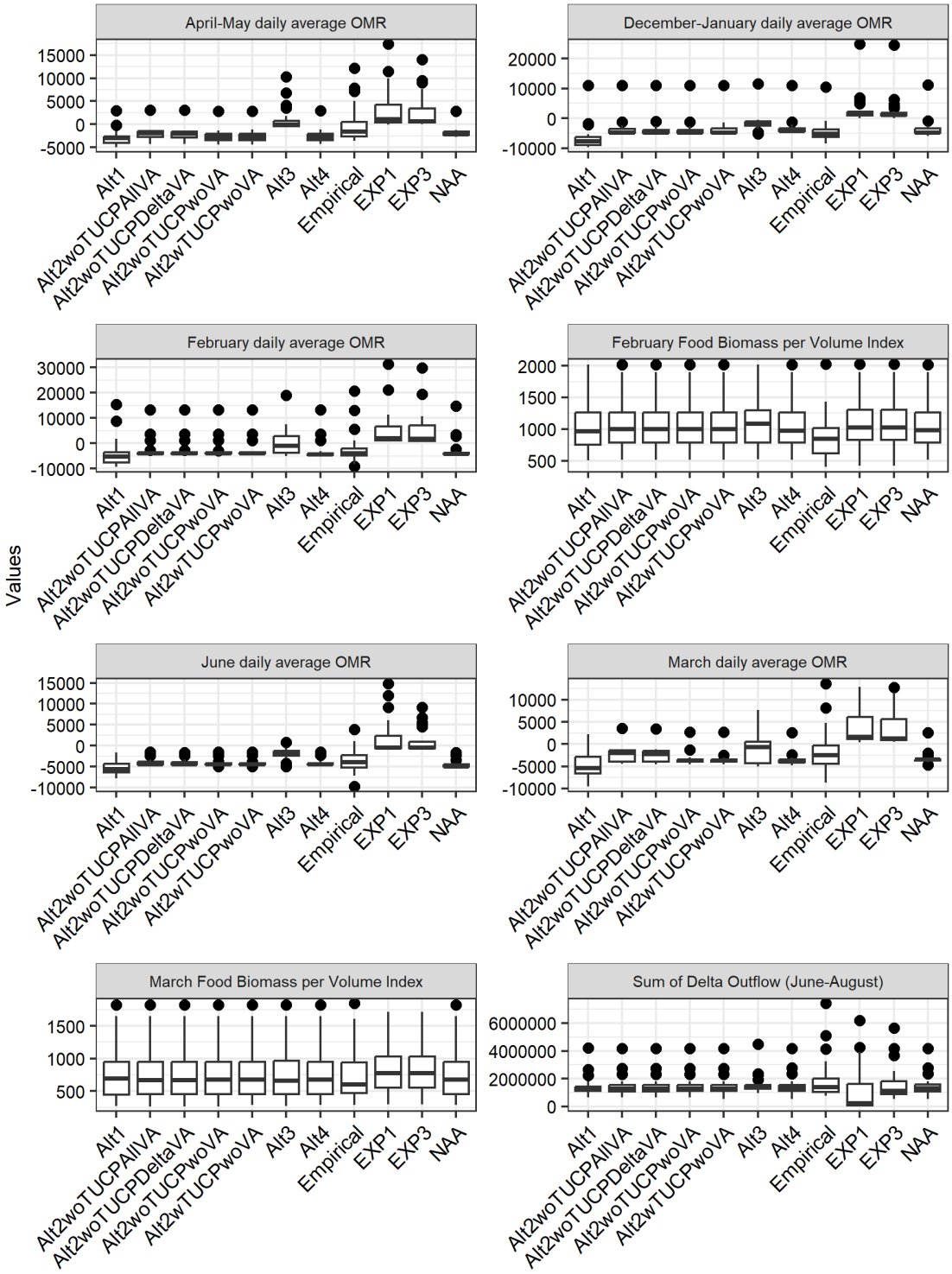


Figure F.4-5. Box plot of covariate values for cohort year 1995 to 2015 sorted by alternative.

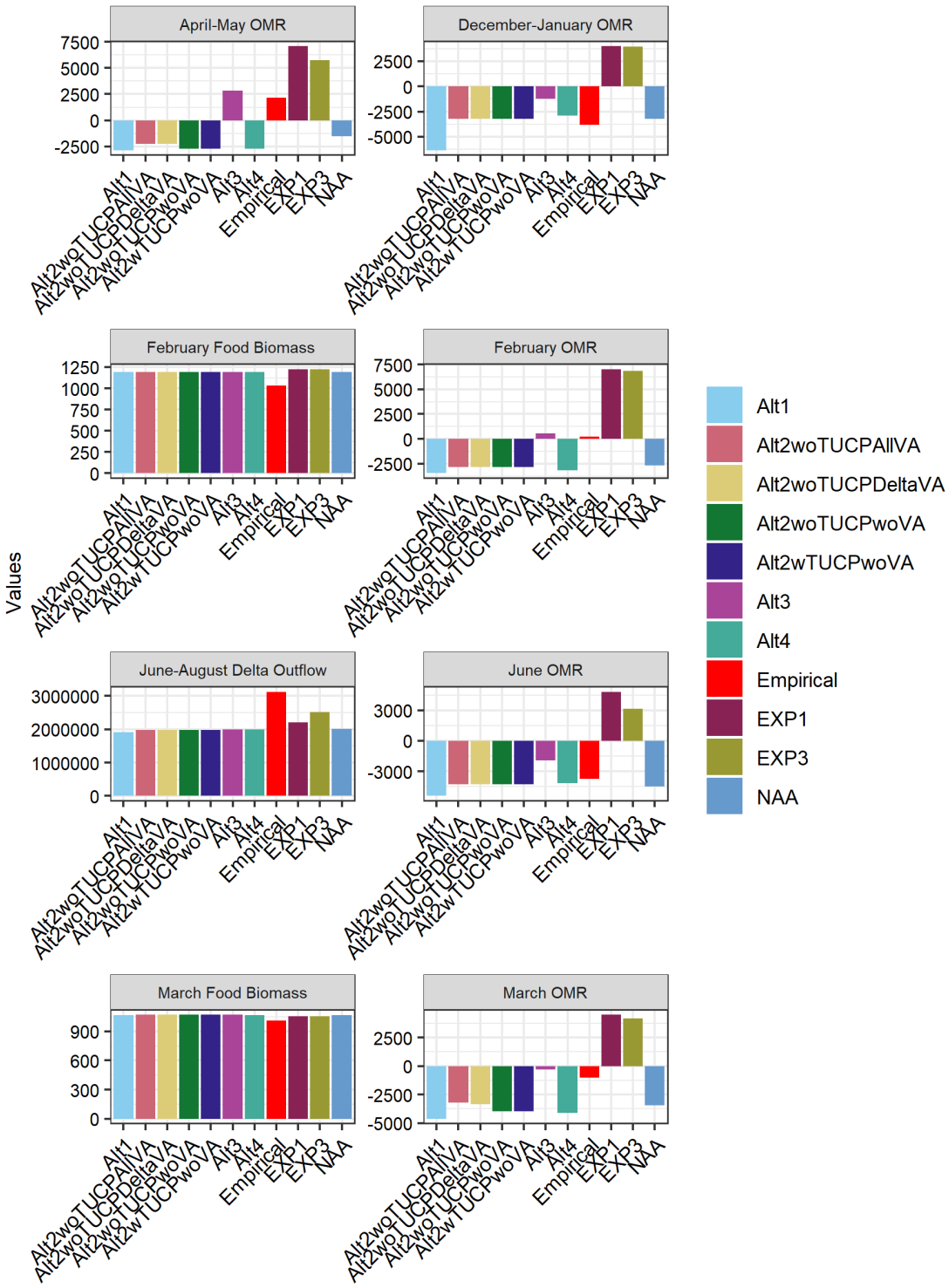


Figure F.4-6. Mean covariate values used in the LCME for Wet and Above Normal year types. Note that cohort year was matched with the water year that the cohort was born in (e.g., cohort year 1995 = water year 1995).

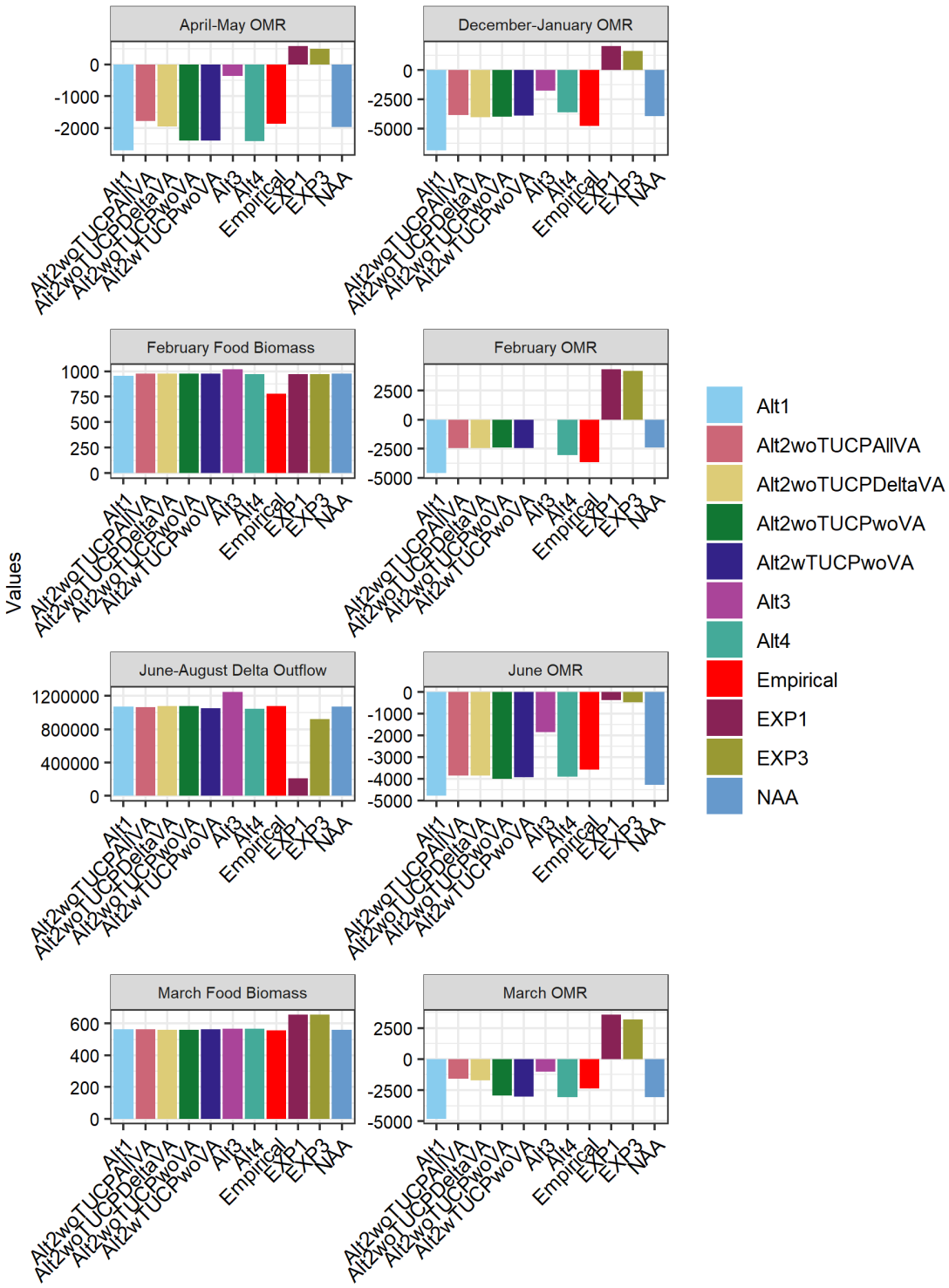


Figure F.4-7. Mean covariate values used in the LCME for Below Normal, Dry, or Critically Dry year Note that cohort year was matched with the water year that the cohort was born in (e.g., cohort year 1995 = water year 1995).

Estimates of population growth rate (λ; the abundance of current year divided by abundance from previous year) are provided for each cohort year and alternative (Table F.4-4; Figure F.4-8). Generally, dry years showed lower geometric mean λ than wet years (Table F.4-5), and wet years occurred with greater frequency at the beginning of the time series (1995-1999) compared to the end of the time series (2006-2015).

Summarized across all years by calculating the geometric mean of λ for the full 21-year time series (1995-2015), predicted flow and zooplankton conditions associated with EXP3 resulted in the highest mean λ, followed by Alt3 (Figure F.4-9). Meanwhile predicted conditions associated with Alt1 resulted in the lowest value of mean λ. All other alternatives resulted in mean λ between 0.88 and 1.02 (Table F.4-5). Relative to the no action alternative (NAA), Alt3 and EXP3 mean projected λ were the highest among all alternatives, and Alt1 was the lowest (Figure F.4-9). Decomposition of mean λ into time series plots of % change of population growth rate for a given alternative divided by the population growth rate of NAA demonstrated that EXP3- and Alt3-projected λ were greater than NAA in most years (Figure F.4-8). EXP1-projected λ were relatively greater than NAA in wet years, but less than NAA-projected λ in all other years.

NAA, the various versions of Alt2, and Alt4 performed similarly to the empirical data. While these CalSim-generated scenarios/alternatives resulted in higher λ than the empirical data during dry years, they also resulted in lower λ than the empirical data during wet years (Table F.4-5). The CalSim-generated scenarios/alternatives (NAA, the various versions of Alt2, and Alt4) may have produced higher λ during dry years due to the more positive OMR values for multiple months and higher zooplankton estimates in February (Figure F.4-6). Meanwhile, these same CalSim-generated scenarios/alternatives (NAA, the various versions of Alt2, and Alt4) may have produced lower λ than the empirical data during wet years because of the lower June-August Delta Outflow values and more negative OMR values for certain months (Figure F.4-7).

Table F.4-4. Predicted population growth rate (λ; abundance of current year divided by abundance from previous year) for each cohort year by alternatives. λ for cohort year 1995 was calculated by using a static abundance estimate for cohort year 1994. Empirical indicates the observed data used by the LCME.

Year	Empirical	Alt1	Alt2woTUCPAllVA	Alt2woTUCPDeltaVA	Alt2woTUCPwoVA	Alt2wTUCPwoVA	Alt3	Alt4	EXP1	EXP3	NAA	Sacramento Valley Water Year Index
1995	3.56	1.66	1.81	1.81	1.86	1.80	2.28	1.82	4.57	4.35	1.89	Wet
1996	1.37	0.66	0.64	0.63	0.64	0.64	1.07	0.63	1.20	1.47	0.73	Wet
1997	0.68	0.38	0.57	0.58	0.58	0.58	0.79	0.57	0.58	1.01	0.59	Wet
1998	4.78	1.74	1.67	1.65	1.66	1.65	3.02	1.73	5.18	4.91	1.81	Wet
1999	0.79	0.56	0.69	0.69	0.68	0.70	0.88	0.65	1.00	1.35	0.78	Wet
2000	0.69	0.42	0.80	0.79	0.84	0.86	0.78	0.72	0.77	1.15	0.86	Above Normal
2001	0.11	0.11	0.28	0.28	0.31	0.32	0.47	0.28	0.29	0.47	0.29	Dry
2002	0.55	0.69	0.92	0.93	1.00	1.00	1.20	0.92	0.74	1.23	0.90	Dry
2003	0.87	0.73	1.45	1.46	1.53	1.53	1.56	1.36	1.63	2.52	1.50	Above Normal
2004	0.44	0.43	0.79	0.79	0.82	0.82	0.91	0.74	0.65	1.11	0.82	Below Normal
2005	1.94	1.05	1.28	1.28	1.34	1.36	1.58	1.29	2.96	3.18	1.32	Above Normal
2006	3.37	1.87	2.09	2.12	2.19	2.22	2.79	1.99	3.15	3.46	2.21	Wet
2007	0.51	0.36	0.59	0.60	0.60	0.60	0.85	0.55	0.48	0.79	0.60	Dry
2008	0.95	0.92	1.01	1.00	1.07	1.08	1.37	1.01	0.64	1.02	1.01	Critically Dry
2009	0.64	0.49	0.73	0.73	0.73	0.72	0.81	0.71	0.62	1.16	0.73	Dry
2010	1.26	1.15	1.26	1.25	1.29	1.25	1.42	1.26	2.21	2.32	1.28	Below Normal
2011	3.65	2.66	2.77	2.76	2.76	2.77	2.60	2.80	4.78	4.69	2.74	Wet
2012	0.95	0.67	0.87	0.88	0.89	0.88	1.00	0.88	0.56	0.93	0.91	Below Normal
2013	0.90	0.79	0.83	0.83	0.84	0.84	1.01	0.83	0.44	0.73	0.83	Dry
2014	0.43	0.36	0.41	0.44	0.43	0.43	0.65	0.38	0.30	0.45	0.40	Critically Dry

Year	Empirical	Alt1	Alt2woTUCPAllVA	Alt2woTUCPDeltaVA	Alt2woTUCPwoVA	Alt2wTUCPwoVA	Alt3	Alt4	EXP1	EXP3	NAA	Sacramento Valley Water Year Index
2015	0.66	0.65	0.55	0.63	0.63	0.62	0.73	0.54	0.41	0.60	0.55	Critically Dry

Table F.4-5. Geometric mean of predicted population growth rate (λ) across all years and binned into wetter and drier years for all alternatives. Empirical scenario indicates the LCME fit to observed data, while all alternative models represent simulations using CalSim output.

Category	Alt1	Alt2woTUCPAllVA	Alt2woTUCPDeltaVA	Alt2woTUCPwoVA	Alt2wTUCPwoVA	Alt3	Alt4	EXP1	EXP3	Empirical	NAA
1995-2015	0.69	0.94	0.94	0.91	0.90	1.16	0.88	0.96	1.02	1.42	0.93
Below Normal, Dry, or Critically Dry years	0.52	0.73	0.73	0.71	0.70	0.90	0.68	0.58	0.56	0.88	0.70
Wet and Above Normal years	0.96	1.24	1.23	1.21	1.21	1.53	1.18	1.68	1.98	2.40	1.28

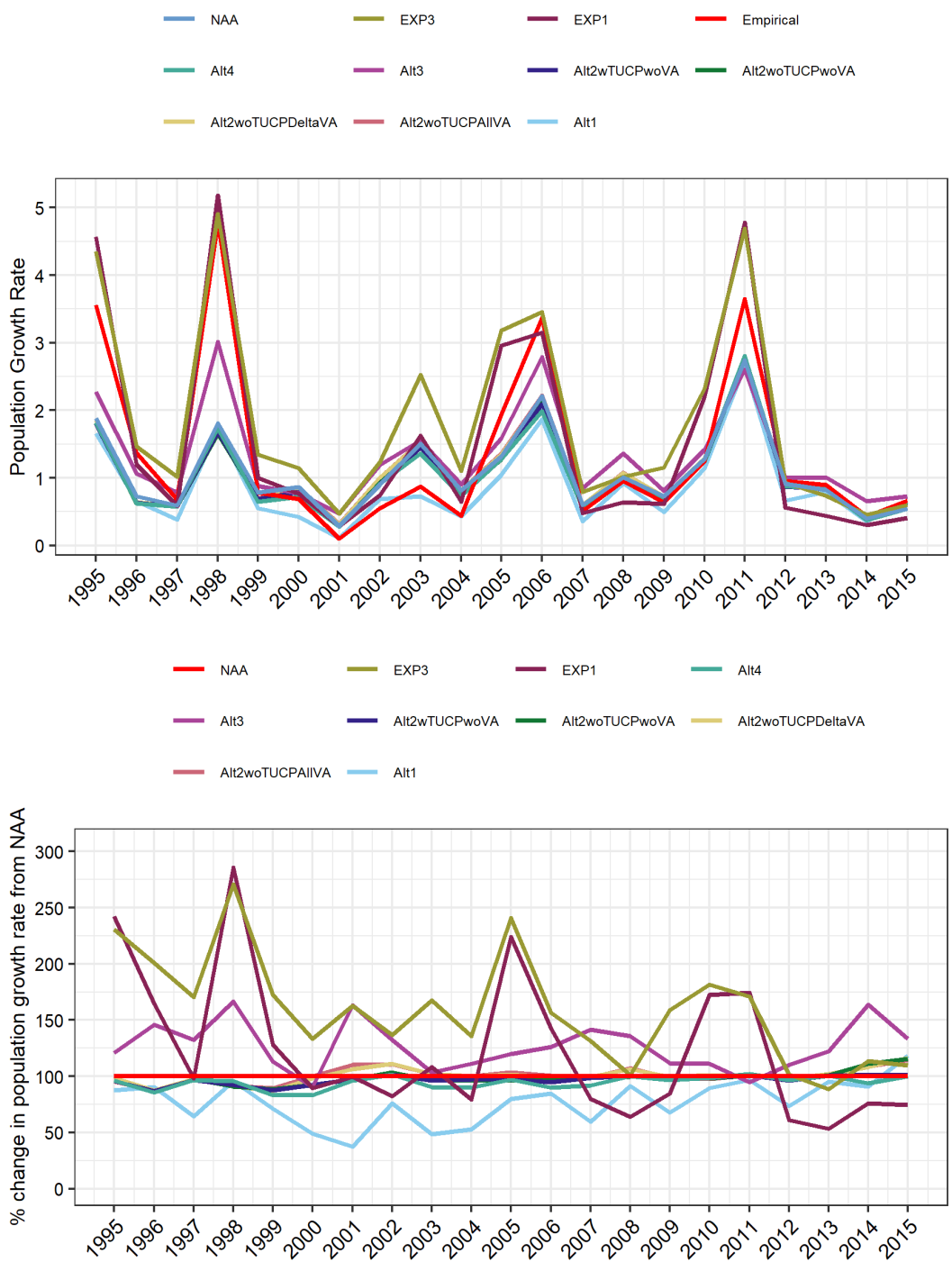


Figure F.4-8. Annual time series of delta smelt population growth rate. Top: Line plot of population growth rate (λ) across alternatives as seen in Table F.4-4. Bottom: Line plot showing % change calculated as λ for a given alternative divided by estimated population growth rate for NAA (no action alternative); no change from NAA = 100. Note the color change for NAA in the bottom figure.

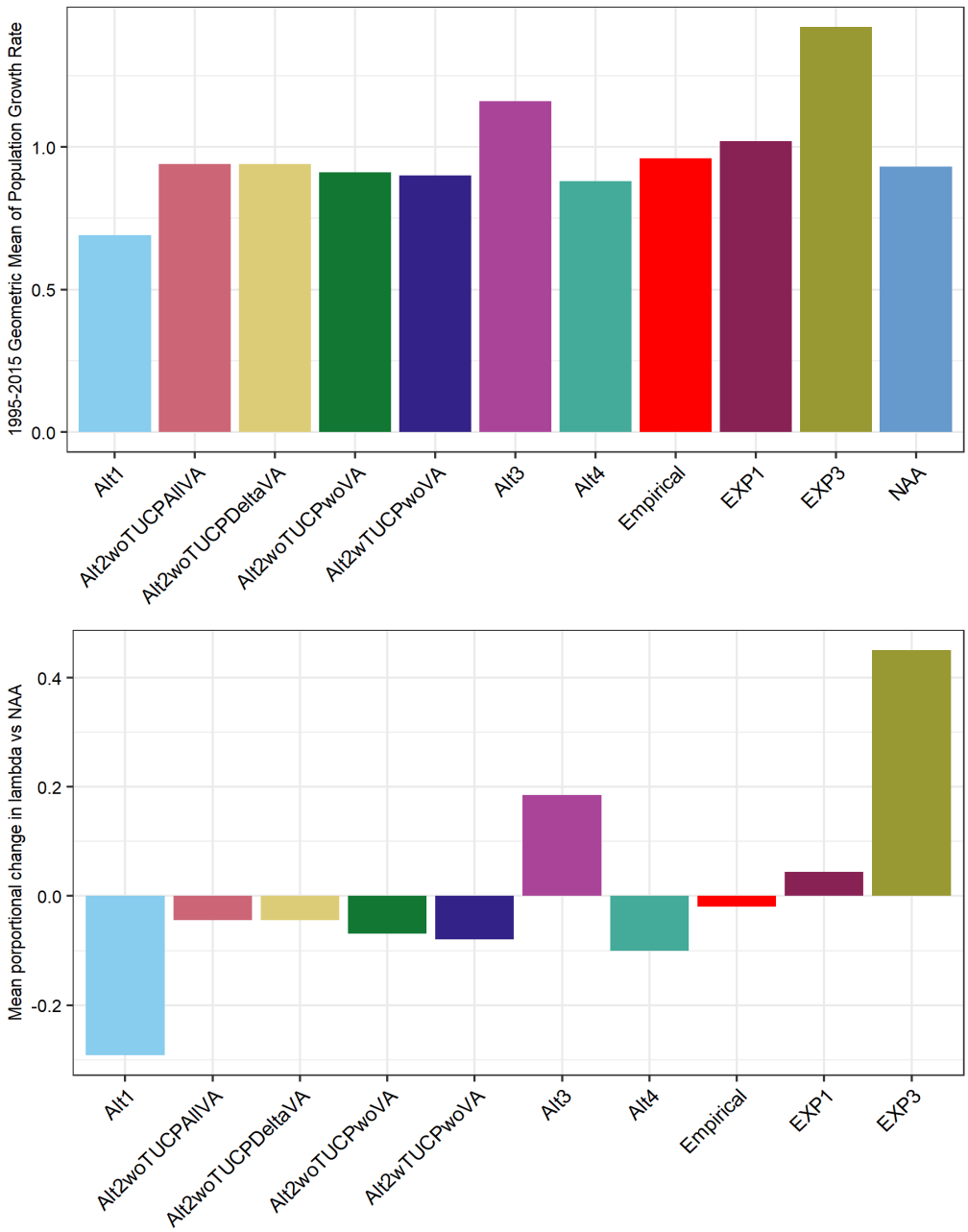


Figure F.4-9. Mean population growth rates aggregated across the years. Top: Bar plot demonstrating the geometric mean of population growth rate (lambda) from 1995 to 2015 for the various alternatives as seen in Table F.4-5. Bottom: Bar plot demonstrating the relative difference in geometric mean of population growth rate (1995-2015) for each alternative compared to the no action alternative ($[\lambda_{alternative} - \lambda_{no\ action}]/\lambda_{no\ action}$). Negative numbers indicate alternatives that result in poorer conditions for delta smelt and positive numbers indicate alternatives that are predicted to improve conditions.

F.4.3.1 Key Takeaways

- Geometric mean of population growth rate from 1995 to 2015 only showed considerable differences from the observed data and/or NAA for EXP3, Alt1, and Alt3 scenarios, where EXP3 and Alt3 performed better than most scenarios/alternatives (i.e., higher λ) and Alt1 performed worse than most alternatives (i.e., lower λ).
- EXP3 and Alt3 scenarios likely produced in higher λ due to more positive OMR flows for most months and the relatively high June-August Delta Outflow during dry years (Figure F.4-5, Figure F.4-6).
- Alt1 scenario likely produced lower λ relative to most scenarios due to the more negative OMR flows during most months (Figure F.4-5, Figure F.4-6).
- NAA, all components of Alt2, and Alt4 did not produce considerably higher λ than the empirical data despite OMR restrictions that should reduce entrainment. This may be due to how flow and hydrology are calculated in CalSim 3 that resulted in somewhat lower June-August Delta Outflow in wet years (Figure F.4-6).

F.4.3.2 BA Takeaways

The Delta Smelt Life Cycle Model with Entrainment Analysis (LCME) estimated values for larval recruitment and survival at the subsequent life stages (Smith et al. 2021). The most statistically supported model used means of December-June Old and Middle River (OMR) values, June-August outflow aggregated from monthly values or longer timescales, and aggregated food/prey metric from January to March. The model is used to calculate expected annual population growth rate (λ; the abundance of current year divided by abundance from previous year) as a performance measure of Delta seasonal flow operations influence on OMR and outflow over a twenty year time period (1995-2015).

The general statistical prohibition against extrapolation suggests that model predictions are more uncertain when explanatory variables are outside the range of observations to which the model was fit. Most CalSim-predicted flows and zooplankton predictions were not outside the range of observations to which the Delta smelt LCME was fit, but some alternatives did include out-of-range values. EXP1 included much lower June-August Delta Outflows than observed and higher (more positive) OMR values than observed in some years. EXP3 OMR values were similar to EXP1, but EXP3 June-August Delta Outflows were within the observed range. The multiple components of the PA also contained some April-May OMR values that were more negative than the observed range. Overall, CalSim-predicted June-August Outflow values were generally lower than the empirical data under Wet or Above Normal years. Predicted prey biomass for all alternatives was within the observed range, except for certain years where higher prey biomass were predicted than the empirical data for all alternatives.

The geometric mean of the expected population growth across years (1995-2015), λ, for the PA components ranged from 0.9 to 0.94 (Table F.4-6). The means of the expected population growth rate varied more widely across water year types, and showed positive growth rates under wetter meteorology and negative growth rates under drier meteorology. Note that wetter years also occurred with greater frequency at the beginning of the time series (1995-1999) compared to the end of the time series (2006-2015). Predicted flow and zooplankton conditions associated with

EXP3 resulted in the highest geometric mean λ (1.42), whereas NAA and the various components of PA produced geometric mean λ similar to the empirical data (0.9-0.94 vs. 0.96). While NAA and the various components of the PA resulted in higher λ than the empirical data during drier years, they also resulted in lower λ than the empirical data during wetter years (Table F.4-6).

NAA and the various components of the PA may have produced higher λ during drier years due to the more positive OMR values for multiple months and higher zooplankton estimates in February. Meanwhile, NAA and the PA components may have produced lower λ than the empirical data during wetter years because of the lower June-August Delta Outflow values and more negative OMR values for some months. NAA and the PA components did not produce higher λ despite OMR restrictions that should reduce entrainment of Delta smelt. This may be due to the apparent trade-off between OMR flow and summer Delta outflow that somehow occurred between PA components and the empirical data.

Table F.4-6. Geometric mean of predicted population growth rate (λ) across all years and binned into wetter and drier years for all alternatives. Empirical scenario indicates the LCME fit to observed data, while all alternative models represent simulations using CalSim output.

Category	EXP1	EXP3	NAA	PAwoTUCP woVA	PAwoTUCP DeltaVA	PAwoTUCP SystemwideVA	Empirical
1995-2015	1.02	1.42	0.93	0.91	0.94	0.94	0.96
Below Normal, Dry, or Critically Dry years	0.56	0.88	0.7	0.71	0.73	0.73	0.58
Wet and Above Normal years	1.98	2.4	1.28	1.21	1.23	1.24	1.68

F.4.3.3 EIS Takeaways

The Delta Smelt Life Cycle Model with Entrainment Analysis (LCME) produces estimated values for larval recruitment and survival at the subsequent life stages (Smith et al. 2021). The most statistically supported model used means of December-June Old and Middle River (OMR) values, June-August outflow aggregated from monthly values or longer timescales, and aggregated food/prey metric from January to March. The model is used to calculate expected annual population growth rate (λ; the abundance of current year divided by abundance from previous year) as a performance measure of Delta seasonal flow operations influence on OMR and outflow over a twenty year time period (1995-2015).

The general statistical prohibition against extrapolation suggests that model predictions are more uncertain when explanatory variables are outside the range of observations to which the model was fit. Most CalSim-predicted flows and zooplankton predictions were not outside the range of observations to which the Delta smelt LCME was fit, but some alternatives did include out-of-range values. Alternative 1, the multiple phases of Alternative 2, and Alternative 4 contained some April-May OMR values that were more negative than the observed range. Alternative 1 also contained OMR values more negative than the observed data for the months of December-

January and March. Overall, CalSim-predicted June-August Outflow values were generally lower than the empirical data under Wet or Above Normal years. Predicted prey biomass for all alternatives was within the observed range, except for certain years where higher prey biomass were predicted than the empirical data for all alternatives.

The geometric mean of the expected population growth across years (1995-2015), λ, for the Alternative 2 phases ranged from 0.9 to 0.94 (Table F.4-7). The means of the expected population growth rate varied more widely across water year types, and showed positive growth rates under wetter meteorology and negative growth rates under drier meteorology. Note that wetter years also occurred with greater frequency at the beginning of the time series (1995-1999) compared to the end of the time series (2006-2015). Predicted flow and zooplankton conditions associated with Alternative 3 resulted in the highest geometric mean λ (1.16), whereas conditions associated with Alternative 1 resulted in the lowest geometric mean λ (0.69). No Action Alternative, the various phases of Alternative 2, and Alternative 4 produced geometric mean λ similar to the empirical data (0.9-0.94 vs. 0.96). While No Action Alternative, the various phases of Alternative 2, and Alternative 4 resulted in higher λ than the empirical data during drier years, they also resulted in lower λ than the empirical data during wetter years (Table F.4-7).

No Action Alternative, the various phases of Alternative 2, and Alternative 4 may have produced higher λ during drier years relative to the empirical data due to the more positive OMR values for multiple months and higher zooplankton estimates in February. Meanwhile, No Action Alternative, the various phases of Alternative 2, and Alternative 4 may have produced lower λ than the empirical data during wetter years because of the lower June-August Delta Outflow values and more negative OMR values for some months. No Action Alternative, the various phases of Alternative 2, and Alternative 4 did not produce higher λ despite OMR restrictions that should reduce entrainment of Delta smelt. This may be due to the apparent trade-off between OMR flow and summer Delta outflow that somehow occurred between these alternatives and the empirical data.

Table F.4-7. Geometric mean of predicted population growth rate (λ) across all years and binned into wetter and drier years for all alternatives. Empirical scenario indicates the LCME fit to observed data, while all alternative models represent simulations using CalSim output.

Category	Alt1	Alt2wo TUCPwoVA	Alt2wTUCP woVA	Alt2woTUCP DeltaVA	Alt2woTUCP SystemwideVA	Alt3	Alt4	Empirical	NAA
1995-2015	0.69	0.91	0.9	0.94	0.94	1.16	0.88	0.96	0.93
Below Normal, Dry, or Critically Dry years	0.52	0.71	0.7	0.73	0.73	0.9	0.68	0.58	0.7
Wet and Above Normal years	0.96	1.21	1.21	1.23	1.24	1.53	1.18	1.68	1.28

F.4.4 References

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[FR Doc. 2024–16570 Filed 7–29–24; 8:45 am]

BILLING CODE 6560–50–P

DEPARTMENT OF HEALTH AND HUMAN SERVICES

42 CFR Part 71

[Docket No. CDC–CDC–2023–0051]

RIN 0920–AA82

Control of Communicable Diseases; Foreign Quarantine: Importation of Dogs and Cats; Correction

AGENCY: Centers for Disease Control and Prevention (CDC), Department of Health and Human Services (HHS).

ACTION: Final rule; correction.

SUMMARY: The Centers for Disease Control and Prevention (CDC) in the Department of Health and Human Services (HHS) announces a technical correction to the final rule published on May 13, 2024, regarding the importation of dogs and cats into the United States. The final rule contained a technical error. HHS/CDC is therefore publishing this amendment to the final rule correcting an error in amending instructions to the Office of the Federal Register.

DATES: Effective on August 1, 2024.

FOR FURTHER INFORMATION CONTACT: Ashley C. Altenburger, J.D., Division of Global Migration Health, Centers for Disease Control and Prevention, 1600 Clifton Road NE, MS–H16–4, Atlanta, Georgia 30329. Telephone: 1–800–232–4636.

SUPPLEMENTARY INFORMATION: On May 13, 2024, HHS/CDC published a final rule (89 FR 41726) that included a technical error. Therefore, HHS/CDC is publishing this notice to correct the technical error that was made in the final rule.

Section 553(b)(B) of the Administrative Procedure Act (APA), 5 U.S.C. 553(b)(B), provides that, when an agency for good cause finds that notice and public procedure are impracticable, unnecessary, or contrary to the public interest, the agency may issue a rule without providing notice and an opportunity for public comment. We have determined that it is unnecessary to provide prior notice and the opportunity for public comment because the technical correction being made, as discussed below, address only a minor publication error that does not substantially change agency actions taken in the final rule. For the same reasons we find good cause to make the correction effective on publication.

Summary of the Technical Correction to 42 CFR Part 71—Foreign Quarantine; Importation of Dogs and Cats

The final rule contains instructions to the Office of the Federal Register explaining how § 71.51, Dogs and cats, should appear once published in the Code of Federal Regulations. In amending instruction 3.j. to § 71.51, appearing at 89 FR 41837, HHS/CDC included instructions ''adding paragraphs (h) through (ff).'' However, the final rule contained updated provisions through paragraph (gg) and should have indicated that HHS/CDC is ''adding paragraphs (h) through (gg).'' We are therefore making this technical correction to ensure that paragraph (gg) is published in the Code of Federal Regulations as HHS/CDC intended and as discussed in the final rule.

Correction

For the reasons noted above, in FR Doc. 2024–09676, beginning on page 41726 in the **Federal Register** of Monday, May 13, 2024, the following correction is made:

§ 71.51 [Corrected]

■ 1. On page 41837, in the third column, in amendment 3.j. for § 71.51, the instruction ''Adding paragraphs (h) through (ff)'' is corrected to read ''Adding paragraphs (h) through (gg)''.

Elizabeth Gramling,

Executive Secretary, Department of Health and Human Services.

[FR Doc. 2024–16681 Filed 7–29–24; 8:45 am]

BILLING CODE 4163–18–P

DEPARTMENT OF DEFENSE

Defense Acquisition Regulations System

48 CFR Part 236

[Docket DARS–2024–0019]

RIN 0750–AM16

Defense Federal Acquisition Regulation Supplement: Architect and Engineering Service Fees (DFARS Case 2024–D019); Delay of Effective Date

AGENCY: Defense Acquisition Regulations System, Department of Defense (DoD).

ACTION: Final rule; delay of effective date.

SUMMARY: DoD is postponing the effective date of the final rule published in the **Federal Register** on June 27, 2024. As published, the rule was to be effective August 26, 2024.

DATES: The effective date for the final rule published June 27, 2024, at 89 FR 53502, is delayed from August 26, 2024, to September 16, 2024.

FOR FURTHER INFORMATION CONTACT: Ms. Jennifer D. Johnson, telephone 703–717–8226.

SUPPLEMENTARY INFORMATION: On June 27, 2024, DoD published a final rule amending the Defense Federal Acquisition Regulation Supplement (DFARS) to implement section 2881 of the National Defense Authorization Act for Fiscal Year 2024 (Pub. L. 118–31). Section 2881 increased the statutory limitation on the amount that may be earned by contractors providing certain architect and engineering services under contracts with the Departments of the Army, Navy, and Air Force. The effective date of the final rule has been postponed from August 26, 2024, to September 16, 2024, to comply with the Congressional Review Act; the final rule is a major rule as defined by 5 U.S.C. 804.

Jennifer D. Johnson,

Editor/Publisher, Defense Acquisition Regulations System.

[FR Doc. 2024–16715 Filed 7–29–24; 8:45 am]

BILLING CODE 6001–FR–P

DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

50 CFR Part 17

[Docket No. FWS–R8–ES–2022–0082; FXES1111090FEDR–245–FF09E21000]

RIN 1018–BG07

Endangered and Threatened Wildlife and Plants; Endangered Species Status for the San Francisco Bay-Delta Distinct Population Segment of the Longfin Smelt

AGENCY: Fish and Wildlife Service, Interior.

ACTION: Final rule.

SUMMARY: We, the U.S. Fish and Wildlife Service (Service), determine endangered species status under the Endangered Species Act of 1973, as amended (Act), for the San Francisco Bay-Delta distinct population segment (DPS) of longfin smelt (*Spirinchus thaleichthys*), a fish species of the Pacific Coast. This rule extends the protections of the Act to this DPS and adds it to the List of Endangered and Threatened Wildlife.

DATES: This rule is effective August 29, 2024.

ADDRESSES: This final rule is available on the internet at *https://www.regulations.gov.* Comments and materials we received on the proposed rule are available for public inspection at *https://www.regulations.gov* at Docket No. FWS–R8–ES–2022–0082.

Availability of supporting materials: Supporting materials we used in preparing this rule, such as the species status assessment report, are available at *https://www.regulations.gov* at Docket No. FWS–R8–ES–2022–0082.

FOR FURTHER INFORMATION CONTACT: Donald Ratcliff, Field Supervisor, U.S. Fish and Wildlife Service, San Francisco Bay-Delta Fish and Wildlife Office, 650 Capitol Mall, Suite 8–300, Sacramento, CA 95814; telephone 916–930–5603. Individuals in the United States who are deaf, deafblind, hard of hearing, or have a speech disability may dial 711 (TTY, TDD, or TeleBraille) to access telecommunications relay services. Individuals outside the United States should use the relay services offered within their country to make international calls to the point-of-contact in the United States.

SUPPLEMENTARY INFORMATION:

Executive Summary

Why we need to publish a rule. Under the Act, a species (including a distinct population segment of a species) warrants listing if it meets the definition of an endangered species (in danger of extinction throughout all or a significant portion of its range) or a threatened species (likely to become endangered within the foreseeable future throughout all or a significant portion of its range). If we determine that a species warrants listing, we must list the entity promptly and designate the species' critical habitat to the maximum extent prudent and determinable. We have determined that the San Francisco Bay-Delta distinct population segment (DPS) of longfin smelt (hereafter Bay-Delta longfin smelt) meets the definition of an endangered species; therefore, we are listing it as such. Listing a species as an endangered species can be completed only by issuing a rule through the Administrative Procedure Act rulemaking process (5 U.S.C. 551 *et seq.*).

What this document does. This rule lists the Bay-Delta longfin smelt as an endangered species under the Endangered Species Act and adds the Bay-Delta longfin smelt to the List of Endangered and Threatened Wildlife in title 50 of the Code of Federal Regulations at 50 CFR 17.11(h).

The basis for our action. Under the Act, we may determine that a species is an endangered or threatened species because of any of five factors: (A) The present or threatened destruction, modification, or curtailment of its habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) the inadequacy of existing regulatory mechanisms; or (E) other natural or manmade factors affecting its continued existence. We have determined that the Bay-Delta longfin smelt is endangered due to the following threats: altered hydrology (Factor A; largely attributable to water management, including water diversions and channel modifications), nonnative species (Factors C and E), and the effects of climate change (Factor E; by exacerbating drought, decreasing river and stream flows, and increasing air and water temperatures).

Section 4(a)(3) of the Act requires the Secretary of the Interior (Secretary), to the maximum extent prudent and determinable, to designate critical habitat concurrent with listing. We will publish a proposed critical habitat rule in a future edition of the **Federal Register**.

Previous Federal Actions

On October 7, 2022, the proposed rule to list the Bay-Delta longfin smelt published in the **Federal Register** (87 FR 60957). On February 27, 2023, we published a document in the **Federal Register** (88 FR 12304) that announced a public hearing and reopened the comment period for the proposed rule. Please see the proposed listing rule for a detailed description of previous Federal actions concerning this species.

On December 22, 2023, San Francisco Baykeeper (Baykeeper), a nonprofit corporation, filed a complaint in the U.S. District Court for the Northern District of California (*San Francisco Baykeeper* v. *United States Fish and Wildlife Service, et al.* (Case 4:23–cv–06601–LB)) challenging the Service's failure to complete its statutory obligations to make a final listing determination for the Bay-Delta longfin smelt. On April 5, 2024, the court issued an order requiring us to submit the final listing determination to the Office of the Federal Register by July 22, 2024. This document satisfies our requirement for completion of our final listing rule.

Peer Review

A species status assessment (SSA) team prepared an SSA report for the Bay-Delta longfin smelt. The SSA team was composed of Service biologists, in consultation with other species experts including those from the California Department of Fish and Wildlife. The SSA report represents a compilation of the best scientific and commercial data available concerning the status of the Bay-Delta longfin smelt, including the impacts of past, present, and future factors (both negative and beneficial) affecting the species.

In accordance with our joint policy on peer review published in the **Federal Register** on July 1, 1994 (59 FR 34270), and our August 22, 2016, memorandum updating and clarifying the role of peer review of listing actions under the Act, we solicited independent scientific review of the information contained in the draft Bay-Delta longfin smelt SSA report (Service 2021, entire). As discussed in the proposed rule, we sent the 2021 SSA report to five independent peer reviewers and received three responses. The peer reviews can be found at *https://www.regulations.gov* at Docket No. FWS–R8–ES–2022–0082 under supporting documents. In preparing the proposed rule, we incorporated the results of these reviews, as appropriate, into a revised draft SSA report (Service 2022, entire) and made this 2022 version available to the public on *https://www.regulations.gov* at Docket No. FWS–R8–ES–2022–0082 during the open comment periods. We have since incorporated any additional edits as appropriate based on public comments received during the public comment periods into the current SSA report, which is the foundation for this final rule (Service 2024, entire). A summary of the peer review comments and our responses can be found in the Summary of Comments and Recommendations, below.

Summary of Changes From the Proposed Rule

Since the October 7, 2022, proposed rule and February 27, 2023, document reopening the comment period and announcing a public hearing were published, we received comments from the public on the proposed listing rule and 2022 SSA report. After the second comment period closed on March 29, 2023, we also received additional comments on August 16, October 23, and December 12, 2023, from representatives of the Coalition for a Sustainable Delta. The Coalition also provided comments during the open comment periods for the proposed rule to *https://www.regulations.gov* at Docket No. FWS–R8–ES–2022–0082 (see documents FWS–R8–ES–2022–0082–0009 and –0024).

After reviewing all the information we received, we updated and revised the 2022 SSA report to incorporate any grammatical edits, clarifications, and

formatting changes. We also revised the count-based population viability analysis (see Service 2024, appendix B) based on information received from comments on the proposed rule and peer review of the information associated with publication of the analysis (Tobias et al. 2023, entire) in a scientific journal. The revisions to the analysis were associated with our methodology of estimating population growth rates and assisted in accounting for observation error in our estimates. Additionally, we also compared density-dependent model formulations to confirm sufficiency of our density-independent model. The changes made to the SSA report and appendix B did not significantly change our determination of status of the Bay-Delta longfin smelt in this final rule.

Additionally, in the October 7, 2022, proposed listing rule, we determined the designation of critical habitat for the DPS to be not determinable due to a lack of necessary information to complete our analysis. We are currently in the review process of determining critical habitat for the DPS, and a proposed rule will be forthcoming (see Critical Habitat, below).

Summary of Comments and Recommendations

In the proposed rule published on October 7, 2022 (87 FR 60957), and reopening document published on February 27, 2023 (88 FR 12304), we requested that all interested parties submit written comments on the proposal by December 6, 2022, and March 29, 2023, respectively. We also contacted appropriate Federal and State agencies, scientific experts and organizations, and other interested parties and invited them to comment on the proposal. A newspaper notice inviting general public comment was published in the Sacramento Bee on October 10, 2022 (McClatchy 2022, entire). On November 21, 2022, we received a written request from the public for a public hearing (Barajas et al. 2022, entire). We held a virtual public hearing on the proposed listing rule on March 14, 2023, as described in our February 27, 2023, **Federal Register** document (88 FR 12304 at 12305) reopening the comment period and announcing the virtual public hearing. All substantive information received during both comment periods or subsequently has either been incorporated directly into this final determination as appropriate or is addressed below.

Peer Reviewer Comments

As discussed in Peer Review above, we received comments from three peer reviewers on the draft SSA report (Service 2021, entire). We reviewed all comments we received from the peer reviewers for substantive issues and new information regarding the information contained in the 2021 SSA report. The peer review comments primarily fell into two main categories: (1) comments that provided grammatical or editorial corrections or minor clarifications of fact or that had no suggestions for changes to the SSA report but were either just expressing agreement or opposition and (2) comments that would affect the interpretation of available data and information presented in the SSA report. Peer review greatly assisted us to clarify our presentation of the substantial body of scientific information on Bay-Delta longfin smelt in the revised draft SSA report (Service 2022, entire) and the current SSA report (Service 2024, entire). Peer reviewer comments are addressed in the following summary. As discussed above, because we conducted this peer review prior to the publication of our proposed rule, we had already incorporated all applicable peer review comments into 2022 version of the SSA report, which was the foundation for the proposed rule. Additional comments received from the public have also been incorporated into the current version of the SSA report as necessary (Service 2024, entire) and together they are the foundation of this final rule.

(1) Comment: One peer reviewer stated that use of survey indices is not a good metric for data analysis and suggested using catch data instead.

Our response: We acknowledge that survey indices are not absolute census measures. However, we did compare existing Bay-Delta longfin smelt catch data (catch-per-unit-effort (CPUE) values from Stompe et al. 2020, entire) to the survey indices used in our analysis and got very similar results to what is presented in appendix B of the 2022 and 2024 SSA reports (Service 2022, pp. 110–123, Service 2024, pp. 115–128). As noted in the introduction of the technical note, CPUE and the Interagency Ecological Program (IEP) estimates both constitute indices of Bay-Delta longfin smelt abundance. Our use of abundance indices was not to determine the exact number of Bay-Delta smelt individuals but to use the multiple survey indices to determine trend information for the Bay-Delta longfin smelt. We consider the multiple survey efforts, which use varying sampling methods, as an appropriate measure to determine such trend information.

(2) Comment: One peer reviewer mentioned that our analysis included a ''temporal bias'' in the 20-mm and fall midwater trawl (FMWT) survey data (referring to the ''wet'' period starting the data series compared to the recent dry period). The reviewer suggested that we consider comparing wet periods to wet periods and dry periods to dry periods to more appropriately demonstrate the decline.

Our response: The section of the draft SSA report referred to by the reviewer (Service 2021, pp. 124–158) was intended to be a descriptive discussion on the statistical analysis of the abundance indices by the various survey efforts through time. In the revised draft SSA report (Service 2022, pp. 107–192), we revised the section to focus more on the descriptive statistics without calculating percent declines to avoid any possible perception of false precision within a decades-long data series. In the population viability analysis presented in our revised draft SSA report (Service 2022, appendix B) and now the updated modeling effort (the analysis has since published as Tobias et al., 2023, entire) described in appendix B of the current SSA report (Service 2024, appendix B), we interpret the full-time course of the suite of available data that includes any variance between wet or dry years.

(3) Comment: One peer reviewer suggested that the geographic and depth bias to FMWT make it ineffective as a survey for longfin smelt due to the fact that the FMWT does not sample the entire estuary or the entire water column.

Our response: The comment correctly points out that there are differences in spatial coverage both geographically and within the water column amongst the various survey efforts. However, we did not rely entirely on the FMWT as the only information in our analysis. To avoid any bias from any one survey effort, we used the information from all survey efforts in our modeling and overall analysis. Because none of the currently existing long-term monitoring surveys' sampling designs provides estuary-wide and full water column coverage, we consider that our methodology as identified in appendix B of the current SSA report (Service 2024) to determine trend information and extinction risk is a robust technical analysis as it helps reduce the effects of inflated variance through inverse weighting and is based on the best available scientific information available.

(4) Comment: One peer reviewer disagreed with our description of the importance of freshwater flow into the San Francisco Bay estuary and cited an analysis that indicated that freshwater flow was not the primary factor contributing to the decline of Bay-Delta longfin smelt (Phillis 2019, entire). The peer reviewer points to information indicating that freshwater flow into the estuary did not cause as substantial population rebound during recent wet periods as has been observed in prior decades and that the relationship between freshwater flow and population abundance is weakening (Tamburello et al. 2019, entire). The peer reviewer further points out that juvenile-to-adult survival was not significantly affected by freshwater flow into the estuary (Nobriga and Rosenfield 2016, entire).

Our response: We acknowledge that the relationship between increased freshwater flow and population abundance has recently been found to be decreasing. As a result, we have amended the SSA report to state that freshwater flow, while a primary driver of abundances, is not necessarily the primary driver influencing abundance of the current population (Service 2024, pp. 28–43). However, the peer reviewer's statement overlooks the influence of a massively declining adult population on the abundance of offspring that can be produced when favorable freshwater flow conditions exist. Other factors such as the size of the current adult breeding population and food resource limitations also play a role in the current status of the DPS, and we made clarifications in the current SSA report to also point to these other factors (Service 2024, pp. 35–36).

Public Comments

Influences on Water Temperature and Salinity

(5) Comment: We received several comments contending that the 2022 SSA report and proposed rule mischaracterize the relationship between water temperatures and San Francisco Bay Delta outflow. The commenters stated that estuary water temperatures are governed by a multitude of complex factors related to prevailing atmospheric conditions and are not influenced by San Francisco Bay Delta outflow.

Our response: In the 2022 SSA report and proposed rule, we acknowledge the complexity of factors influencing water temperature in the San Francisco Bay Delta. We agree with commenters that estuary water temperatures are governed by a multitude of complex factors and that atmospheric forcing is the dominant factor in determining water temperature in the estuary. However, existing literature suggests increased inflow can influence San Francisco Bay Delta temperatures. Some studies have found evidence that increased San Francisco Bay Delta inflow can lead to cooler than predicted temperatures in the San Francisco Bay Delta over short timespans (Wagner et al. 2011, p. 551; Wagner 2012, p. 78). The current SSA report better clarifies the relationship between San Francisco Bay Delta inflow/outflow and water temperatures (Service 2024, pp. 27–28).

(6) Comment: One commenter suggested that our conclusions regarding water temperature conditions are not accurate. Specifically, the commenter stated that temperature conditions in the San Francisco Bay-Delta never exceed 20 degrees Celsius (°C) (68 degrees Fahrenheit (°F)) in the winter-spring in the low-salinity zone, suggesting that high water temperatures are not a threat to larval and post-larval Bay-Delta longfin smelt.

Our response: The commenter's statement that water temperatures never exceed 20 °C (68 °F) in the winter and spring within the low-salinity zone is incorrect. Water temperatures in May within the low-salinity zone have been found to exceed this temperature (California Department of Water Resources 2020, entire). Lab studies sampling Bay-Delta longfin smelt collected within the San Francisco Bay have shown that water temperatures above 20 °C (68 °F) cause molecular stress responses (Jeffries et al. 2016, entire) and that temperatures greater than 15 °C (59 °F) may impair the viability of larval Bay-Delta longfin smelt (Yanagitsuru et al. 2021, entire). Water temperatures are predicted to increase in the estuary as a result of climate change and are likely to be an important factor in the future condition of the DPS (Service 2024, pp. 63–71).

Food Limitation

(7) Comment: A commenter stated that the 2022 SSA report fails to support the conclusion that food limitation may act as a stressor on the Bay-Delta longfin smelt.

Our response: In our discussion of prey availability and the decline of the DPS in the 2022 SSA, we presented the current scientific understanding and our conclusions are based on the best scientific information available (Service 2022, pp. 35–36). We acknowledge in the current SSA report that the prey species *Eurytemora affinis* has not been statistically linked to the survival of larval longfin smelt (Service 2024, pp. 35–36). We also acknowledge in the 2024 SSA report that although DPS abundance was observed to decline following the overbite clam (*P. amurensis*) invasion, no direct statistical support for a bottom-up longfin-mysid link has been established (Service 2024, pp. 35–36). The two primary prey taxa of the DPS have substantially declined compared to historical levels when overbite clam was not present (Kimmerer and Orsi 1996, p. 412). Research into the invasion of the nonnative overbite clam into the estuary, although not definitive, does suggest there is a possible link between the invasion and the subsequent decline of longfin smelt (Kimmerer 2002, p. 47; Feyrer et al. 2003, pp. 284–286; Thomson et al. 2010, p. 1443) with more recent research finding a significant positive correlation between available prey biomass and feeding success of Bay-Delta longfin smelt in the estuary, suggesting prey availability could influence growth and survival (Barros et al. 2022, p. 1773). The inability to statistically link declines in prey directly with declines in the DPS does not eliminate the likelihood that prey abundances are somehow linked to survival, but it suggests that factors affecting survival are complex and they may be partly attributable to prey abundances.

Entrainment

(8) Comment: We received two comments regarding entrainment rates of the DPS as a result of water diversion from the estuary. One commenter suggested the entrainment rates cited in the 2022 SSA report were overestimates because estimates were based on data that did not cover the full geographic extent of DPS larvae within the San Francisco Bay-Delta. The other commenter stated that the 2022 SSA report underestimates entrainment in the studies cited in the 2022 SSA report because the models excluded juvenile/adult entrainment, underestimated the length of time larvae are vulnerable to entrainment, and failed to account for indirect mortality.

Our response: The research discussed in the 2022 SSA report regarding entrainment has since been published and is publicly available (Gross et al. 2022, entire; Kimmerer and Gross 2022, entire). The authors of these papers acknowledged longfin smelt extend beyond the geographic extent of the sampling scheme used in these studies and took measures to correct for this bias. The results of these studies suggest that under current Old and Middle Rivers (OMR) flow management strategies, entrainment of Bay-Delta longfin smelt has not been substantial

enough to affect DPS population dynamics. Estimates from these two studies currently represent the best commercial and scientific data available and are discussed in the current SSA report (Service 2024, pp. 41–43). A pilot study examining entrainment of larval smelts is ongoing and aims to answer some of the current uncertainties. Over the next few years, life cycle modeling work by the Service will better quantify the cumulative impact of entrainment of multiple life stages on the DPS.

Contribution of Bay Tributaries

(9) Comment: We received multiple comments expressing concerns regarding how the 2022 SSA report and proposed rule addressed portions of the population that inhabit areas outside of the sampling footprint of most long-term surveys, particularly areas in and around Bay tributaries. Some of these commenters claim the Service disregarded this portion of the population in our determination and ignored the contribution of San Francisco Bay-Delta tributaries and recent restoration efforts of these areas to the DPS's current abundance, pointing to recent research (*i.e.,* Lewis et al. 2019a and 2019b) as evidence of population redundancy. The commenters concluded that because the Service did not consider the DPS's use of these areas in evaluating abundance of the Bay-Delta longfin smelt, the DPS is not experiencing population declines as evidenced by continued inhabitation of all geographic units in its range, and, therefore, the DPS is not at substantial risk of extinction or extirpation in any portion of its documented range.

Our response: Both the 2022 and 2024 SSA reports frequently acknowledge the inhabitation of Bay tributaries and recognize the important role they may have in reproduction, particularly in wet years (Service 2022, pp. 12, 19, 24; Service 2024, pp. 12, 21, 25). However, substantial contributions of recruits from these tributaries appear to be limited to wet years, and typically the majority of suitable spawning and rearing habitat is still the estuary's major low-salinity zone, which is usually located between Suisun Bay and the Delta (Kimmerer et al. 2013, figure 2, p. 7; Lewis et al. 2019a, p. 3; Lewis et al. 2019b, p. 6). Targeted sampling of some of these habitats in Bay tributaries has begun only recently; therefore, no time series exist on the time scale required to analyze population trends in these habitats. As such, there was no practical method to include data from these limited studies to inform our statistical analyses. However, recruits that hatched in Bay tributaries are available to the San Francisco Bay Study (SFBS) once they enter the larger Bays, so population contributions from Bay tributaries should be captured by SFBS indices.

We agree with commenters that restoration of the Bay tributaries would provide valuable benefits to the DPS. However, these benefits would likely occur during the spawning and rearing season of wet years rather than during the extended summer/fall occupancy of these habitats as the commenters suggested. During the summer and fall when Bay-Delta longfin smelt are occupying these areas, they appear to be mostly inhabiting deeper habitat as evidenced by the higher catch in the otter trawl surveys compared to the mid-water trawl surveys (Rosenfield and Baxter 2007, p. 1586). As such, restoration of shallower tidal wetlands in the lower Bay tributaries may not provide suitable habitat during the summer and early fall, as temperatures in these shallow habitats can approach and exceed the thermal tolerance of Bay-Delta longfin smelt during these times.

We disagree with the statement that the Bay-Delta longfin smelt is not experiencing population declines as evidenced by continued inhabitation of all geographic units in its range. The complex hydrodynamics of the estuary results in planktonic larvae and post-larvae with limited mobility to be widely distributed throughout the estuary, regardless of whether the habitat is suitable for any length of time. Likewise, spawning adults appear to be primarily oriented to cool water temperatures, which results in inhabitation of the ephemerally cool temperatures of Bay tributaries (Lewis et al. 2019b, p. 19). Inhabitation and spawning in these waters beyond the wettest years likely acts as a population sink as opposed to a source, as was observed by lack of recruitment from these habitats in most years. Lack of successful recruitment in most years from these tributaries suggests these habitats do not provide meaningful population redundancy.

Outflow-Abundance Relationship

(10) Comment: Some commenters took issue with our discussion on the relationship between freshwater flow and Bay-Delta longfin smelt abundance and questioned the validity and predictive power of the outflow-abundance relationship pointing to recent years when observed abundance indices were below indices predicted by the relationship.

Our response: The relationship of freshwater flow and longfin smelt production has consistently been reaffirmed for decades based on the published literature (Stevens and Miller 1983, pp. 431–432; Jassby et al. 1995, p. 285; Kimmerer 2002, p. 47; Rosenfield and Baxter 2007, p. 1585; Sommer et al. 2007, p. 274; Kimmerer et al. 2009, p. 381; MacNally et al. 2010, p. 1422; Thomson et al. 2010, pp. 1439–1440; Maunder et al. 2015, p. 108; Nobriga and Rosenfield 2016, p. 53; Kimmerer and Gross 2022, fig. 2, p. 2735). Nonetheless, we acknowledge that freshwater outflow is not a perfect predictor of Bay-Delta longfin smelt abundance due to the complexity and variable nature of habitat within the estuary and Bay-Delta longfin smelt population dynamics. The 2022 and 2024 SSA reports acknowledge what may be step-declines (where populations decline to a lower abundance level and do not rebound to previous levels) or changes in the intercept of the relationship (Service 2022, pp. 35–37; Service 2024, pp. 35–37) and acknowledge the decreasing explanatory power of the flow-abundance model (Service 2022, p. 37; Service 2024, p. 37). The decline of adult stock and its resulting egg supply is the most parsimonious reason why observed indices have been below what were predicted from data in the increasingly distant past. The SSA reports illustrate the point that when declining spawning stock is considered in the outflow-abundance model, the explanatory power of the flow-abundance model did not degrade over time (Service 2022, fig. 3.3, p. 38; Service 2024, p. 38). Understanding the biological mechanisms behind the flow-abundance relationship is an ongoing topic of research for the Service and the broader scientific community in the estuary.

Low-Salinity Zone Mechanism

(11) Comment: We received a comment suggesting the expansion of the low-salinity zone due to increased freshwater flow should not be considered a mechanism by which freshwater flow might influence Bay-Delta longfin smelt productivity. The commenter cited Kimmerer et al. (2013) as evidence that the volume of low-salinity habitat in the estuary is unrelated to the DPS's abundance.

Our response: We agree that current literature suggests higher outflow expanding the volume of the low-salinity zone is inadequate in explaining the population growth observed during wet years. In the 2022 SSA report, we acknowledged that expansion of the low-salinity zone may be beneficial to larvae but did not suggest this

mechanism was a primary driver of the outflow-abundance relationship (Service 2022, p. 21). Dr. Kimmerer provided an additional comment on the proposed rule suggesting a predominant mechanism behind the outflow-abundance relationship may be a result of favorable larval transport and retention in the low-salinity zone and elevated prey concentrations occurring during periods of greater freshwater flow, resulting in higher survival and abundance. In the 2022 SSA report, we identified and discussed this mechanism as well as other postulated mechanisms that may contribute to the outflow-abundance relationship (Service 2022, p. 32). In the proposed rule summarizing the information from the 2022 SSA report, we may have underrepresented the transport and retention mechanism. We have further described this mechanism in the 2024 SSA report (Service 2024, pp. 21–22) and this final rule (see *Summary of Biological Status and Threat*s below).

(12) Comment: We received a comment regarding the habitat use of larval/post-larval Bay-Delta longfin smelt. The commenter cited Yanagitsuru et al. (2022), Rahman et al. (2023), and a presentation to the Estuarine Ecological Team by Levi Lewis (2023) as new evidence that the Bay-Delta longfin smelt larvae are distributed further downstream and at higher salinities than previously thought.

Our response: The 2024 SSA report acknowledges the range of salinities that larval smelt are known to utilize in the estuary (Service 2024, p. 34). The lab studies that the commenter cites are informative of the physiological salinity tolerances of larvae but are not necessarily indicative of habitat use within the estuary, as such studies ignore predation, hydrodynamics, and other relevant physical and ecological processes. In the wild, larval abundance has been shown to rapidly decline above 10 practical salinity unit (PSU) (Lewis et al. 2019b, p. 30), and peak recruitment and abundance occur in the range of 2 to 4 PSU (Hobbs et al. 2010, p. 564; Grimaldo et al. 2017, p. 8; Grimaldo et al. 2020, pp. 12–14).

DPS Status

(13) Comment: We received comments stating that the San Francisco Bay-Delta longfin smelt does not meet the criteria of a DPS according to the Service's policy because it is not ''markedly separated'' from the rest of the longfin smelt population and that the Bay-Delta population is regularly mixing with other populations. The commenter points to research conducted since our 2012 DPS determination regarding dispersal and connectivity between the Bay-Delta population and nearby populations farther north along the California coast.

Our response: Our DPS policy does not require absolute separation of a DPS from other members of its species, because this can rarely be demonstrated in nature for any population of organisms (61 FR 4724; February 7, 1996). Our determination that the Bay-Delta longfin smelt meets the criteria of a DPS was published in the April 2, 2012, **Federal Register** (77 FR 19756). In the 2022 and 2024 SSA reports as well as our proposed listing rule, we identified and considered more recent research and reaffirmed that the Bay-Delta longfin smelt is a valid DPS (87 FR 60958–60959; October 7, 2022); Service 2022, p. 10; Service 2024, p. 10). Studies that have examined longfin smelt genetics have all found evidence that the Bay-Delta population is distinct from other northern populations (Israel and May 2010, p. 230; Sağlam et al. 2021, p. 1793).

We acknowledge that Sağlam et al. (2021) found evidence of northern dispersal of some individuals from the Bay-Delta population, as evidenced by some shared genetic structure with smaller populations in Northern California estuaries and the Columbia River. However, the study detected no significant gene flow from any northern estuaries southward into the Bay-Delta population, suggesting gene flow is unidirectional in a northerly fashion. These findings suggest the Bay-Delta population is genetically isolated, as it does not appear to be receiving immigrants from any northern populations. Sağlam et al. (2021, pp. 1793, 1802) concluded that the Bay-Delta population was distinct and is likely an important source for maintaining nearby populations.

A recent study published after the 2022 SSA report examined other, much smaller, longfin smelt populations along the California coast (Brennan et al. 2022, entire). The authors determined that estuaries in proximity of the San Francisco Bay estuary may not be permanently inhabited by longfin smelt, and that the Bay-Delta population may therefore lack the resilience typically provided by metapopulations. The authors also noted that abundance and distribution of longfin smelt appears to have declined in other estuaries along the California coast (Brennan et al. 2022, p. 12). This information may further support the hypothesis that reduction of the Bay-Delta longfin smelt abundance decreases the DPS's contribution to outside populations.

Population Viability Analysis

(14) Comment: We received comments expressing numerous concerns regarding our population viability analysis (PVA) and its use of population survey indices rather than actual abundance estimates for our determination of the status for the Bay-Delta longfin smelt. Specifically, the commenters took issue that the PVA relies on population indices data, which they contend do not fully sample the entire water column or habitat strata and introduce too much uncertainty on the size of the population. The commenters state that, because of these issues, the Service may have underestimated the size of the population and therefore overestimated the impact of threats facing the DPS; and the Service cannot use the PVA to determine time of extinction or base a listing decision on such uncertain data.

Our response: As we described in the 2022 and 2024 SSA reports (Service 2022, appendix B, pp. 111–123; Service 2024, appendix B, pp. 115–128), exact population abundance information is not necessary. A count-based PVA can be applied to index values, where a population index represents some portion of the total population as long as the proportion of the population that is observed remains relatively constant over time (Morris and Doak 2002, p. 51). General interpretation of the abundance indices for longfin smelt or any other species also requires this assumption or a correction for major deviations or inconsistencies. In the SSA reports, we acknowledge the different limitations of the long-term surveys and utilize them collectively to reduce potential biases that may be present in any single survey. As we state in our description of the methods used to conduct the PVA (Service 2022, appendix B, pp. 111–123; Service 2024, appendix B, pp. 115–128), we incorporated and applied our analysis in the PVA using several datasets that index the abundance of Bay-Delta longfin smelt, and these data capture the landscape of the available information regarding the estimation of abundance for the DPS.

In our review of the status of the Bay-Delta longfin smelt, we used all the best commercial and scientific information available to make our determination. The PVA was just one of the many tools we used in our analysis. We consider the PVA to be one of many appropriate tools that provide useful information for our decision on the listing status of the DPS. PVAs are best suited to test a range of possible conditions or demographic assumptions to provide a range of likely fates for a population (Morris et al.

1999, p. 2). We conducted sensitivity analyses to examine the effects of different potential starting population sizes and different minimum viable population sizes (quasi-extinction thresholds). The sensitivity analyses showed that in most demographic scenarios tested, the population is at a high risk of quasi-extinction in the near future (Service 2024, figures 3 & 4, pp. 121 and 122). It is noteworthy that the count-based PVA presented in appendix B predicts the time to quasi-extinction under current environmental conditions and does not account for the worsening environmental conditions associated with increases in drought intensity or frequency, warming water temperatures, and sea-level rise that are occurring and are predicted to continue.

(15) Comment: Some public commenters noted that the meta-analysis was conducted on multiple surveys and as a result obscured data from the San Francisco Bay study otter-trawl (SFBS OT) age-1 survey. The commenters stated that the mean population growth rate for the SFBS OT age-1 survey estimated in the PVA indicated a trend of increasing abundance and therefore is inconsistent with the Service's finding that the DPS is currently in danger of extinction.

Our response: In an effort to include all available data and produce the best estimates of population growth rates, a meta-analysis was performed using all surveys. The meta-analysis presented in appendix B of the 2022 and 2024 SSA reports (a method proposed by a reviewer of the 2021 SSA report) illustrates how pooling information from the full suite of available survey data can help reduce uncertainty in our overall estimate of the mean population growth rate. Two important features of a meta-analysis are that the meta-mean will be closer to the population mean on average than any of the individual surveys (sample means) and the confidence interval will be narrower, making it a more precise estimator of the population mean than the estimates from the samples.

Based on review and comment of the meta-analysis (Service 2022, appendix B, pp. 111–123), we revised our methods slightly for estimating mean population growth rates for the DPS. As a result of this modification, we obtained different estimates for each survey than were reported in the 2022 SSA report (Service 2022, appendix B, p. 120). Results indicate that all of the abundance indices show long-term population declines, and all except for the SFBS age-0 otter trawl are statistically significant downward trends (Service 2024, appendix B, table 2). In both the original and revised analyses, seven of the eight surveys examined indicate a negative mean growth rate. To exclude evidence from all seven other surveys and base our decision off the single potentially positive survey time series would sacrifice a substantial volume of available data and bias the results of the analysis.

(16) Comment: Several commenters stated that the 2022 SSA report and proposed rule indicate confirmation bias by relying on data from the FMWT survey without acknowledging the limitations of those data or evaluating competing data from the SFBS OT age-1 survey, which indicates that Bay-Delta longfin smelt DPS abundance is not declining. They further state that the FMWT survey should not be used as the primary determinant due to its limited extent in areas surveyed and that it may underrepresent the abundance of the DPS due to its sampling methods.

Our response: In the 2022 and 2024 SSA reports, we do not rely on the FMWT data any more than any other survey and acknowledge limitations of all available surveys within the estuary (Service 2022, pp. 107–109; Service 2024, pp. 108–110). As stated above, we utilized information from the entire suite of surveys including the SFBS OT. As previously discussed, the commenter's statement that the SFBS OT data imply that the DPS is not declining is not supported by analyses of those data (Tobias et al. 2023, entire).

In the SSA reports, we acknowledge that the FMWT does not sample the entire range of Bay-Delta longfin smelt during the fall; however, the survey does sample all of the low-salinity zone and some of the mesohaline water west of the low-salinity zone when Bay-Delta longfin smelt return to the upper estuary in the fall and early winter (Service 2022, pp. 43–46; Service 2024, pp. 43–47). Research on the Bay-Delta longfin smelt has found that a significant proportion of age-0 (fish younger than 1 year of age) and older individuals inhabit the low-salinity zone prior to and during spawning, making FMWT sampling in this region a reasonable index for population abundance (Rosenfield and Baxter 2007, p. 1590). We also acknowledge that the FMWT does not specifically target benthic habitat; however, except for a few particularly deep sampling stations, fish within a geographic location can still be sampled by the FMWT whether they are in the middle or the bottom of the water column (Service 2022, p. 44; Service 2024, p. 44). The ability of the FMWT to track changes in the DPS population is evidenced by its high correlation with SFBS data (Rosenfield and Baxter 2007, p. 1590). Ultimately, both the FMWT and SFBS OT surveys use their own standardized methods that are capable of indexing changes in relative abundance.

(17) Comment: One commenter pointed to the increase of FMWT abundance indices in the years of 2021–2022 as evidence of population resilience and stated that the Service needs to evaluate this information.

Our response: In our 2022 SSA report, we included analyses of the 2021 FMWT abundance indices (Service 2022, appendix B, p. 119) and considered this information in our proposed rule to list the Bay-Delta longfin smelt. Because the 2022 index was calculated after we had concluded our analyses and published our proposed rule, we reviewed the 2022 indices in making our final determination. The Bay-Delta longfin smelt population has historically had highly variable population growth and declines, and such short-term population changes are not unexpected based on the trend information over the full FMWT survey effort (Service 2024, p. 47). Similar increases in abundance have occurred in 2000, 2006, 2011, 2013, and 2017; however, all of these increases resulted in abundance estimates that are well below those prior to the declines experienced in the 1980s when population numbers were several orders of magnitude greater than those currently experienced by the DPS.

(18) Comment: Several comments point to the wide confidence intervals of estimated growth rates in the count-based PVA and interpret these intervals as evidence that the Bay-Delta longfin smelt population may be stable or increasing.

Our response: The commenters are correct in their interpretation of the confidence intervals around the mean growth rates for individual surveys. Bay-Delta longfin smelt population growth rates are highly variable from year to year depending on prevailing environmental conditions and spawning stock size. The wide confidence intervals are a result of this variation where in some years the population grows even though in most years it declines. To account for this variability and differences in confidence intervals of the studies, we developed the meta-analysis to pool estimates of the population growth rates from the individual surveys to get a more precise estimate. Based on the overall population trend over the length of all the studies, we determined that the Bay-Delta longfin smelt population is in decline.

(19) Comment: One commenter provided a technical review of the analyses presented in the appendices of the 2022 SSA report and stated the information presented in appendix B required additional documentation of the evaluation of the assumptions, reconsideration of how the analysis accounts for sample error, and comparisons of density-independent and density-dependent formulations of the models. In response, the commenter provided their own meta-analysis utilizing alternative methods to capture uncertainty.

Our response: The analysis and text of appendix B has gone through revisions since the publication of the 2022 SSA report and some of the points and corrections identified by the commenter have been addressed in our current SSA report (Service 2024, appendix B). Regarding the commenter's alternative analysis, we could not fully incorporate it because the methods used are not sufficiently described for us to fully evaluate them. However, taking their reported results at face value, an increase in confidence limits would suggest a less stable population growth rate than was reported in the SSA report. This scenario would tend to produce a shorter time to quasi-extinction, likely offsetting the small increase in mean that they also report. Therefore, we conclude that this alternative analysis would also support a conclusion that the DPS is at risk of quasi-extinction.

Contemporary Versus Pre-Development Outflow

(20) Comment: We received a comment that there has been no statistically significant reduction in outflow throughout the winter-spring period, nor on an annual basis when comparing contemporary to pre-development conditions. The commenter also critiqued our use of Reis et al. (2019) in the SSA report, stating that the study failed to account for evapotranspiration that occurred prior to development in the estuary.

Our response: One study comparing the pre-development conditions and contemporary conditions of the Central Valley and potential changes in the annual average San Francisco Bay Delta outflow found that the outflow has not changed substantially (Fox et al. 2015, pp. 4267–4271). However, annual average San Francisco Bay Delta outflow is not biologically relevant to Bay-Delta longfin smelt. Contemporary water operations have resulted in less San Francisco Bay Delta outflow during the winter and spring months and increased outflow during the summer months (Hutton et al. 2017a, fig. 5, p. 2507; Gross et al. 2018, fig. 4, p. 10). Winter and spring months are when Bay-Delta longfin smelt in the estuary spawn and larvae rear in the low-salinity zone, and reduced outflow in the winter and spring months has been repeatedly linked to reduced juvenile Bay-Delta longfin smelt production (Stevens and Miller 1983, pp. 431–432; Jassby et al. 1995, p. 285; Kimmerer 2002, p. 47; Rosenfield and Baxter 2007, p. 1585; Sommer et al. 2007, p. 274; Kimmerer et al. 2009, p. 381; MacNally et al. 2010, p. 1422; Thomson et al. 2010, pp. 1439–1440; Maunder et al. 2015, p. 108; Nobriga and Rosenfield 2016, p. 53; Kimmerer and Gross 2022, fig. 2, p. 2735). The reductions in February, April, and May outflows have been primarily attributed to reductions in San Francisco Bay Delta outflow as a result of water diversions (Hutton et al. 2017b, table 3, p. 2523).

We do not agree with the commenter's critique of Reis et al. (2019). Differences in evapotranspiration rates are primarily a concern when comparing pre-development and post-development outflows in the estuary because extensive wetlands and floodplains pre-development theoretically increased evapotranspiration (reducing San Francisco Bay Delta outflow), but levees constructed during development largely disconnected floodplains and resulted in decreased evapotranspiration. By 1930, almost the entire San Francisco Bay Delta had been leveed and reclaimed (Whipple et al. 2012, p. 25). Reis et al. (2019) examined the years 1930–2018 when no substantial changes in estuary evapotranspiration occurred. As a result, the authors were able to reasonably assess the relative impact of water operations on San Francisco Bay Delta outflows.

Underestimate of Threats

(21) Comment: One commenter stated that the Service understated the risk to the DPS by not considering specific current proposals to increase diversions of freshwater from the San Francisco Bay estuary and its watershed and that if realized these projects could result in changes to outflow in the estuary and further impact the Bay-Delta longfin smelt. The commenter points to several proposed projects including the Delta Conveyance Project, Sites Reservoir, and the State Water Resources Control Board's voluntary agreement process.

Our response: In the 2022 SSA report and proposed rule, we identified reduced freshwater flow and diversion as one of the primary threats driving the current and future status of the Bay-Delta longfin smelt (Service 2022, pp. 28–30; 71–73). In our analysis of future conditions, we also acknowledged that changes in water demand may be more severe as the information used in our analysis identified impacts only out to the year 2030 due to the uncertainty and difficulty in accurately identifying changes with the necessary specificity (Knowles et al. 2018, p. 7638). However, due to the ongoing refinement of these proposed projects a specific assessment of each project was not possible within the timeframe for completion of our final rule. These projects, and any attendant risk or benefits to Bay-Delta longfin smelt, will be evaluated as they are implemented through appropriate regulatory processes, including section 7 consultations and/or section 10 permits, and future recovery planning and implementation for the Bay-Delta longfin smelt.

(22) Comment: One commenter stated that neither the 2022 SSA report nor the proposed rule evaluated the threat posed by harmful algal blooms (HABs) such as the one observed in the summer of 2022.

Our response: We agree that in our 2022 SSA report and proposed rule, we did not include HABs as a threat to the Bay-Delta longfin smelt or a growing stressor to fish populations in the estuary. We agree that marine and brackish water HABs, such as the bloom that occurred in parts of San Francisco Bay in 2022, could plausibly affect Bay-Delta longfin smelt individuals in localized areas, but the impact of such blooms on the DPS overall is not well studied, and the best available information to date does not support a conclusion that HABs are a substantial threat to the status of the DPS. However, we agree that recent events are concerning. As emerging threats are defined and better understood, they will inform future consultations, permits, and recovery planning.

I. Final Listing Determination

Background

The longfin smelt is a small fish species 9–11 centimeters (cm) (3.5–4.3 inches (in)) in length with a relatively short lifespan of approximately 2 to 3 years. The Bay-Delta longfin smelt DPS occupies the San Francisco Bay estuary and areas of the Pacific Ocean out to the Farallon Islands (see figure 1). A thorough review of the taxonomy, life history, and ecology of the Bay-Delta longfin smelt is presented in the current SSA report (Service 2024, pp. 6–23).

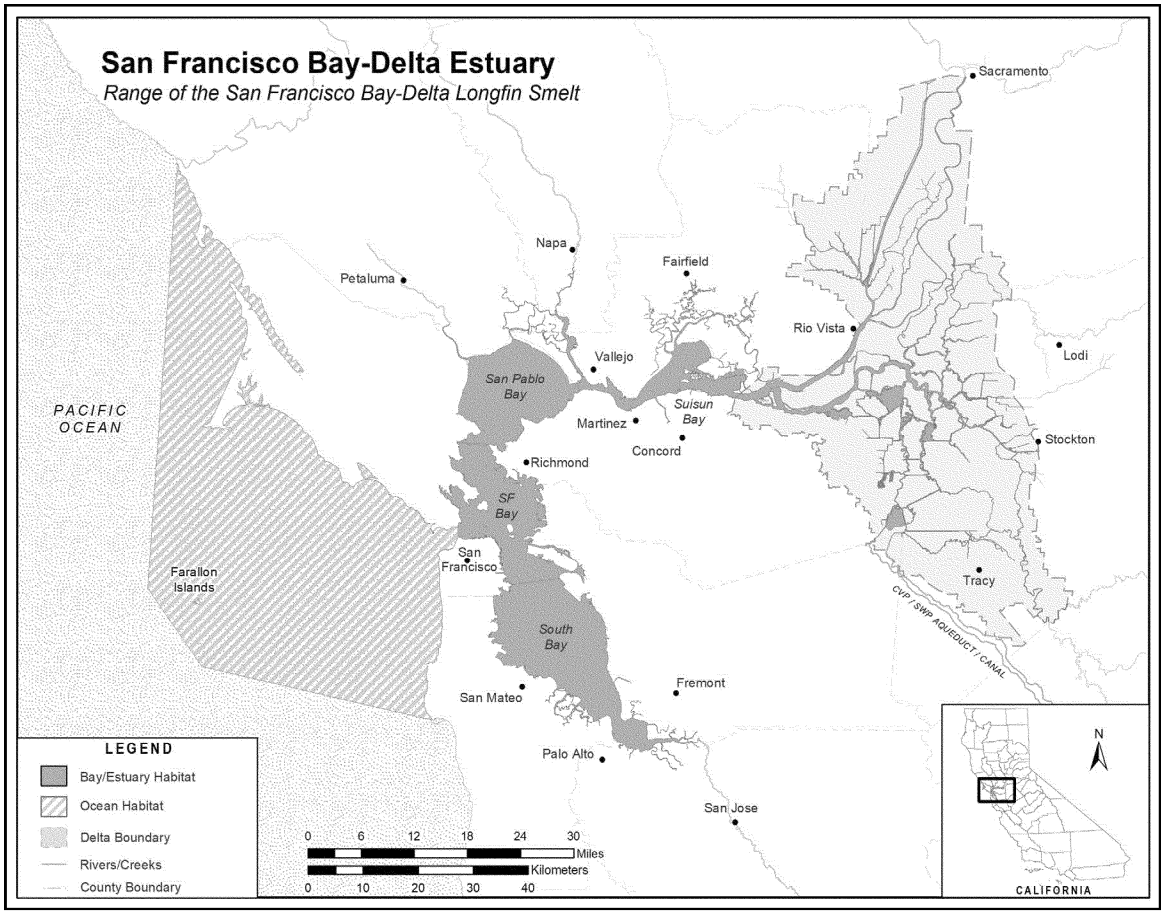


Figure 1. Range of the San Francisco Bay-Delta longfin smelt.

Regulatory and Analytical Framework

Regulatory Framework

Section 4 of the Act (16 U.S.C. 1533) and the implementing regulations in title 50 of the Code of Federal Regulations set forth the procedures for determining whether a species is an endangered species or a threatened species, issuing protective regulations for threatened species, and designating critical habitat for endangered and threatened species. On April 5, 2024, jointly with the National Marine Fisheries Service, the Service issued a final rule that revised the regulations in 50 CFR part 424 regarding how we add, remove, and reclassify endangered and threatened species and what criteria we apply when designating listed species' critical habitat (89 FR 24300). On the same day, the Service published a final rule revising our protections for endangered species and threatened species at 50 CFR part 17 (89 FR 23919). These final rules are now in effect and are incorporated into the current regulations. Our analysis for this final decision applied our current regulations. Given that we proposed listing this species under our prior regulations (revised in 2019), we have also undertaken an analysis of whether our decision would be different if we had continued to apply the 2019 regulations; we concluded that the decision would be the same. The analyses under both the regulations currently in effect and the 2019 regulations are available on *https://www.regulations.gov.*

The Act defines an ''endangered species'' as a species that is in danger of extinction throughout all or a significant portion of its range, and a ''threatened species'' as a species that is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. The Act requires that we determine whether any species is an endangered species or a threatened species because of any of the following factors:

(A) The present or threatened destruction, modification, or curtailment of its habitat or range;

(B) Overutilization for commercial, recreational, scientific, or educational purposes;

(C) Disease or predation;

(D) The inadequacy of existing regulatory mechanisms; or

(E) Other natural or manmade factors affecting its continued existence.

These factors represent broad categories of natural or human-caused actions or conditions that could have an effect on a species' continued existence. In evaluating these actions and conditions, we look for those that may have a negative effect on individuals of the species, as well as other actions or conditions that may ameliorate any negative effects or may have positive effects.

We use the term ''threat'' to refer in general to actions or conditions that are known to or are reasonably likely to negatively affect individuals of a species. The term ''threat'' includes actions or conditions that have a direct impact on individuals (direct impacts), as well as those that affect individuals through alteration of their habitat or required resources (stressors). The term ''threat'' may encompass—either together or separately—the source of the action or condition or the action or condition itself.

However, the mere identification of any threat(s) does not necessarily mean that the species meets the statutory definition of an ''endangered species'' or a ''threatened species.'' In determining whether a species meets either definition, we must evaluate all identified threats by considering the species' expected response and the effects of the threats—in light of those actions and conditions that will ameliorate the threats—on an individual, population, and species level. We evaluate each threat and its expected effects on the species, then analyze the cumulative effect of all of the threats on the species as a whole. We also consider the cumulative effect of the threats in light of those actions and conditions that will have positive effects on the species, such as any existing regulatory mechanisms or conservation efforts. The Secretary determines whether the species meets the definition of an ''endangered species'' or a ''threatened species'' only after conducting this cumulative analysis and describing the expected effect on the species.

The Act does not define the term ''foreseeable future,'' which appears in the statutory definition of ''threatened species.'' Our implementing regulations at 50 CFR 424.11(d) set forth a framework for evaluating the foreseeable future on a case-by-case basis, which is further described in the 2009 Memorandum Opinion on the foreseeable future from the Department of the Interior, Office of the Solicitor (M–37021, January 16, 2009; ''M-Opinion,'' available online at *https://www.fws.gov/library/collections/national-listing-and-classification-guidance*). The foreseeable future extends as far into the future as the U.S. Fish and Wildlife Service and National Marine Fisheries Service (hereafter, the Services) can make reasonably reliable predictions about the threats to the species and the species' responses to those threats. We need not identify the foreseeable future in terms of a specific period of time. We will describe the foreseeable future on a case-by-case basis, using the best available data and taking into account considerations such as the species' life-history characteristics, threat-projection timeframes, and environmental variability. In other words, the foreseeable future is the period of time over which we can make reasonably reliable predictions. ''Reliable'' does not mean ''certain''; it means sufficient to provide a reasonable degree of confidence in the prediction, in light of the conservation purposes of the Act.

Analytical Framework

The current SSA report documents the results of our comprehensive biological review of the best scientific and commercial data available regarding the status of the Bay-Delta longfin smelt, including an assessment of the potential threats to the DPS. The SSA report does not represent our decision on whether the DPS should be listed as an endangered or threatened species under the Act. However, it does provide the scientific basis that informs our regulatory decisions, which involve the further application of standards within the Act and its implementing regulations and policies.

To assess the Bay-Delta longfin smelt's viability, we used the three conservation biology principles of resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 306–310). Briefly, resiliency is the ability of the species to withstand environmental and demographic stochasticity (for example, wet or dry, warm or cold years); redundancy is the ability of the species to withstand catastrophic events (for example, droughts, large pollution events), and representation is the ability of the species to adapt to both near-term and long-term changes in its physical and biological environment (for example, climate conditions, pathogens). In general, species viability will increase with increases in resiliency, redundancy, and representation (Smith et al. 2018, p. 306). Using these principles, we identified the DPS's ecological requirements for survival and reproduction at the individual, population, and DPS levels, and described the beneficial and risk factors influencing the DPS's viability.

The SSA process can be categorized into three sequential stages. During the first stage, we evaluated the individual DPS's life-history needs. The next stage involved an assessment of the historical and current condition of the DPS's demographics and habitat characteristics, including an explanation of how the DPS arrived at its current condition. The final stage of the SSA involved making predictions about the DPS's responses to positive and negative environmental and anthropogenic influences. Throughout all of these stages, we used the best available information to characterize viability as the ability of the DPS to sustain populations in the wild over time. We use this information to inform our regulatory decision.

The following is a summary of the key results and conclusions from the current Bay-Delta longfin smelt SSA report (Service 2024, entire); the full SSA report can be found at Docket No. FWS–R8–ES–2022–0082 on *https://www.regulations.gov.*

Summary of Biological Status and Threats

In this discussion, we review the biological condition of the DPS and its resources, and the threats that influence the DPS's current and future condition, in order to assess the DPS's overall viability and the risks to that viability.

The needs of the Bay-Delta longfin smelt can be categorized into three main resource and biological condition categories, and include: (1) appropriate habitat salinity, by life stage; (2) appropriate habitat temperature conditions, by life stage; and (3) adequate food resources and availability, by life stage. As the Bay-Delta longfin smelt utilizes both freshwater and saline water conditions across its life cycle, its habitat is extremely variable. These variable conditions, along with other factors including the interaction among these dynamic variables, exert a strong influence on habitat suitability across space and time.

Bay-Delta longfin smelt have temperature tolerances that impact the volume and seasonality of suitable habitat. It is logical to presume that the Bay-Delta DPS, inhabiting as it does the most southern portion of the species' range, has historically been (and is) at the uppermost temperature tolerance range of the species. Larvae appear to be adapted to cool water conditions. Larvae inhabit water temperatures between 8 and 12 °C (46 and 54 °F) (Grimaldo et al. 2017, p. 8). Available research indicates that temperatures approaching and exceeding 15 °C (59 °F) impair larval viability, and hence this benchmark may be ecologically significant. At later larval stages, longfin smelt are still likely restricted to water temperatures below 20 °C (68 °F) (Jeffries et al. 2016, p. 1709). In general, age-1 and age-0 fish inhabit 16–18 °C (61–64 °F) water in summer and fall. Adults are thought to be limited by water temperature >22 °C (>72 °F) during the summer and likely spend the majority of this time in cooler Bay habitats and the open ocean (see Service 2024, p. 25). Fish return to spawn where water quality conditions are favorable for egg survival. These conditions vary in location depending on Delta outflow, as well as flows from Bay Area tributaries. Spawning occurs at or below water temperatures of 13 °C (55 °F), but has been documented at up to 16 °C (61 °F) (see Service 2024, p. 25).

Bay-Delta longfin smelt have a defined salinity tolerance range that

increases as fish mature. This is consistent with their anadromous life history requiring spawning and early rearing in fresher San Francisco Bay Delta and Bay tributary waters. Yolk-sac larvae survive the longest and grow the largest at 5 and 10 parts per thousand (ppth) and are able to maintain water balance equally between 0.4 and 10 ppth, but the same lifestage is unable to survive at 32 ppth (ocean salinity). Yolk-sac larvae cannot complete yolk resorption in freshwater. In field surveys, peak yolk-sac larval densities have been found at 2–4 ppth (Grimaldo et al. 2017, p. 8), which is also concordant with the early life salinities that produce highest survival to later life stages (Hobbs et al. 2010, p. 564). The California Department of Fish and Wildlife's (CDFW) 20-mm surveys have also shown peak larval distribution near this same salinity zone (*i.e.,* 2–4 ppth), although larvae have been detected in salinities as high as 12 ppth (see Service 2024, pp. 25–26). The 20-mm survey gets its name from the size at which Bay-Delta longfin smelt are retained and readily identifiable at the fish facilities associated with the State and Federal pumping facilities.

Available data clearly indicate a very strong association of Bay-Delta longfin smelt to turbid water conditions. Aquatic turbidity provides simultaneous feeding and predator avoidance advantages for larval fishes (Utne-Palm 2002, p. 115; Pangle et al. 2012, pp. 10–11). Turbidity enhances prey avoidance at detection distances typical of predatory fish species. Further, the sediment and algal particles often backlight relatively translucent zooplankton, helping larval fishes see these prey more easily (Utne-Palm 2002, p. 119). In contrast, larger fishes that may prey on fish larvae have longer search and reactive distances so more sediment and algal particles are in between these larger fish and their potential prey (Utne-Palm 2002, pp. 122–123). We assume that these turbidity mechanisms that apply broadly to larval fishes also apply to later life stages of Bay-Delta longfin smelt. Taken together, recent laboratory experiments, combined with field results, provide an assessment of early-life habitat needs favoring more turbid conditions (Utne-Palm 2002, entire; Pangle et al. 2012, entire).

Bay-Delta longfin smelt exhibit high prey-specificity. During the time that longfin smelt larvae and small juveniles are feeding in low-salinity habitats, they appear to focus on only two prey taxa. Smaller larvae appear to primarily use the copepod *Eurytemora affinis* as prey, while larger larvae and small juveniles appear to require mysids as prey. Longfin smelt pre-spawning adults in Suisun Marsh have shown a strong dietary preference for mysids, while relying on copepods and amphipods when mysids are scarce (see Service 2024, pp. 26–27). As longfin smelt exhibit very little variation in prey use, they are considered more susceptible to food web changes than some other fishes (Feyrer et al. 2003, p. 281). The current SSA report (Service 2024, p. 27) discusses some observations from various studies regarding affects to the low-salinity zone from invasion by the overbite clam (*Potamocorbula amurensis*) in the 1980s and potential impacts to the Bay-Delta longfin smelt as a food limitation stressor (see below, and also Service 2024, pp. 35–37).

Threats Influencing the Bay-Delta Longfin Smelt

The threats facing the Bay-Delta DPS of the longfin smelt include habitat alteration (Factor A) and changes to hydrology associated with reduced and altered freshwater flows (Factor A); increased water temperatures (Factor A); reduced food resource availability (Factor E); predation (Factor C); entrainment from freshwater diversion facilities (Factor E); and contaminants (Factor E). We consider reduced and altered freshwater flows resulting from human activities and impacts associated with current climate change conditions (increased magnitude and duration of drought and associated increased temperatures) as the main threat facing the Bay-Delta longfin smelt due to the importance of freshwater flows to maintaining the life-history functions and species needs of the DPS. However, because the Bay-Delta longfin smelt is an aquatic species and the needs of the species are closely tied to freshwater input into the estuary, the impact of many of the other threats identified above are influenced by the amount of freshwater inflow into the system (*i.e.,* reduced freshwater inflows reduce food availability, increase water temperatures, and increase entrainment potential).

Reduced and Altered Freshwater Flows

The development of dams and water delivery infrastructure built throughout the Sacramento and San Joaquin River basins for flood protection and water supply for agriculture and human consumption has greatly impacted freshwater flows into the San Francisco Bay estuary (Service 2024, section 3.1.1). The creation of this water storage and delivery system, where water is stored during the wet season and conveyed to farms and cities during the dry season, has resulted in one of the largest human-altered water systems in the world (Nichols et al. 1986, p. 569). Operation of this system has resulted in a broader, flatter hydrograph with less seasonal variability, thus changing the timing, magnitude, and duration of freshwater flows into the San Francisco Bay-Delta (Kimmerer 2004, p. 15; Andrews et al. 2017, p. 72; Gross et al. 2018, p. 8). It is estimated that the Federal and State water projects annually reduce an average of about 5 million acre-feet (MAF) of freshwater into the San Francisco Bay Delta, while other municipal or private reservoirs or diverters annually divert an additional 8 MAF of potential freshwater into the San Francisco Bay Delta (Hutton et al. 2017b, fig. 4, p. 2523). The cumulative effect of this annual average of about 13 MAF of freshwater supplies has resulted in a long-term decline in freshwater inflow into the estuary during the period of February through June relative to estimates of what flows would have been available absent water development (Gross et al. 2018, fig. 6, p. 12; Reis et al. 2019, fig. 3, p. 12). This situation has further increased the frequency of very low outflow years that, prior to water development, would have been very rare and associated only with extreme drought (Reis et al. 2019, fig. 3, p. 12).

From 1956 to the 1990s, water exports (water removed from the San Francisco Bay Delta as a result of State (State Water Project) and Federal (CVP) water projects) increased, rising from approximately 5 percent of the Delta freshwater inflow to approximately 30 percent of the Delta inflow (Cloern and Jassby 2012, p. 7). By 2012, an estimated 39 percent of the estuary's unimpaired freshwater flow in total was either consumed upstream or diverted from the estuary (Cloern and Jassby 2012, p. 8). Water exports continue to the present day and are expected to continue in the future.

A reduction in freshwater flows into the estuary influences and impacts the location and function of the low-salinity zone (spawning and rearing habitat for longfin smelt). Freshwater inflow into the estuary and other co-linear indicators of wet versus dry conditions during the winter and spring have been statistically associated with recruitment of larvae to the juvenile life stage of Bay-Delta longfin smelt (Service 2024, section 3.1.1). Prior to large-scale water exports and reduced freshwater flows, the location of the low-salinity zone (as represented by the 2 percent bottom salinity position, known as X2) reached the ≤55-km (≤34-mi) point in the estuary (monthly averages from February

through May) in about half of all years. More recently the position of X2 reaching at least the 55-km (34-mi) point occurred only very rarely as a result of wet year conditions (Gross et al. 2018, fig. 6, p. 12 and fig. 7, p. 13) (Service 2024, section 3.1.1). In the case of Bay-Delta longfin smelt, optimal growth and rearing conditions (food and water conditions (salinity, turbidity, circulation patterns)), especially for early life stage fish, is directly linked to freshwater inflow to the estuary.

Drought Conditions

California's precipitation patterns can be extremely variable, and several years of dry conditions have occurred over numerous extended periods resulting in varying levels of drought (California Department of Water Resources (DWR) 2020, entire). Drought periods can be characterized as having less freshwater flow, as well as shorter duration and lower magnitude of peak flows. The current trend in drought conditions has recently increased in frequency, duration, and magnitude (Swain et al. 2018, pp. 427–433). Prior to the 21st century, dry and critically dry years occurred approximately 33 percent of the time. However, since the year 2000, the dry and critically dry year frequency has increased to 43 percent. Based on soil moisture reconstruction, the period between 2000 and 2021 was probably the driest 22-year period on record (Williams et al. 2022, p. 1). As the existing impacts from climate change (*i.e.,* warmer temperatures) increase evapotranspiration in the watershed, the aforementioned water supply needs can exacerbate the magnitude of realized dry conditions over and above these natural patterns in precipitation and reduced San Francisco Bay Delta freshwater inflow.

Bay-Delta longfin smelt exhibit poor survival and reproduction during droughts (Thomson et al. 2010, pp. 1438–1446; Mahardja et al. 2021, pp. 9–10). The survival of Bay-Delta longfin smelt through their early life stages is lower during dry conditions and higher during wet conditions, as evidenced by Bay-Delta longfin smelt abundance indices nearly always declining sharply during dry periods then rebounding when wet weather returns (Mahardja et al. 2021, pp. 9–10). However, such recovery does not always occur after each drought cycle, leading to lower baseline numbers for the DPS (Moyle 2002, p. 237; Sommer et al. 2007, pp. 270–276). In addition, extended dry years compound the negative impacts to Bay-Delta longfin smelt as the DPS has not shown an ability to quickly recover and reoccupy upstream spawning habitats following drought. These drought conditions have exacerbated the impact of reduced freshwater flows from human activities and have been attributed to accelerating the establishment of the overbite clam (*Potamocorbula amurensis*) (see *Reduced Food Resources* and *Pelagic Organism Decline (POD),* below) by making saline water conditions more available throughout areas typically associated with more freshwater (Carlton et al. 1990, pp. 90–91).

Habitat Alteration

Large-scale habitat alteration such as channelization and dredging of streams and bays, building of levees and canals, and draining of wetlands has occurred since the 1850s. The impacts of such in-water and adjacent upland habitat alterations greatly affected and continues to impact the bathymetry of the estuary by collectively making the estuary deeper and less hydrodynamically connected to the surrounding landscape (Andrews et al. 2017, fig. 5, p. 64). The altered waterways create more space and avenues for the incoming tides to bring more saline water landward. Specifically, landscape changes since 1850 are estimated to have resulted in an average landward shift of X2 of over 3 km (2 mi) (Andrews et al. 2017, p. 68). This change along with reductions in freshwater input into the estuary (see *Reduced and Altered Freshwater Flows,* above) has caused a winter-spring upstream (landward) shift of X2 on the order of 10–20 km (6–12 mi). Taken together, the landscape changes discussed above and changes to the estuary's flow regime have changed how mixing processes function, and thus altered the habitat and food resource opportunities available for the estuary's biota, including the Bay-Delta longfin smelt.

Water Temperature Alterations

As described in the *Life History and Biology* section of the current SSA report (Service 2024, section 2.4), Bay-Delta longfin smelt spawning occurs within cool-water conditions below 16 °C (61 °F), while larvae and young juveniles show a preference for temperatures below 15 °C (59 °F) and 20 °C (68 °F), respectively. The embryonic through early juvenile life stages are when Bay-Delta longfin smelt are believed to be most vulnerable to warming temperatures because these early life stages do not possess the ability to migrate to the cooler waters of central San Francisco Bay and the coastal ocean due to limited motility and increases in potential predation. Subadults and adults are thought to be limited to water temperature below 22 °C (72 °F). Studies and datasets indicate water temperatures in the San Francisco Bay Delta commonly exceed 22 °C (72 °F) during the summer (Vroom et al. 2017, p. 9904; data from California Data Exchange Center, Central & Northern California Ocean Observing System, and U.S. Geological Survey (Blodgett et al. 2011, entire). Age-1 to age-3 individuals should possess the capacity to move to cooler waters during such times, making them less vulnerable than larvae and age-0 juveniles. However, climate change is predicted to substantially increase the number of days that water temperatures are inhospitable to all life stages, likely decreasing the duration of suitable spawning and larval rearing windows (Service 2024, section 4.2.2).

Reduced Food Resources

As discussed above and in the current SSA report (Service 2024, section 3.1.2), the Bay-Delta longfin smelt historically limited their diet to a relatively small number of crustacean meso- and macrozooplankton taxa. Bay-Delta longfin smelt larvae have diets dominated by a copepod, *Eurytemora affinis,* that is common in the low-salinity zone during the spring (CDFW, unpublished data). The two most common prey taxa for larger longfin smelt are epibenthic mysids and amphipods (Burdi 2022, pers. comm.; CDFW unpub. Diet Study Data). The copepod *E. affinis* was also at one time an important prey item for a now much-depleted mysid species, *Neomysis mercedis* (Knutson and Orsi 1983, p. 478), a prey species of juvenile and adult Bay-Delta longfin smelt.

Since the 1970s, the *Eurytemora affinis* population in the estuary has been in decline, but beginning in the late 1980s, the zooplankton community for the San Francisco Bay estuary started undergoing about a decade of rapid change in species composition, trophic structure, and utility for fish production (Winder and Jassby 2011, pp. 683–685; Kratina et al. 2014, p. 1070; Brown et al. 2016, p. 8). This decline coincided with the rapid invasion of the estuary by the nonnative overbite clam (Carlton et al. 1990, pp. 81 and 85, fig. 3) and with an extended drought in the Central Valley in the period 1987–1994 (Rosenfield and Baxter 2007, p. 1589).

The overbite clam is a filter feeder that is thought to have diverted food resources from the primary food sources of, or fed directly on, the nauplii (first larval stage) of common calanoid copepods and resulted in their decline. These native copepods are one of the

main sources of prey of larval Bay-Delta longfin smelt (Carlton et al. 1990, pp. 90–91; Kimmerer et al. 1994, p. 87; Feyrer et al. 2003, pp. 284–286; Rosenfield and Baxter 2007, p. 1589). Mysids have experienced an over tenfold decrease in abundance and accounted for less than 4 percent of total zooplankton biomass within the estuary after 1994 (Winder and Jassby 2011, p. 684). In addition to lower abundance, the average individual sizes of mysids in the estuary have decreased over time, with a species composition shift towards *Hyperacanthomysis longirostris,* an invasive species that reaches maturity at a smaller mass than *Neomysis* species (Hennessy 2011, entire). Although Bay-Delta longfin smelt consume these nonnative species, they are not preferred and the change in food resources most likely results in an increased effort for the DPS to meet its food resource needs.

To further exacerbate the impacts of the change in food resources, the decline of the Bay-Delta longfin smelt's historical prey base has not been accompanied by a large change in prey use by the DPS (Barros et al. 2019, p. 15; Feyrer et al. 2003, p. 285). This finding suggests that Bay-Delta longfin smelt had formed strong predator–prey interactions with their primary prey, a hypothesis supported by empirical data (MacNally et al. 2010, p. 1426). Because the DPS exhibits very little variation in prey use despite the reduction in natural prey availability, they are considered more susceptible to food web changes than some other fishes (Feyrer et al. 2003, p. 281). The decline in food resources is likely affecting juvenile and adult longfin smelt growth and fitness as well as increasing the effort needed to meet food resource demands (Kimmerer and Orsi 1996, pp. 418–419; Feyrer et al. 2003, p. 281).

Predation

In the proposed rule and 2024 SSA report, we discussed the role predation may play on individuals and various life stages of the Bay-Delta longfin smelt (see the 2024 SSA report and proposed rule for additional information). Because little information is available on the exact predators of the Bay-Delta longfin smelt or the impact predation has on the status of the DPS, we do not consider the impacts from predation to be a primary driver, but we still include this consideration as part of the cumulative impact from all threats for the DPS, especially during poor habitat conditions when food is lacking.

Entrainment

Freshwater diversion occurs throughout the estuary through intake structures for agricultural, municipal, and environmental purposes and in some cases may lead to entrainment of Bay-Delta longfin smelt. Entrainment occurs when the suction caused by pumping water creates an opportunity for fish to follow or be captured by the flow of water and become trapped and transported by the hydrodynamic footprint of those diversions. This entrainment may result in fish, especially early-life-stage fish, being killed or removed from the estuary. Bay-Delta longfin smelt can be entrained in water exported by the major pumping facilities in the South Delta (see *Water Project Exports,* below) when adults and commingling age-1 individuals move upstream into the freshwater portions of the San Francisco Bay Delta (CDFW 2020a, fig. 13, p. 53). Bay-Delta longfin smelt larvae and small juveniles that are either rearing or being tidally dispersed landward of X2 can also be entrained (CDFW 2020a, fig. 13, p. 53). During periods of high freshwater flow into the estuary, Bay-Delta longfin smelt (adults, juveniles, and larvae) are much less likely to be entrained by the major pumping facilities in the South Delta because the low-salinity zone is further downstream (or seaward) of the San Francisco Bay Delta. However, changes to the estuary's landscape (see *Habitat Alteration,* above) have caused the tidal flows to reach further into the Old and Middle Rivers (Andrews et al. 2017, p. 66), which, as discussed below, may further impact Bay-Delta longfin smelt (see *Water Project Exports,* below). Below we describe the types of freshwater diversions and exports and their impacts on Bay-Delta longfin smelt.

Water Project Exports: The State of California through the DWR and the U.S. Bureau of Reclamation (Reclamation) operate freshwater diversion facilities and infrastructure associated with the SWP and CVP, respectively, which export fresh water from the Delta. The operation of these facilities can exert a strong influence on regional hydrodynamics (Kimmerer and Nobriga 2008, fig. 7, p. 12; Hutton et al. 2019, fig. 7, p. 11). That hydrodynamic influence can result in the entrainment of fish, sometimes from considerable distances (Kimmerer 2008, p. 2, fig. 1, p. 3). Several methods have been implemented to limit and offset the entrainment impacts at the SWP and CVP facilities, including construction of forebays (areas used to collect fish before they enter the pumps), fish screens, gate systems (used to divert fish away from pumps), and salvage operations (active collection and transport of fish back into the estuary). In most years, Bay-Delta longfin smelt have been collected (''salvaged'') in the fish facilities that are in front of each pumping plant. The salvage of fish is an indicator that individuals are being entrained by pumping of water at these facilities and either being killed or removed from the estuary. The peak of salvage of age-1 and older Bay-Delta longfin smelt typically occurred in January (Grimaldo et al. 2009, fig. 5, p. 1262). These adult and age-1 fish likely represented individuals searching for spawning habitats and immature individuals commingling with the adults. The peak of salvage of age-0 fish typically occurred in April or May as larval fish reach sizes at which they could be retained on the fish screens of the CVP and SWP fish collection facilities. However, it is likely some larvae began to be entrained once they started hatching in December or January, but remained undetected until about March, with salvage efficiency increasing in April–May as the fish grow larger.

It is possible that past entrainment and loss of Bay-Delta longfin smelt may have reached levels of concern (CDFW 2020a, fig. 10, p. 47). However, since 2009, the entrainment of longfin smelt has not been substantial (Service 2024, fig. 3.4), perhaps partly due to monitoring and management of flows in the Old and Middle Rivers (OMR) between the Sacramento/San Joaquin River confluence and the export facilities. When net OMR flow is positive, San Joaquin River water is generally moving seaward through the San Francisco Bay Delta and away from the pumping facilities. The more net negative OMR is flowing, the more the water in the San Francisco Bay Delta is moving back upstream toward the pumping plants and the faster that water is moving south, thereby increasing entrainment potential. The additional negative flow causes Sacramento River water entering the northwest portion of the San Francisco Bay Delta to be diverted southward toward the pumping facilities rather than seaward, which allows saltier tidal flows to move further toward the San Francisco Bay Delta.

In order to address and minimize effects to federally listed fish species (delta smelt (*Hypomesus transpacificus*), chinook salmon (Sacramento River winter-run and Central Valley spring-run salmon (*Oncorhynchus tshawytscha*), Central Valley steelhead (*Oncorhynchus mykiss*), and green sturgeon (*Acipenser*

medirostris)), restrictions to pumping and other water operations management strategies have been implemented by the DWR and Reclamation to limit negative OMR flows and associated entrainment through the section 7 process of the Act (Service 2008, entire; National Oceanic and Atmospheric Administration, National Marine Fisheries Service [NMFS] 2009, entire; Service 2019, entire; NMFS 2019, entire). In addition, the DWR has implemented similar measures for State-listed species (including longfin smelt) (CDFW 2009b Incidental Take Permit (ITP), entire; CDFW 2020b, ITP, entire).

The results of two different analytical approaches using smelt larval survey (SLS) data suggest that entrainment of fish has not exceeded 3 percent since 2009 (Kimmerer and Gross 2022). Gross et al. (2022) coupled particle tracking modeling with the SLS data set and found an upper 95 percent credible interval of proportional entrainment was 2.9 percent in the critically dry winter of 2013 and nearly zero in the wet winter of 2017. Kimmerer and Gross (2022) analyzed all of the SLS data in the period 2009–2020. Similarly, this approach also found proportional entrainment was unlikely to have exceeded 3 percent (range = 0.5 to 2.9 percent) (Kimmerer and Gross 2022, table 1). We interpret these findings, as well as previously published information (CDFW 2020a, entire), to indicate that the OMR management strategies in place since 2009 have been an effective conservation strategy for limiting the impact of entrainment and its consequences for the Bay-Delta longfin smelt. As a result, the best information currently available indicates that management actions for operating water diversion facilities are assisting in limiting entrainment impacts for the Bay-Delta longfin smelt.

Contaminants

The San Francisco Bay estuary has been identified as an impaired water body due to it containing numerous and persistent contaminant compounds (California State Water Resources Control Board 2018, appendix A). The list of contaminant compounds identified within the estuary includes elemental contaminants or 'metals' (*e.g.,* mercury and selenium), toxic organic compounds (dioxins, furans, polychlorinated biphenyls), and pesticides (chlordane dieldrin, DDT). Additional emerging contaminants of concern include newer pesticides, flame retardants, nutrients, naturally occurring toxins, microplastics, and pharmaceuticals and personal care products (*i.e.,* plastic microbeads, insect repellant, sunscreen, cosmetics, etc.) (Klosterhaus et al. 2013, pp. 97–98, table 1; Sutton et al. 2017, entire). Ongoing analysis of water in the San Francisco Bay Delta suggests that on average 10 new synthetic organic pesticide chemicals are detected every year (California Department of Pesticide Regulation 2020, dataset). Water sampling in one study of the San Francisco Bay Delta indicated the presence of more than 50 chemical compounds from a single 1-liter (L) (34-ounce (oz)) water sample (Moschet et al. 2017, pp. 1557–1560).

The sources of contaminants include discharge from municipal wastewater treatment plants, agricultural outfalls, stormwater runoff, anti-fouling paints on boat and ship hulls, and direct human application of pest and aquatic plant control compounds (Service 2024, section 3.1.6). Legacy contaminants in the Bay-Delta (those from historical loading, such as organochlorine chemicals (*e.g.,* DDT) from past agricultural use and mercury from past mining activity) have been shown to persist in the environment and continue to impact ecosystems and can bioconcentrate through the food web, posing additional health risks (Connor et al. 2006, pp. 87–88; Marvin-DiPasquale and Cox 2007, p. 2). Regulation has reduced the use of some contaminants, only to be replaced by other more potent alternative water-soluble chemicals such as neonicotinoids, which have additional impacts on nontarget species such as aquatic invertebrates and fish (Buzby et al. 2020, pp. 15–21).

Field-based toxicity is difficult to determine, as impacted fish are not recovered in order to be examined (*i.e.,* fish either die from direct exposure and resulting disease, or are eaten). Risk of exposure and effect, as determined by comparison to other species (*e.g.,* delta smelt and the introduced inland silverside (*Menidia beryllina*)), potentially include direct effects on development, growth, and reproduction; impacts resulting from impairments to bioenergetic demands; and impaired locomotion, reducing feeding success, which can lead to increased susceptibility to predation, disease, and entrainment (Brander et al. 2012, p. 2854; Connon et al. 2009, p. 12; Hasenbein et al. 2014, p. 696; Jeffries et al. 2015a, p. 17407; Jeffries et al. 2015b, p. 55; Cole et al. 2016, p. 219; DeCourten and Brander 2017, p. 2).

Pelagic Organism Decline (POD)

Between the years 2002 through 2004, abundance indices for multiple fish species within the San Francisco Bay estuary declined abruptly in what is known as the pelagic organism decline, or POD. Specifically, the POD referred to a drop in survey catches of four fish species (Bay-Delta longfin smelt, delta smelt, striped bass (*Morone saxatilis*), and threadfin shad (*Dorosoma petenense*)) (Sommer et al. 2007, p. 273). The POD event is generally recognized as a population step-decline for numerous fish species in the estuary. The coincident declines of multiple species suggested a possible common cause, but a single mechanism for decline that applied to all four fish has not been identified (MacNally et al. 2010, p. 1426; Thomson et al. 2010, pp. 1442–1443). As a result, researchers have focused on multiple causes, from habitat changes, reductions in freshwater inflow, water diversions, food resource changes, competition, predation, and contaminants, as contributing to the POD (Sommer et al. 2007, pp. 271–276; MacNally et al. 2010, p. 1418; Fong et al. 2016, pp. 20–21). As outlined above, all of these factors have been identified as threats impacting the Bay-Delta longfin smelt to varying degrees. Although the POD event is not a threat in itself, but is instead most likely a result of multiple threats, the subsequently smaller DPS population is more susceptible to poor habitat conditions and has a reduced capability of rebounding from lower abundance years.

Bay-Delta Longfin Smelt Current Condition

Current Abundance

Several long-term survey efforts have been established for monitoring San Francisco Bay estuary fish populations including the Bay-Delta longfin smelt. These established survey efforts include the 20-mm survey, the SFBS, and the FMWT. The 20-mm survey has been conducted since 1995, and although it does not produce an abundance index for Bay-Delta longfin smelt, we adapted the methods for the delta smelt abundance index for the Bay-Delta longfin smelt. Our methods and information on how we adapted the study information are outlined in the current SSA report (Service 2024, appendix B). The SFBS has sampled low-salinity to fully marine waters of the estuary using standardized sampling methods since 1980. However, sampling was more sporadic in the 1990s and again in several recent years. The SFBS samples near bottom as well as midwater to surface-oriented fishes (Feyrer et al. 2015, Fig. 5, p. 3614) and provides separate abundance indices for ages 0, 1, and 2+ Bay-Delta longfin

smelt. The longest of these survey efforts is the FMWT, which was initiated in 1967 and has surveyed pelagic waters from the Delta into San Pablo Bay monthly from September through December each year. The FMWT captures mostly juvenile and adult fish 50–150 mm (2–6 in) in length and has been used to monitor the abundance of sampled fish species since the late 1970s (Stevens and Miller 1983, pp. 431–432). In the case of Bay-Delta longfin smelt, the FMWT samples adults and juveniles, most likely those returning from more marine environments to low-salinity and freshwater areas associated with spawning. Similar abundance estimates are reflected in the 20-mm survey, SFBS, and other modeling efforts (Service 2024, section 3.2.1). Collectively, these survey efforts encompass abundance estimates of all life stages of the Bay-Delta longfin smelt in the estuary.

Figure 2 identifies FMWT abundance information for Bay-Delta longfin smelt since its inception in 1967 with emphasis on the years 2000 to 2020. The FMWT time series is an index of fish numbers returning to spawn in the San Francisco Bay Delta, which is an indicator of abundance patterns as observed over this relatively longer time interval presented to give a simplified visual presentation of overall population trend during the last several decades. A more detailed analysis of overall trends and attendant risk is discussed, below.

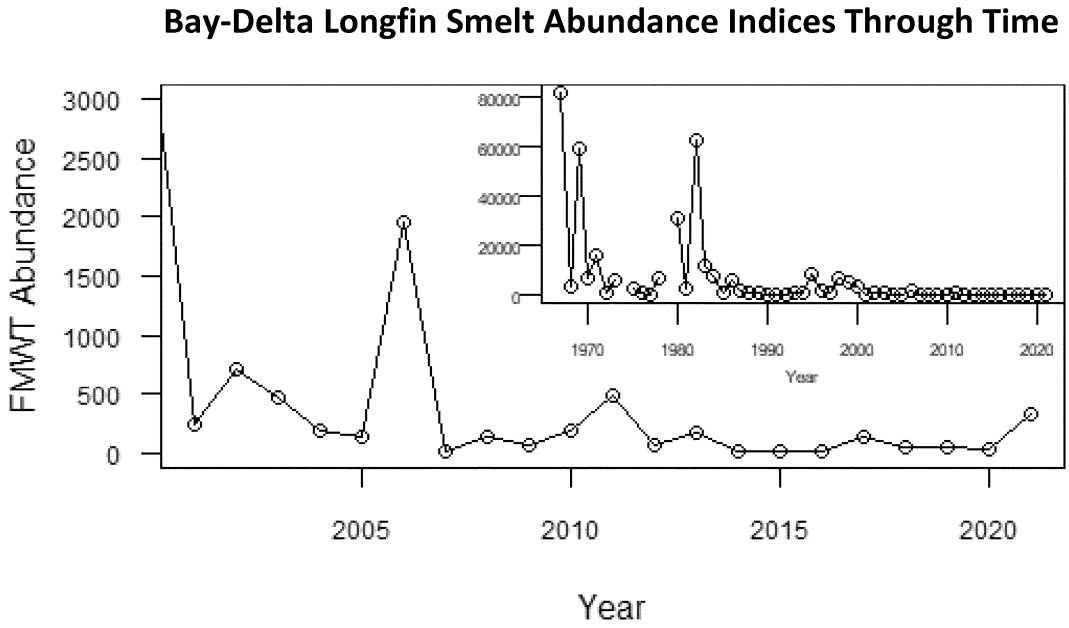


Figure 2. San Francisco Bay-Delta longfin smelt abundance indices for the period 2000–2020 from the fall midwater trawl survey. (Inset displays time series since 1967; source: adapted from California Department of Fish and Wildlife 2021.)

Population Trends and Risk of Quasi-Extinction

All the best available field surveys for documenting long-term abundance trends indicate Bay-Delta longfin smelt numbers have substantially declined over time, with current relative abundance reflecting small fractions of the species' historical relative abundance and representing a decline of three to four orders of magnitude over the course of available historical abundance records. Even considering the small periodic increases in numbers in occasional years in the most recent survey results (past 20 years), the general trend over time has been lower highs and lower lows in abundance for the DPS. This finding supports the conclusion that abundance of all life stages has declined substantially over the course of several decades and that the overall decline has continued in recent years (Service 2024, section 3.2).

A meta-analysis of annual population growth rates derived from the monitoring data showed that the DPS has a negative population growth rate (Service 2024, section 3.2.2). Figure 3 displays quasi-extinction risk projections (including confidence intervals) over time for the Bay-Delta longfin smelt from our risk assessment (Tobias et al. 2023, fig. 4, p. 7; Service 2024, appendix B).

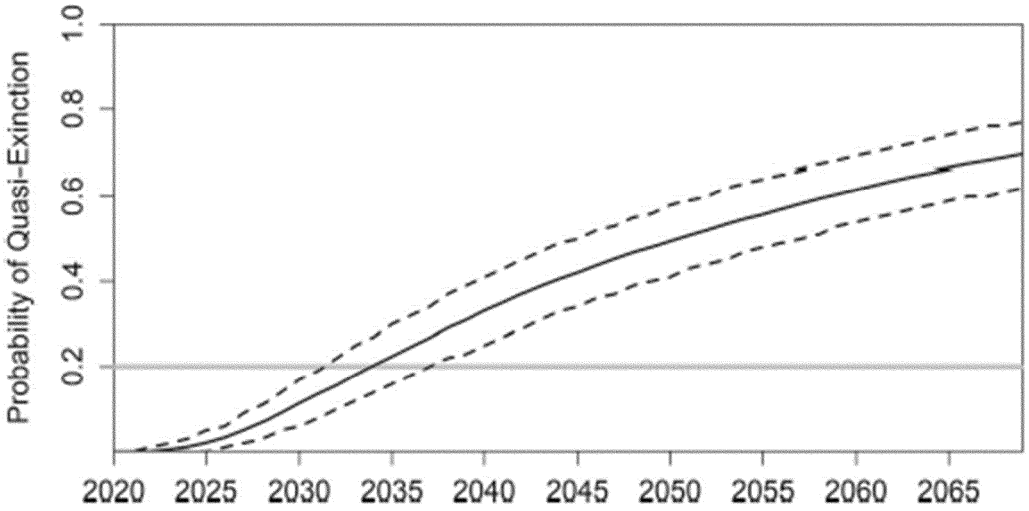


Figure 3. Quasi-extinction risk projections (including confidence intervals) over time for the San Francisco Bay-Delta longfin smelt. (Mean probability of quasi-extinction (solid line), with bootstrapped 95 percent confidence bands (dashed lines). The wide, horizontal line (gray line) highlights a 20 percent probability of quasi-extinction.)

Effects of Threats Impacting the Bay-Delta Longfin Smelt

Reduced and altered freshwater flows into the estuary greatly impact the physical and ecological processes important to Bay-Delta longfin smelt spawning and larval rearing habitat. Reductions in freshwater flow reduce the number of young that survive to later reproduce. Reduced freshwater flows also require the DPS to move farther inland to find appropriate low-salinity conditions for spawning and rearing. Although management actions to limit the impact of water diversions at export facilities have been implemented, this movement farther inland makes the DPS's larvae and young more vulnerable to entrainment as a result of water diversion from water export facilities.

The amount of freshwater input into the estuary is dependent on natural wet/dry precipitation patterns. These patterns have been influenced by the effects of current climate change conditions, which have resulted in more frequent, prolonged, and intense drought conditions (reduced flows) and increased water temperatures (poor habitat conditions). Freshwater flows into the estuary have also been greatly influenced by human-caused alteration of rivers and streams leading into the estuary as well as diversion and export of freshwater from the estuary. These human-caused impacts of water management have exacerbated the impacts of environmental variability of natural wet/dry precipitation patterns.

In addition to altered habitat conditions for the Bay-Delta longfin smelt, the available food resources for the DPS have also been severely impacted. A rapid change to the zooplankton community in the estuary beginning in the late 1980s along with the introduction of the nonnative species such as the overbite clam and others has greatly reduced the natural prey base for the DPS and replaced it with a smaller nonnative mysid. Because the fish in the DPS continue to exhibit very little variation in prey use despite the reduction in natural prey availability, they are considered more susceptible to food web changes than some other fishes. The decline in food resources is likely affecting juvenile and adult longfin smelt growth and fitness as well as increasing the effort needed to meet food resource demands.

After the review of the threats of predation, entrainment, and contaminants, we have determined that they are not primary driving factors currently influencing the Bay-Delta longfin smelt. However, these threats are likely still contributing cumulatively to the overall impacts acting on the DPS.

We note that, by using the SSA framework to guide our analysis of the scientific information documented in the SSA report, we have analyzed the cumulative effects of identified threats and conservation actions on the species. To assess the current and future condition of the species, we evaluate the effects of all the relevant factors that may be influencing the species, including threats and conservation efforts. Because the SSA framework considers not just the presence of the factors, but to what degree they collectively influence risk to the entire species, our assessment integrates the cumulative effects of the factors and replaces a standalone cumulative-effects analysis.

Resiliency, Redundancy, and Representation for the Bay-Delta Longfin Smelt

In the current SSA report for the Bay-Delta longfin smelt (Service 2024, chapter 3), we evaluated the Bay-Delta longfin smelt's resiliency, redundancy, and representation under our SSA framework (Service 2016, entire).

Resiliency describes the ability of a species to withstand stochastic disturbance. Because the Bay-Delta longfin smelt is a single, intermixed population, we did not identify multiple resiliency units but looked at the population as a whole. As discussed above, the Bay-Delta longfin smelt is subject to multiple interacting threats, including saltwater intrusion and reduced freshwater flows, that are altering and degrading habitat conditions. The resulting impact of these threats limits the extent, duration, and availability of appropriate habitat conditions needed for spawning, rearing, and ultimate recruitment of individuals into the population. These threats include anthropogenic actions (such as freshwater management, freshwater diversion, and physical alterations to the bathymetry of the estuary) or poor or altered

environmental conditions (such as increased frequency and magnitude of drought resulting from current climate change conditions). Disruptions to the estuary's food web associated with reductions in freshwater flow or introductions of nonnative species are also limiting resiliency for the DPS.

Redundancy is the ability of a species to withstand catastrophic events. The Bay-Delta longfin smelt is a single intermixed population and occurs in areas within the San Francisco Bay estuary as dictated by the extremely modified and altered habitat and resource conditions. The San Francisco Bay estuary is also subject to extreme environmental variability as a result of climate change conditions resulting in increased temperatures and frequency, magnitude, and duration of drought. As a result of these changes, the ability of the system and organisms within the estuary to withstand catastrophic events and rebound during periods of more favorable conditions is greatly reduced. Large-scale estuary-wide ecosystem population collapses of fish and native zooplankton have occurred in the estuary. Although no single cause for the collapses has been identified, both native and nonnative fish populations have not recovered. The result has been step-declines of the Bay-Delta longfin smelt population size since the mid-1980s, thereby reducing the redundancy of the DPS.

Representation describes the ability of a species to adapt to changing environmental conditions over time. This definition includes the ability of a species to adapt to both near-term and long-term changes in its physical and biological environments. The Bay-Delta longfin smelt population occurs in the San Francisco Bay estuary and is a single, genetically indistinguishable population. The Bay-Delta longfin smelt represents the southern extent of the species as a whole and most likely is a source for populations along the coast north of San Francisco Bay, but the number of individuals contributing to populations further north has substantially declined. Due to ocean currents and the species' poor swimming capability, populations north of the San Francisco Bay are unlikely to have the ability to move south and assist in reestablishing a population in the San Francisco Bay-Delta once they have been extirpated from the San Francisco Bay-Delta. The DPS's ability to adapt behaviorally to environmental changes (to have adaptive capacity) is also limited. This limitation is exemplified by the DPS's behavioral tendency of not adapting to food resource changes. As discussed, food resources for the DPS have changed significantly yet the DPS's behavior has not shifted to adapt to those changes.

In our evaluation of the current condition of the Bay-Delta longfin smelt, we evaluated several population viability analyses (PVAs) that quantitatively derive probabilities of extinction over time based on the DPS's historical and current abundance estimates (Service 2024; appendix B). The PVAs used information from the existing suite of surveys, including the FMWT, the 20-mm survey, and the SFBS (Service 2024, figure 3.11). The PVAs modeled extinction probability based on a continuation of existing threats currently facing the DPS under varying levels of population recruitment. Population growth rates were further synthesized by conducting a meta-analysis on the growth rates of the different surveys. The results of the count-based PVA meta-analysis identified that the probability of quasi-extinction for the Bay-Delta longfin smelt is estimated at 33 percent over 20 years and reaches 50 percent in 30 years (Service 2024, appendix B).

As a result of our review of the best scientific and commercial data available on the Bay-Delta longfin smelt, we have determined that the DPS's resiliency is low. Numerous decades of declining abundance indices for the Bay-Delta longfin smelt document the inability of the DPS to rebound during more favorable environmental conditions and respond to the threats it is facing in the contemporary San Francisco Bay estuary. The Bay-Delta longfin smelt also has extremely limited redundancy because it effectively represents a single, small population inhabiting the San Francisco Bay-Delta and nearshore ocean environment, and because it continues to be impacted by large-scale stochastic events and is subject to catastrophic events. We have determined that the representation of the Bay-Delta longfin smelt is limited as well, reflecting that same declining abundance trend and no discernible and quantifiable compensatory adaptation to current ecological conditions. Based on our evaluation of the current resiliency, redundancy, and representation for the Bay-Delta longfin smelt, we conclude the current ability of the DPS to maintain populations in the wild is low.

Future Conditions

As part of the SSA, we also developed future-condition scenarios to capture the range of uncertainties regarding future threats and the projected responses by the Bay-Delta longfin smelt. To assess the future condition of the Bay-Delta longfin smelt, we used published information related to the varying environmental conditions of the San Francisco Estuary, including future climate change information and projected increases in water demand, and how these changes may impact how well the estuary can support the Bay-Delta longfin smelt into the future. In our analyses, we considered two plausible future scenarios based on representative concentration pathways (RCP) 4.5 and 8.5 as the bookends for our analysis. The scenarios assessed climate change information (temperature increases, changes in precipitation patterns, sea-level rise) through 2100, as published information was available. The information identified that declines in Bay-Delta longfin smelt population abundance will continue into the future under both the RCP 4.5 and 8.5 scenarios. Because we determined that the current condition of the Bay-Delta longfin smelt was consistent with an endangered species (see Determination of the Bay-Delta Longfin Smelt's Status, below), we are not presenting the results of the future scenarios in this final rule. Please refer to the current SSA report (Service 2024, chapter 4) for the full analysis of future scenarios.

Conservation Efforts and Regulatory Mechanisms

Numerous efforts have been initiated regarding conservation and regulation of the San Francisco Bay estuary and its resources, including managing water flows into and export from the estuary, improving water quality, conducting habitat restoration, and implementing measures or regulations to protect native fish. This effort includes establishment of multiagency collaborations such as the Interagency Ecological Program (IEP), which focuses on coordinating and prioritizing science needs and research to meet responsibilities under State and Federal regulatory requirements (IEP 2014, entire).

The State of California listed the longfin smelt in the San Francisco Bay estuary and along the California Coast as a threatened species under the California Endangered Species Act in 2009 (CDFW 2009a, entire; California Natural Diversity Database 2022, entire) and has issued restrictions and requirements for the export of water for the State Water Project (see *Entrainment,* Water Project Exports, above). Several other fish species (delta smelt, several salmonid species) are listed under both the Act and the California Endangered Species Act, and the Service and NMFS have also issued biological opinions regarding the effects to these species and their habitats for

delivery and export of water from the estuary (see *Entrainment,* Water Project Exports, above). The State Water Board is responsible for issuing water quality standards and monitors contaminants within the estuary (see *Contaminants,* above). However, despite efforts such as those identified above, the current condition of the estuary and continued threats facing the estuary and Bay-Delta longfin smelt, such as reduced freshwater inflow, severe declines in population size, and disruptions to the DPS's food resources, have not been ameliorated.

Determination of the Bay-Delta Longfin Smelt's Status

Section 4 of the Act (16 U.S.C. 1533) and its implementing regulations (50 CFR part 424) set forth the procedures for determining whether a species meets the definition of an endangered species or a threatened species. The Act defines an ''endangered species'' as a species in danger of extinction throughout all or a significant portion of its range and a ''threatened species'' as a species likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. The Act requires that we determine whether a species meets the definition of endangered species or threatened species because of any of the following factors: (A) The present or threatened destruction, modification, or curtailment of its habitat or range; (B) Overutilization for commercial, recreational, scientific, or educational purposes; (C) Disease or predation; (D) The inadequacy of existing regulatory mechanisms; or (E) Other natural or manmade factors affecting its continued existence.

Status Throughout All of Its Range

The Bay-Delta longfin smelt's current abundance and density throughout the San Francisco Bay estuary have substantially declined. Currently, the DPS exists in very low abundance despite periods when appropriate habitat conditions, which typically would allow for population rebounds, are available. The best scientific and commercial information available and our analysis of that information revealed that several threats are causing or contributing to this decline and currently pose a meaningful risk to the viability of the DPS. These threats have put the Bay-Delta longfin smelt largely into a state of chronic population decline due to habitat loss (reduction in freshwater flows into the estuary), which is exacerbated by limited food resources and the impacts associated with climate change, thereby limiting its resiliency and ability to withstand catastrophic events (reduced redundancy). This decline in numbers of the Bay-Delta longfin smelt is also a reflection of the DPS's ability to adapt to the ecosystem changes.

As a result of the DPS's poor performance in adapting to the suite of stressors acting upon it, we consider the Bay-Delta longfin smelt's adaptive capacity and, therefore, its current representation to be low. The Bay-Delta longfin smelt's continued reduced population size makes the DPS vulnerable to varying habitat conditions from year to year due to both anthropogenic and environmental conditions that are being influenced by the effects of climate change. Historically, with a larger population size, the DPS was more resilient to such stochastic and catastrophic events due to its ability to rebound in abundance when habitat conditions and resources would allow. The habitat changes, limitations to food resources, and resulting small population size now limit the DPS's ability to maintain its current population.

After evaluating threats to the DPS and assessing the cumulative effect of the threats under the section 4(a)(1) factors, we find that the threats facing the San Francisco Bay-Delta DPS of the longfin smelt are current and ongoing and include habitat degradation and loss from reduced freshwater flow into the estuary (Factor A), increased intrusion of saltwater into spawning habitat areas (Factor A), alteration of food resources and availability (Factor E), nonnative species competition and food resource effects (Factor E), and the effects associated with climate change such as increased temperatures and frequency, magnitude, and duration of drought (Factor E). Because these threats are ongoing and currently impacting the DPS, and have already been shown to have caused a significant decline in the DPS's current resiliency, redundancy, and representation, the DPS meets the Act's definition of endangered status.

Thus, after assessing the best scientific and commercial information available, we determine that the San Francisco Bay-Delta DPS of the longfin smelt is in danger of extinction throughout all of its range.

Status Throughout a Significant Portion of Its Range

Under the Act and our implementing regulations, a species may warrant listing if it is in danger of extinction or likely to become so in the foreseeable future throughout all or a significant portion of its range. We have determined that the San Francisco Bay-Delta DPS of the longfin smelt is in danger of extinction throughout all of its range and accordingly did not undertake an analysis of any significant portions of its range. Because the San Francisco Bay-Delta DPS of the longfin smelt warrants listing as endangered throughout all of its range, our determination does not conflict with the decision in *Center for Biological Diversity* v. *Everson,* 435 F. Supp. 3d 69 (D.D.C. 2020), because that decision related to significant portion of the range analyses for species that warrant listing as threatened, not endangered, throughout all of their range.

Determination of Status

Our review of the best scientific and commercial information available indicates that the San Francisco Bay-Delta DPS of the longfin smelt meets the definition of an endangered species. Therefore, we are listing the San Francisco Bay-Delta DPS of the longfin smelt as an endangered species in accordance with sections 3(6) and 4(a)(1) of the Act and our February 7, 1996, policy regarding distinct population segments (61 FR 4722).

Available Conservation Measures

Conservation measures provided to species listed as endangered or threatened species under the Act include recognition as a listed species, planning and implementation of recovery actions, requirements for Federal protection, and prohibitions against certain practices. Recognition through listing results in public awareness, and conservation by Federal, State, Tribal, and local agencies, foreign governments, private organizations, and individuals. The Act encourages cooperation with the States and other countries and calls for recovery actions to be carried out for listed species. The protection required by Federal agencies, including the Service, and the prohibitions against certain activities are discussed, in part, below.

The primary purpose of the Act is the conservation of endangered and threatened species and the ecosystems upon which they depend. The ultimate goal of such conservation efforts is the recovery of these listed species, so that they no longer need the protective measures of the Act. Section 4(f) of the Act calls for the Service to develop and implement recovery plans for the conservation of endangered and threatened species. The goal of this process is to restore listed species to a point where they are secure, self-sustaining, and functioning components of their ecosystems.

The recovery planning process begins with development of a recovery outline made available to the public soon after a final listing determination. The recovery outline guides the immediate implementation of urgent recovery actions while a recovery plan is being developed. Recovery teams (composed of species experts, Federal and State agencies, nongovernmental organizations, and stakeholders) may be established to develop and implement recovery plans. The recovery planning process involves the identification of actions that are necessary to halt and reverse the species' decline by addressing the threats to its survival and recovery. The recovery plan identifies recovery criteria for review of when a species may be ready for reclassification from endangered to threatened (''downlisting'') or removal from protected status (''delisting''), and methods for monitoring recovery progress. Recovery plans also establish a framework for agencies to coordinate their recovery efforts and provide estimates of the cost of implementing recovery tasks. Revisions of the plan may be done to address continuing or new threats to the species, as new substantive information becomes available. The recovery outline, draft recovery plan, final recovery plan, and any revisions will be available on our website as they are completed (*https://www.fws.gov/program/endangered-species*), or from our San Francisco Bay-Delta Fish and Wildlife Office (see **FOR FURTHER INFORMATION CONTACT**).

Implementation of recovery actions generally requires the participation of a broad range of partners, including other Federal agencies, States, Tribes, nongovernmental organizations, businesses, and private landowners. Examples of recovery actions include habitat restoration (*e.g.,* restoration of native vegetation), research, captive propagation and reintroduction, and outreach and education. The recovery of many listed species cannot be accomplished solely on Federal lands because their range may occur primarily or solely on non-Federal lands. To achieve recovery of these species requires cooperative conservation efforts on private, State, and Tribal lands.

Once this species is listed, funding for recovery actions will be available from a variety of sources, including Federal budgets, State programs, and cost-share grants for non-Federal landowners, the academic community, and nongovernmental organizations. In addition, pursuant to section 6 of the Act, the State of California will be eligible for Federal funds to implement management actions that promote the protection or recovery of the San Francisco Bay-Delta DPS of the longfin smelt. Information on our grant programs that are available to aid species recovery can be found at: *https://www.fws.gov/service/financial-assistance.*

Please let us know if you are interested in participating in recovery efforts for the San Francisco Bay-Delta DPS of the longfin smelt. Additionally, we invite you to submit any new information on this species whenever it becomes available and any information you may have for recovery planning purposes (see **FOR FURTHER INFORMATION CONTACT**).

Section 7 of the Act is titled Interagency Cooperation and mandates all Federal action agencies to use their existing authorities to further the conservation purposes of the Act and to ensure that their actions are not likely to jeopardize the continued existence of listed species or adversely modify critical habitat. Regulations implementing section 7 are codified at 50 CFR part 402.

Section 7(a)(2) states that each Federal action agency shall, in consultation with the Secretary, ensure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of a listed species or result in the destruction or adverse modification of designated critical habitat. Each Federal agency shall review its action at the earliest possible time to determine whether it may affect listed species or critical habitat. If a determination is made that the action may affect listed species or critical habitat, formal consultation is required (50 CFR 402.14(a)), unless the Service concurs in writing that the action is not likely to adversely affect listed species or critical habitat. At the end of a formal consultation, the Service issues a biological opinion, containing its determination of whether the Federal action is likely to result in jeopardy or adverse modification.

Examples of discretionary actions for the San Francisco Bay-Delta DPS of the longfin smelt that may be subject to consultation procedures under section 7 are land management or other landscape-altering activities on Federal lands administered by the Service, National Park Service, Department of Defense, Reclamation, U.S. Army Corps of Engineers, or U.S. Department of Agriculture, as well as actions on State, Tribal, local, or private lands that require a Federal permit (such as a permit from the U.S. Army Corps of Engineers under section 404 of the Clean Water Act (33 U.S.C. 1251 *et seq.*) or a permit from the Service under section 10 of the Act) or that involve some other Federal action (such as funding from the Federal Highway Administration, Federal Aviation Administration, or the Federal Emergency Management Agency). Federal actions not affecting listed species or critical habitat—and actions on State, Tribal, local, or private lands that are not federally funded, authorized, or carried out by a Federal agency—do not require section 7 consultation. Federal agencies should coordinate with the local Service Field Office (see **FOR FURTHER INFORMATION CONTACT**) with any specific questions on section 7 consultation and conference requirements.

The Act and its implementing regulations set forth a series of general prohibitions and exceptions that apply to endangered wildlife. The prohibitions of section 9(a)(1) of the Act, and the Service's implementing regulations codified at 50 CFR 17.21, make it illegal for any person subject to the jurisdiction of the United States to commit, to attempt to commit, to solicit another to commit or to cause to be committed any of the following acts with regard to any endangered wildlife: (1) import into, or export from, the United States; (2) take (which includes harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect) within the United States, within the territorial sea of the United States, or on the high seas; (3) possess, sell, deliver, carry, transport, or ship, by any means whatsoever, any such wildlife that has been taken illegally; (4) deliver, receive, carry, transport, or ship in interstate or foreign commerce, by any means whatsoever and in the course of commercial activity; or (5) sell or offer for sale in interstate or foreign commerce. Certain exceptions to these prohibitions apply to employees or agents of the Service, the National Marine Fisheries Service, other Federal land management agencies, and State conservation agencies.

We may issue permits to carry out otherwise prohibited activities involving endangered wildlife under certain circumstances. Regulations governing permits for endangered wildlife are codified at 50 CFR 17.22, and general Service permitting regulations are codified at 50 CFR part 13. With regard to endangered wildlife, a permit may be issued: for scientific purposes, for enhancing the propagation or survival of the species. The statute also contains certain exemptions from the prohibitions, which are found in sections 9 and 10 of the Act.

It is the policy of the Services, as published in the **Federal Register** on July 1, 1994 (59 FR 34272), to identify,

to the extent known at the time a species is listed, specific activities that will not be considered likely to result in violation of section 9 of the Act. To the extent possible, activities that will be considered likely to result in violation will also be identified in as specific a manner as possible. The intent of this policy is to increase public awareness of the effect of a listing on proposed and ongoing activities within the range of the species.

As discussed above, certain activities that are prohibited under section 9 may be permitted under section 10 of the Act. In addition, to the extent currently known, the following activities will not be considered likely to result in violation of section 9 of the Act: (1) take of the longfin smelt outside the range of the DPS as identified in figure 1 above; (2) take as a result of recreational fishing as permitted by the State of California; or (3) recreational boating on open-water areas of the San Francisco Bay-Delta Estuary.

This list is intended to be illustrative and not exhaustive; additional activities that will not be considered likely to result in violation of section 9 of the Act may be identified during coordination with the local field office, and in some instances (*e.g.,* with new information), the Service may conclude that one or more activities identified here will be considered likely to result in violation of section 9.

To the extent currently known, the following is a list of examples of activities that will be considered likely to result in violation of section 9 of the Act in addition to what is already clear from the descriptions of the prohibitions found at 50 CFR 17.21: (1) handling or collecting individuals of the DPS; (2) destruction/alteration of the Bay-Delta longfin smelt's habitat by discharge of fill material, dredging, draining, ditching, or stream channelization or diversion; (3) unauthorized diversion or alteration of surface flow into the San Francisco Bay-Delta estuary by removal of freshwater from rivers, streams, wetlands, and other aquatic features; (4) introduction of contaminants that may degrade water quality of the San Francisco Bay-Delta estuary; or (5) introduction of nonnative species that compete with or prey upon the Bay-Delta longfin smelt or alter food resources for the DPS.

This list is intended to be illustrative and not exhaustive; additional activities that will be considered likely to result in violation of section 9 of the Act may be identified during coordination with the local field office, and in some instances (*e.g.,* with new or site-specific information), the Service may conclude that one or more activities identified here will not be considered likely to result in violation of section 9.

Questions regarding whether specific activities would constitute violation of section 9 of the Act should be directed to the San Francisco Bay-Delta Fish and Wildlife Office (see **FOR FURTHER INFORMATION CONTACT**).

II. Critical Habitat

Section 4(a)(3) of the Act and implementing regulations (50 CFR 424.12) require that we designate critical habitat at the time a species is determined to be an endangered or threatened species, to the maximum extent prudent and determinable. In the October 7, 2022, proposed listing rule (87 FR 60957 at 60970), we determined that designation of critical habitat was prudent but not determinable because specific information needed to analyze the impacts of designation was lacking. Since the publication of the proposed listing rule, we have obtained the necessary information and are in the process of developing a proposed critical habitat designation for the Bay-Delta longfin smelt. We plan to publish a proposed critical habitat rule in the near future and complete a final designation as required by sections 4(a)(3) and 4(b)(6)(C)(ii) of the Act.

Required Determinations

National Environmental Policy Act (42 U.S.C. 4321 et seq.)

It is our position that, outside the jurisdiction of the U.S. Court of Appeals for the Tenth Circuit, we do not need to prepare environmental analyses pursuant to the National Environmental Policy Act (42 U.S.C. 4321 *et seq.*) in connection with regulations adopted pursuant to section 4(a) of the Act. We published a notice outlining our reasons for this determination in the **Federal Register** on October 25, 1983 (48 FR 49244). This position was upheld by the U.S. Court of Appeals for the Ninth Circuit (*Douglas County* v. *Babbitt,* 48 F.3d 1495 (9th Cir. 1995), cert. denied 516 U.S. 1042 (1996)).

Government-to-Government Relationship With Tribes

In accordance with the President's memorandum of April 29, 1994 (Government-to-Government Relations with Native American Tribal Governments; 59 FR 22951, May 4, 1994), Executive Order 13175 (Consultation and Coordination with Indian Tribal Governments), the President's memorandum of November 30, 2022 (Uniform Standards for Tribal Consultation; 87 FR 74479, December 5, 2022), and the Department of the Interior's manual at 512 DM 2, we readily acknowledge our responsibility to communicate meaningfully with federally recognized Tribes and Alaska Native Corporations on a government-to-government basis. In accordance with Secretaries' Order 3206 of June 5, 1997 (American Indian Tribal Rights, Federal-Tribal Trust Responsibilities, and the Endangered Species Act), we readily acknowledge our responsibilities to work directly with Tribes in developing programs for healthy ecosystems, to acknowledge that Tribal lands are not subject to the same controls as Federal public lands, to remain sensitive to Indian culture, and to make information available to Tribes. No Tribal lands were identified within the range of the Bay-Delta longfin smelt, and we did not receive any information during our development of the SSA report for the DPS or the two open public comment periods. We will continue to reach out and coordinate with Tribal entities during the development of our recovery planning and critical habitat designation processes for the Bay-Delta longfin smelt.

References Cited

A complete list of references cited in this rulemaking is available on the internet at *https://www.regulations.gov* and upon request from the San Francisco Bay-Delta Fish and Wildlife Office (see **FOR FURTHER INFORMATION CONTACT**).

Authors

The primary authors of this rule are the staff members of the Fish and Wildlife Service's Species Assessment Team and the San Francisco Bay-Delta Fish and Wildlife Office.

List of Subjects in 50 CFR Part 17

Endangered and threatened species, Exports, Imports, Plants, Reporting and recordkeeping requirements, Transportation, Wildlife.

Signing Authority

Martha Williams, Director of the U.S. Fish and Wildlife Service, approved this action on June 14, 2024, for publication. On July 22, 2024, Martha Williams authorized the undersigned to sign the document electronically and submit it to the Office of the Federal Register for publication as an official document of the U.S. Fish and Wildlife Service.

Regulation Promulgation

Accordingly, we amend part 17, subchapter B of chapter I, title 50 of the Code of Federal Regulations, as set forth below:

PART 17—ENDANGERED AND THREATENED WILDLIFE AND PLANTS

■ 1. The authority citation for part 17 continues to read as follows:

Authority: 16 U.S.C. 1361–1407; 1531–1544; and 4201–4245, unless otherwise noted.

■ 2. Amend § 17.11 in paragraph (h), in the List of Endangered and Threatened Wildlife, by adding an entry for ''Smelt, longfin [San Francisco Bay-Delta DPS]'' in alphabetical order under FISHES to read as follows:

§ 17.11 Endangered and threatened wildlife.

* * * * *

(h) * * *

Common name	Scientific name	Where listed	Status	Listing citations and applicable rules
*	*	*	*	*
		FISHES		
*	*	*	*	*
Smelt, longfin [San Francisco Bay-Delta DPS].	*Spirinchus thaleichthys* ..	U.S.A. (CA)	E	89 FR [INSERT FIRST PAGE OF **FEDERAL REGISTER** PUBLICATION], 7/30/2024.
*	*	*	*	*

Madonna Baucum,

Regulations and Policy Chief, Division of Policy, Economics, Risk Management, and Analytics of the Joint Administrative Operations, U.S. Fish and Wildlife Service.

[FR Doc. 2024–16380 Filed 7–29–24; 8:45 am]

BILLING CODE 4333–15–P

CALIFORNIA DEPARTMENT OF FISH AND WILDLIFE

California Endangered Species Act

Petition Evaluation for White Sturgeon (*Acipenser Transmontanus*)

Report to the Fish and Game Commission

March 2024



Suggested citation:

California Department of Fish and Wildlife (CDFW). 2024. Report to the Fish and Game Commission. Evaluation of the petition from San Francisco Baykeeper, The Bay Institute, Restore the Delta, and California Sportfishing Protection Alliance to list White Sturgeon (*Acipenser transmontanus)* as threatened under the California Endangered Species Act. California Department of Fish and Wildlife, P.O. Box 944209, Sacramento CA 94244-2090. 19 pp.

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List of Abbreviations, Acronyms, and Terms

CDFW – California Department of Fish and Wildlife

CESA – California Endangered Species Act

cfs - cubic feet per second

Commission – California Fish and Game Commission

Department – California Department of Fish and Wildlife

e.g. – "for example" (*Latin: exempli gratia*)

ESA – Federal Endangered Species Act

et al. – "and others"

FL - fork length, the length of a fish measured from the tip of the snout to the fork in the tail

Id. – "the same"

i.e. – "that is" (*Latin: id est*)

mtDNA - mitochondrial deoxyribonucleic acid

Executive Summary

San Francisco Baykeeper, The Bay Institute, Restore the Delta, and California Sportfishing Protection Alliance submitted a petition (Petition) to the California Fish and Game Commission (Commission) to list White Sturgeon (*Acipenser transmontanus)* as threatened pursuant to the California Endangered Species Act (CESA). White Sturgeon are an anadromous fish native to California, where they primarily reside in San Francisco Bay and the Delta and spawn in the Sacramento and San Joaquin rivers and associated tributaries. They are slow-growing, late maturing, and long-lived, and are able to spawn multiple times during their lives. Historically, abundance estimates for the California population ranged between 175,000-200,000 legal-sized fish; the most recent five-year average abundance estimate indicates there are now approximately 33,000 legal-sized fish in the population.

On December 7, 2023, the Commission referred the Petition to the California Department of Fish and Wildlife (Department) in accordance with Fish and Game Code section 2073 (Cal. Reg. Notice Register 2024, No. 1-Z, p. 26). Pursuant to Fish and Game Code section 2073.5 and California Code of Regulations, title 14, section 670.1, the Department prepared this evaluation report (Petition Evaluation) within 120 days of receiving the Petition. The purpose of the Petition Evaluation is to evaluate the sufficiency of the scientific information contained in the Petition in relation to other relevant information possessed or received by the Department during the evaluation period, and to recommend to the Commission whether the Petition should be accepted and considered.

Summary of the information presented in the petition:

- **Life History, Range, Distribution (including map):** The Petition provides sufficient information regarding the life history, current range, and distribution of White Sturgeon, including a current distribution map.
- **Habitat necessary for survival:** The Petition provides sufficient information regarding the diverse spawning, rearing, holding, and foraging habitats used by White Sturgeon across fresh, estuarine, and ocean waters.
- **Abundance and population trends:** The Petition provides sufficient information regarding current White Sturgeon abundance estimates and population trends based on CDFW monitoring.
- **Factors affecting the ability to survive and reproduce:** The Petition provides sufficient information regarding factors threatening White Sturgeon survival and reproduction. It identifies Central Valley water

management infrastructure and operations, overharvest in the recreational fishery, and harmful algal blooms leading to both direct mortality and impaired water conditions as the three primary concerns. Climate change, vessel strikes, dredging, poaching, and water contaminants are also referenced.

- **Degree and immediacy of threat:** The Petition provides sufficient information to suggest that threats to long-term survival of White Sturgeon will continue or potentially worsen in the future. It identifies the most significant threats as harmful algal blooms, recreational fishing, and water diversions.
- **Impact of existing management efforts:** The Petition contains sufficient information explain concerns that existing and foreseeable regulatory mechanisms and management efforts will not adequately protect White Sturgeon from impacts that threaten their long-term survival. It states concerns that proposed voluntary agreements, updates to the Bay-Delta Water Quality Control Plan, and plans for Sites Reservoir and the Delta Conveyance Project would impair water timing and quality and species recruitment, and states concerns that the levels of harvest targeted in fishing regulations are not consistent with the goal of maintaining or restoring a stable population.
- **Suggestions for future management:** The Petition provides sufficient recommendations of management actions that would reduce key stressors, improving water flows and timing, water quality, reduce export and harvest mortality, and improve monitoring efforts.
- **Availability and sources of information.** The Petition references 68 peer-reviewed or public documents, including numerous documents authored by the Department. The references are current and appropriate.

After reviewing the Petition and other relevant information, the Department has determined that the Petition meets the requirements in Fish and Game Code section 2072.3 and California Code of Regulations, title 14, section 670.1, subdivision (d)(1). In completing its Petition Evaluation, the Department has determined that there is sufficient scientific information to indicate that the petitioned action to list White Sturgeon as threatened under CESA may be warranted. Therefore, the Department recommends that the Commission accept the petition for further consideration pursuant to CESA.

Introduction

Candidacy Evaluation

The Commission has the authority to list certain species or subspecies as endangered or threatened under CESA (Fish & G. Code, §§ 2062, 2067, & 2070). The listing process is the same for species and subspecies (Fish & G. Code, §§ 2070-2079.1). CESA defines the "species" eligible for listing to include "species or subspecies" (Fish & G. Code, §§ 2062, 2067, & 2068), and courts have held that the term "species or subspecies" includes "evolutionarily significant units" (*Central Coast Forest Assn. v. Fish & Game Com.* (2018) 18 Cal.App.5th 1191, 1236, citing *Cal. Forestry Assn., supra*, 156 Cal.App.4th at pp. 1542 and 1549). The range of a species for the Department's petition evaluation and recommendation is the species' California range only (*Cal. Forestry Assn. v. Cal. Fish and Game Com.* (2007) 156 Cal.App.4th 1535, 1551).

CESA sets forth a two-step process for listing a species as threatened or endangered. First, the Commission determines whether to designate a species as a candidate for listing by evaluating whether the petition provides "sufficient information to indicate that the petitioned action may be warranted" (Fish & G. Code, § 2074.2, subd. (e)(2)). If the petition is accepted for consideration, the second step requires the Department to produce, within 12 months (18 months with extension) of the Commission's acceptance of the petition, a peer-reviewed report based upon the best scientific information available that advises the Commission on whether the petitioned action is warranted (Fish & G. Code, § 2074.6). Finally, the Commission, based on that report and other information in the administrative record, determines whether the petitioned action to list the species as threatened is warranted (Fish & G. Code, § 2075.5).

A petition to list a species under CESA must include "information regarding the population trend, range, distribution, abundance, and life history of a species, the factors affecting the ability of the population to survive and reproduce, the degree and immediacy of the threat, the impact of existing management efforts, suggestions for future management, and the availability and sources of information. The petition shall also include information regarding the kind of habitat necessary for species survival, a detailed distribution map, and any other factors that the petitioner deems relevant" (Fish & G. Code, § 2072.3; see also Cal. Code Regs., tit. 14, § 670.1, subd. (d)(1)).

Within 10 days of receipt of a petition, the Commission must refer the petition to the Department for evaluation (Fish & G. Code, § 2073). The Commission must

also publish notice of receipt of the petition in the California Regulatory Notice Register (Fish & G. Code, § 2073.3). Within 90 days of receipt of the petition (or 120 days if the Commission grants an extension), the Department must evaluate the petition on its face and in relation to other relevant information the Department possesses and submit to the Commission a written evaluation report with one of the following recommendations (Fish & G. Code, § 2073.5, subds. (a)-(b)):

- Based upon the information contained in the petition, there is not sufficient information to indicate that the petitioned action may be warranted, and the petition should be rejected; or
- Based upon the information contained in the petition, there is sufficient information to indicate that the petitioned action may be warranted, and the petition should be accepted and considered.

The Department's candidacy recommendation to the Commission is based on an evaluation of whether the petition provides sufficient scientific information relevant to the petition components set forth in Fish and Game Code section 2072.3 and the California Code of Regulations, title 14, section 670.1, subdivision (d)(1) to indicate that the petitioned action to list White Sturgeon as threatened may be warranted.

In *Center for Biological Diversity v. California Fish and Game Commission* (2008) 166 Cal.App.4th 597, the California Court of Appeals addressed the parameters of the Commission's determination of whether a petitioned action should be accepted for consideration pursuant to Fish and Game Code section 2074.2, subdivision (e), resulting in the species being listed as a candidate species. The court began its discussion by describing the standard for accepting a petition for consideration previously set forth in *Natural Resources Defense Council v. California Fish and Game Commission* (1994) 28 Cal.App.4th 1104:

> As we explained in *Natural Resources Defense Council*, "the term 'sufficient information' in section 2074.2 means that amount of information, when considered with the Department's written report and the comments received, that would lead a reasonable person to conclude the petitioned action may be warranted." The phrase "may be warranted" "is appropriately characterized as a 'substantial possibility that listing could occur.'" "Substantial possibility," in turn, means something more than the one-sided "reasonable possibility" test for an environmental impact report but does not require that listing be more likely than not.

(*Center for Biological Diversity, supra*, 166 Cal.App.4th at pp. 609-10 [internal citations omitted]). The court acknowledged that "the Commission is the finder of fact in the first instance in evaluating the information in the record" (*Id*. at p. 611). However, the court clarified:

> [T]he standard, at this threshold in the listing process, requires only that a substantial possibility of listing could be found by an objective, reasonable person. The Commission is not free to choose between conflicting inferences on subordinate issues and thereafter rely upon those choices in assessing how a reasonable person would view the listing decision. Its decision turns not on rationally based doubt about listing, but on the absence of any substantial possibility that the species could be listed after the requisite review of the status of the species by the Department under [Fish and Game Code] section 2074.6.

(*Ibid.*)

CESA Petition History

On November 29, 2023, the Commission received the Petition from San Francisco Baykeeper, The Bay Institute, Restore the Delta, and California Sportfishing Protection Alliance to list White Sturgeon (*Acipenser transmontanus*) as threatened under CESA. On December 6, 2023, the Commission referred the Petition to the Department for evaluation. At its meeting on December 14, 2023, the Commission officially received the Petition. At its meeting on February 15, 2024, the Commission granted the Department's request for a 30-day extension of the period to review the Petition and prepare this Petition Evaluation.

Federal ESA Petition History

On November 29, 2023, pursuant to Section 4(b) of the Endangered Species Act, 16. U.S.C. § 1533(b); Section 553(e) of the Administrative Procedure Act, 5. U.S.C. § 553(e); and 50 C.F.R. § 424.14(a), San Francisco Baykeeper, The Bay Institute, Restore the Delta, and California Sportfishing Protection Alliance provided notice in accordance with 50 C.F.R. § 424.14(b) and (c)(9) that they intended to petition the Secretary of Commerce, through the National Marine Fisheries Service, to protect the San Francisco Estuary White Sturgeon (*Acipenser transmontanus*) Distinct Population Segment as a threatened species.

Species Taxonomy

Analysis of multiple mitochondrial gene sequences indicates that the closest relatives of White Sturgeon are derived from Asia, including *A. schrenckii*, *A. sinensis*, and *A. dabryanus* (Krieger et al. 2008; Hildebrand et al. 2016). Analysis of multiple mtDNA sequences suggested that White Sturgeon last shared a common ancestor with *A. schrenckii* (Amur Sturgeon) approximately 46 million years ago (Hildebrand et al. 2016 citing Peng et al. 2007). There are multiple populations of White Sturgeon found on the west coast of North America with one genetically identifiable group found in California (Schreier et al. 2013).

Species Overview

White Sturgeon (*A. transmontanus*) are one of two sturgeon species native to California (along with Green Sturgeon, *A. medirostris,* which is listed as "threatened" on the federal ESA list, but not listed under CESA [Cal. Code Regs., tit 14, § 670.5]). There are listed populations (ESA or Canadian SARA) of White Sturgeon in the upper Columbia River (above Grand Coulee Dam), Kootenai River, lower, middle and, upper Fraser River, and Nechako River; unlisted populations are found in the Sacramento-San Joaquin rivers, Columbia River (below Grand Coulee Dam), and Snake River (Hildebrand et al. 2016). White Sturgeon can live in excess of 100 years, and historically grew to sizes of approximately 20 feet and 1300 pounds (Moyle 2002). The species is anadromous and is capable of coastal movements, although this appears to be uncommon and most adults in the Central Valley watershed population reside in San Francisco Bay and the Delta (Miller et al. 2020). Recent studies indicate that there are several different residence strategies in the population, with some fish remaining in the freshwater Delta for most of their lives and a larger proportion residing in the saline areas of the bay (Sellheim et al. 2022).

Adult White Sturgeon make seasonal migrations starting in November to spawn in the major rivers of the Central Valley (Miller et al. 2020). Historically, White Sturgeon likely spawned as far upstream in the Sacramento River watershed as the Pitt River and well into the San Joaquin River (Beamesderfer et al. 2004). Dams and anthropogenic water alterations have reduced access to spawning habitat (Hildebrand et al. 2016). Currently, the majority of spawning is thought to occur on the Sacramento River between river kilometers 127–248) (Schaffter 1997; CDFW 2021, 2022, 2023a), with a lesser amount of spawning on the lower San Joaquin River between river kilometers 115–138 (Jackson et al. 2016). Some additional spawning may occur on major tributaries such as the Feather, Bear,

Yuba, and Tuolumne rivers. White Sturgeon are observed in these rivers, but spawning has not been documented with captured eggs or larvae.

A small proportion of White Sturgeon start to mature at approximately 10 years with males maturing earlier than females; however, the majority of the population can take 14–19 years or more to mature to first reproduction (Chapman et al. 1996; Hildebrand et al. 2016; CDFW 2023b). Once mature, males are believed to spawn every 1–2 years and females every 2–4 years (Chapman et al. 1996). The species is a broadcast spawner, releasing large adhesive eggs into the water column, usually over coarse gravel and small cobble substrate (Moyle 2002). Eggs hatch in 4–12 days (Wang et al. 1985) and larvae rear in the river before moving down to the freshwater Delta where they are detected in the CDFW 20-mm tow-net survey[1]. Successful recruitment to the juvenile population is infrequent, occurring approximately every 6–7 years. Large year classes and successful recruitment are highly correlated with above normal water years as measured by high mean daily Delta outflow (Fish 2010; Gingras et al. 2013).

CDFW began monitoring the abundance of legal-sized sturgeon in the fishery in 1954 using mark-recapture tagging. Estimates were made of the abundance of fish that were "legal sized" based on the regulations at the time. Sampling effort was intermittent and then occurred annually after Green Sturgeon received federal ESA protection in 2006 (CDFW 2023b). Historically, the estimates of 40–60-inch FL White Sturgeon ranged around 125,000–150,000 fish. The most recent CDFW five-year average abundance estimate suggests there are approximately 33,000 40–60-inch FL fish in the population (CDFW 2023b).

There has been a recreational fishery for White Sturgeon in California since 1954. As of November 16, 2023, the fishery is operating under emergency regulations that permit anglers to take one sturgeon per year between 42–48-inch fork length (FL) and limit the total number of sturgeon taken per vessel per day to two. The fishery is open year-round in the San Francisco Bay and Delta. The Sacramento River and tributaries (upstream of the Highway 50 bridge) and the San Joaquin River and tributaries (upstream of the I-5 bridge) are closed to sturgeon fishing from January 1 through May 31 and open the remaining

[1] https://wildlife.ca.gov/Conservation/Delta/20mm-Survey

months. Revised long-term regulations have been under development for implementation in 2025.

Sufficiency of Scientific Information to Indicate the Petitioned Action May Be Warranted

Pursuant to Fish and Game Code section 2072.3 and California Code of Regulations, title 14, section 670.1, subdivision (d)(1), the Department verified that the petition contained information on each of the following petition components:

- Life History;
- Range;
- Distribution;
- Detailed Distribution Map
- Kind of habitat necessary for survival;
- Abundance;
- Population Trend
- Factors affecting the ability to survive and reproduce;
- Degree and immediacy of threat;
- Impact of existing management efforts;
- Suggestions for future management; and
- Availability and sources of information.

The Commission did not receive new information from the public during the Petition Evaluation period (Fish & G. Code, § 2073.4). Pursuant to Fish and Game Code section 2073.5, the Department evaluated based upon the information contained in the petition, whether there is, or is not, sufficient information to indicate that the petitioned action may be warranted. Below is a summary of relevant information from the petition for each of the petition components.

Natural History

Scientific Information in the Petition

The Petition discusses life history of White Sturgeon under "Section 2. Natural History" on pages 6-11 of the Petition, referencing current literature. The petitioners state that recruitment from the egg/larvae stages into the juvenile population is infrequent and correlated with high river flows and Delta Outflow

and discuss the mechanisms underlying this process, referencing Fish 2010, CDFW 2015, Israel et al. 2009 and other sources.

Range and Distribution[2]

Scientific Information in the Petition



Figure 1. Current distribution of White Sturgeon from CDFW (2023b, page 9). This map was included as Figure 2 in the Petition.

[2] Summaries of the information provided about range, distribution, and distribution map have all been included in the Range and Distribution Section

The Petition discusses range and distribution for White Sturgeon (including a map, Figure 1) in "Section 3. Range and Distribution" on pages 11–14 of the Petition. Discussion of the species' range (Section 3.1) addresses river, estuarine, and ocean habitation and includes populations from the Columbia and Fraser river drainages as well as the population in the Central Valley (Hildebrand et al. 2016; CDFW 2015). Distribution is addressed in Section 3.2, focusing on population spatial structure and discussing constraints related to upstream dams and possible threats White Sturgeon may experience in the estuaries.

Kind of Habitat Necessary for Survival

Scientific Information in the Petition

The Petition discusses the kind of habitat necessary for survival for White Sturgeon in California in "Section 5. Habitat Necessary for Species Survival" on pages 22–24 of the Petition. The petitioners address spawning habitat in major Central Valley rivers, dispersal and rearing habitat in the rivers and estuary, foraging and holding habitat in the estuary and Delta for subadults and adults, and infrequent long-distance marine migrations of adults. Critical habitat needs are discussed on page 24.

Abundance

Scientific Information in the Petition

The Petition discusses the abundance of White Sturgeon in California in "Section 4.1. Abundance" on pages 14–17 of the Petition. The references are up to date and rely on Department-authored documents including CDFW 2023b, Fish 2010, and documents presented at Fish and Game Commission meetings. The petitioners also discuss declining trends in both juvenile and adult abundance using data from Department monitoring.

Population Trend

Scientific Information in the Petition

The Petition discusses White Sturgeon population trends in California in "Section 4.2. Population Trends" on pages 17–22 of the Petition. "Subsection 4.2.1. River Flows and Delta Outflow" of the Petition addresses declining trends in juvenile recruitment correlated with low river flows and Delta Outflow (e.g. Fish 2010; CDFW 2015; SWRCB 2017). The petitioners state that monthly average Delta

outflows >37,000 cfs during March–July are sufficiently protective of White Sturgeon (SWRCB 2017), yet flows of this magnitude have only occurred in 4 out of the past 23 years. Subsequent sections discuss mortality due to entrainment of juveniles of water operations (4.2.2), fishing harvest (4.2.3), and recent harmful algal blooms (4.2.3).

Factors Affecting the Ability to Survive and Reproduce

Scientific Information in the Petition

The Petition discusses factors affecting ability to survive and reproduce in "Section 6. Factors Affecting Ability to Survive and Reproduce" on pages 24–32 of the Petition. The petitioners identify the three primary threats as:

1) Central Valley water management infrastructure and operations including dams, altered hydrograph due to water operations, and direct mortality from export operations. Potential impacts of planned projects (e.g. Sites Reservoir, Delta Conveyance Project, Bay-Delta Water Quality Control Plan and Voluntary Agreements) are also addressed.
2) Overharvest in the recreational fishery.
3) Harmful algal blooms leading to fish kills and impaired water quality.

The petitioners also discuss other factors that may influence the survival of the species including poaching, pollution, dredging, vessel strikes, and climate change.

Degree and Immediacy of Threat

Scientific Information in the Petition

The Petition discusses the degree and immediacy of the threat in "Section 7. Degree and Immediacy of Threat" on page 33 of the Petition with an emphasis on the three primary threats identified in Section 6 of the Petition. The petitioners discuss the role that current reservoir and river water management may have in persistent declines in the population and note that flow conditions may be further impacted by major projects currently in development. They also discuss the impacts of high levels of harvest from recreational fishing and the emerging threat of harmful algal blooms.

Impact of Existing Management Efforts

Scientific Information in the Petition

The Petition discusses the impact of existing management efforts in "Section 8. Impact of Existing Management Efforts" on pages 33–35 of the Petition with an emphasis on the three primary threats identified in Section 6 of the Petition. They discuss river and estuarine flow requirements as minimum standards that are not adequate to protect White Sturgeon and note that existing flows may be further reduced in the future. The petitioners discuss the sufficiency of recent efforts by the Department to review and update recreational fishing regulations in the emergency regulation and long-term regulation processes (e.g. CDFW 2023b) in light of recommendations in the literature (Blackburn et al. 2019). They also address concerns about the role anthropogenic sources of nutrient enrichment have in algal blooms in San Francisco Bay, the Delta, and San Joaquin River, noting that "the Regional Board anticipates requiring load reductions in an updated wastewater nutrient permit, [but] changes in infrastructure and operations required to substantially reduce nutrient loads are likely to take many years to implement." Additionally, the Petition states concerns with the status of the implementation of flow standards or voluntary agreements for the San Joaquin River, and their potential impacts on the White Sturgeon population.

Suggestions for Future Management

Scientific Information in the Petition

The Petition discusses the impact of existing management efforts in "Section 9. Recommendations for Future Management" on pages 35–42 of the Petition. Recommendations include: a) restoring freshwater flows between March and July to support juvenile recruitment, b) reducing or limiting migratory barriers caused by low water quality in the Delta, c) reducing mortality associated with entrainment in water operations, d) eliminating harvest in the recreational fishery by shifting to a sustainable catch-and-release fishery, e) reducing nutrient input into San Francisco Bay, and f) enhancing monitoring and research efforts on White Sturgeon.

Availability and Sources of Information

Scientific Information in the Petition

The Petition provides a list of 68 references in Section 10, "Availability and Sources of Information" on pages 42–49 of the Petition. The cited information sources range from peer-reviewed literature, reports and technical literature, and presentations and documents presented at public meetings. The information cited is current and is presented in accordance with standard scientific practice.

Summary of Petition Components

The above petition components were evaluated by the Department for sufficiency of information pursuant to Fish and Game Code section 2073.5. The Department finds that sufficient information was provided on the petition components. If the Commission accepts the petition for further consideration under CESA, the Department will commence a review of the status of the species at that time.

Recommendation to the Commission

Pursuant to Fish and Game Code section 2073.5, the Department evaluated the Petition on its face and in relation to other relevant information the Department possesses. In completing its Petition Evaluation, the Department has determined that the Petition and other relevant information provide sufficient scientific information to indicate that the petitioned action to list the White Sturgeon as threatened may be warranted. Therefore, the Department recommends the Commission accept the Petition for further consideration under CESA.

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Credit: Rene Reyes

Species Status Assessment for the San Francisco Bay-Delta Distinct Population Segment of the Longfin Smelt

Version 2.0

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Chapter 1 - Introduction and Analytical Approach

This report summarizes the results of a Species Status Assessment (SSA) conducted for the San Francisco Bay-Delta Distinct Population Segment of the longfin smelt (*Spirinchus thaleichthys*) (Bay-Delta DPS). The longfin smelt belongs to the northern smelt family Osmeridae and is one of three extant (currently existing) species in the genus *Spirinchus*. The Bay-Delta DPS was proposed for listing as endangered by the U.S. Fish and Wildlife Service (Service) on October 7, 2022. The longfin smelt has been listed as a threatened species throughout its range in California by the California Department of Fish and Wildlife (CDFW) since 2009, under the California Endangered Species Act (CESA).

In the literature, several different terms to describe the various portions of the San Francisco Bay-Sacramento/San Joaquin River Delta are used. In this document, we refer to these areas, as follows: the "Delta" represents the legal delta, as defined by the Delta Protection Act of 1992, encompassing all waters east of Chipps Island. The "Bay" encompasses all waters west of Chipps Island where the legal delta ends. The "Bay-Delta" and "San Francisco Bay-Delta" and "San Francisco Estuary" are synonymous and encompass both the Bay and the Delta.

In this SSA Report, we focus on the biological information and threats facing the Bay-Delta DPS. The reader is referred to the 2012 12-month finding (77 Federal Register [FR] 19756) for information on the status of the species range-wide. The SSA framework (Service 2016, entire) is intended to support an in-depth review of the species' biology, current and future threats, an evaluation of its biological status, and an assessment of the resources and conditions needed to maintain long-term viability. The intent is for the SSA Report to be easily updated as new information becomes available and to support all functions of the Service's Endangered Species Program from candidate assessment, listing, consultations, and recovery actions. As such, the SSA Report will be a living document upon which other documents would be based, such as listing rules, recovery plans, 5-year reviews, and Endangered Species Act §7 or §10 actions.

Ultimately, this SSA Report for the longfin smelt Bay-Delta DPS will provide the biological support for the decision on whether or not to propose to list the species as threatened or endangered and, if so, where to propose designating critical habitat. Importantly, the SSA Report does not result in a decision by the Service on whether this species should be proposed for listing as a threatened or endangered species under the Act. Instead, this SSA Report provides a review of the available information strictly related to the biological status of the longfin smelt. The listing decision will be made by the Service after reviewing this document and all relevant laws, regulations, and policies, and the results of a proposed decision will be announced in the *Federal Register*, with opportunities for public input as appropriate.

To assess the ability of the Bay-Delta DPS to maintain self-sustaining populations over time, we applied the conservation biology principles of resiliency, representation, and redundancy (Shaffer and Stein 2000, pp. 308–311). As described more fully below, resiliency is the species capacity to withstand environmental and demographic stochasticity and disturbances; redundancy is the species capacity to withstand catastrophes; and representation is the species capacity to withstand novel changes in its

biological and physical environment. A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is better able to adapt to novel changes and to tolerate environmental stochasticity and catastrophes. In general, species viability will increase with increases in resiliency, redundancy, and representation (Smith *et al.* 2018, p. 306).

The 3Rs framework is specifically defined and utilized herein, as:

Resiliency is the ability of a species to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions such as temperature, rainfall), periodic disturbances within the normal range of variation (fire, floods, storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity) (Redford *et al.* 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions. We can best gauge resiliency by evaluating population level characteristics such as: demography (abundance and the components of population growth rate—survival, reproduction, and migration), genetic health (effective population size and heterozygosity), connectivity (gene flow and population rescue), in context with habitat quantity, quality, configuration, and heterogeneity. For species prone to spatial synchrony (regionally correlated fluctuations among populations), distance between populations and degree of spatial heterogeneity (diversity of habitat types or microclimates) are also important considerations.

Redundancy is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population heath and for which adaptation is unlikely (Mangal and Tier 1993, p. 1083). We can best gauge redundancy by analyzing the number and distribution of populations relative to the scale of anticipated species-relevant catastrophic events. The analysis entails assessing the cumulative risk of catastrophes occurring over time. Redundancy can be analyzed at a population or regional scale, or for narrow-ranged species, at the species level.

Representation is the ability of a species to adapt to both near-term and long-term changes in its physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (pathogens, competitors, predators, etc.) environments. This ability to adapt to new environments—referred to as adaptive capacity—is essential for viability, as species need to continually adapt to their continuously changing environments (Nicotra *et al.* 2015, p. 1269). Species adapt to novel changes in their environment by either [1] moving to new, suitable environments or [2] by altering their physical or behavioral traits (phenotypes) to match the new environmental conditions through either plasticity or genetic change (Beever *et al.* 2016, p. 132; Nicotra *et al.* 2015, p. 1270). We can best gauge representation by examining the breadth of genetic, phenotypic, and ecological diversity found within a species and its ability to disperse and colonize new areas.

This SSA Report provides a thorough assessment of biology and natural history and assesses demographic risks, stressors, and limiting factors in the context of determining the viability and risks of extinction for the Bay-Delta DPS. We are required to use the best scientific and commercial data

available to guide us in our listing determinations. As mentioned above, this document is strictly intended to utilize the best available scientific information in evaluating the status of the Bay-Delta DPS. In some cases, information that is pertinent to aspects of the species' biology, life history, ecology, etc., may include primary peer-reviewed literature, 'gray' literature from resource agency reports, information gleaned from professional presentations (e.g., solicited presentations from experts at our Expert Elicitation workshops in late summer 2021), unpublished data, and sometimes personal communications from species experts. For our own novel analyses to support our understanding of the status and trend related to current and future condition of the DPS, our methods are presented below in detail within our technical appendices.

This document reflects an extensive peer and partner review that included 21 invited peer and partner review participants, of which 10 replied with over 962 general and specific comments, and this feedback was incorporated into improvements to the final SSA herein. The format for this SSA Report includes: the Bay-Delta DPS' life history, biological and ecological needs (Chapter 2); the Bay-Delta DPS' current condition, including factors affecting the population (Chapter 3); and the Bay-Delta DPS' anticipated future condition, including changes that could affect the Bay-Delta DPS and analytical population modeling factoring in our best available risk profile for the future trajectory of the DPS (Chapter 4). This document is a compilation of the best available scientific and commercial information and a description of past, present, and likely future risk factors to the Bay-Delta DPS.

Chapter 2 – Species Needs, Life History, and Biology

In this chapter, we provide an overview of biological information available for the longfin smelt Bay-Delta DPS, including its taxonomic history, genetics, morphological description, and known life history traits. We then outline the resource needs of individuals and populations of longfin smelt. Here, we report those aspects of the life history of the Bay-Delta DPS that are important to our analysis. Because information and research related to the longfin smelt most often applies to the species as a whole, any references to longfin smelt below also apply to the Bay-Delta DPS unless specifically described otherwise.

2.1. Species Description and Taxonomy

Adult longfin smelt average 9–11 centimeters (cm) (3.5–4.3 inches (in)) standard length, although some individuals as long as 15 cm (5.9 in) have been observed. The sides and lining of the gut cavity appear translucent silver (see Figure 2.1), the back is olive in color with an iridescent purple hue, and mature males are usually darker in color than females. Longfin smelt can be distinguished from other osmerids by their long pectoral fins (extending past the start of the pelvic fins), weak or absent striations on their opercular (covering the gills) bones, incomplete lateral line, low numbers of scales in the lateral series (54 to 65), long maxillary (jaw) bones (in adults, these bones extend past the middle of the eye), and lower jaw extending anterior of the upper jaw (McAllister 1963, p. 10; Miller and Lea 1972, pp. 158–160; Moyle 2002, pp. 234–236).

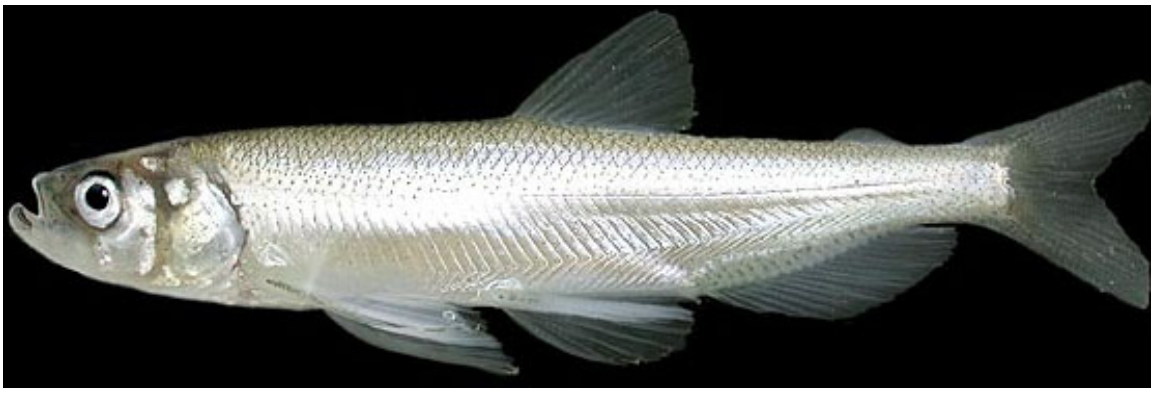


Figure 2.1: Longfin smelt (*Spirinchus thaleichthys*)

The longfin smelt belongs to the northern smelt family Osmeridae and is one of three extant (currently existing) species in the *Spirinchus* genus; the night smelt (*Spirinchus starksi*) also occurs in California, and the shishamo (*Spirinchus lanceolatus*) occurs in northern Japan (McAllister 1963, pp. 10, 15). The longfin smelt range extends scattered through bays, and both small and large estuaries from northern California through Alaska. These include: San Francisco Bay-Delta, Humboldt Bay, Russian River, Mad River, and the Klamath River in northern California (Garwood 2017, p. 98); northward through Yaquina and Coos Bays in Oregon; the lower Columbia River; Skagit Bay, Grays Harbor, and Willapa Bay in Washington; Harrison Lake in British Columbia; to Prince William Sound in Alaska (Moyle, 2002, p. 236), along with a

land-locked population in Lake Washington (a freshwater lake east of the City of Seattle in the State of Washington).

Because of its distinctive physical characteristics, the Bay-Delta population of longfin smelt was once described as a species separate from more northern populations, which were formerly identified as *Spirinchus dilatus* (Moyle 2002, p. 235). McAllister (1963, p. 12) merged the two species *S. thaleichthys* and *S. dilatus* because the difference in morphological characteristics represented a gradual change along the north-south distribution rather than discretely identifiable ones. Stanley *et al*. (1995, p. 395) concluded that longfin smelt from Lake Washington and the Bay-Delta are conspecific (of the same species) despite the large geographic separation but recommended that this isolation indicates that protection and management of this population as a discrete, genetically-independent entity would be best conservation biology practice. Israel and May (2010, p. 230) found moderate levels of differentiation between polymorphic microsatellite markers in the Lake Washington and Bay-Delta population, supporting the conclusion that geographic isolation of these populations has conferred distinctiveness. In our 12-month finding in 2012, we determined that the San Francisco Bay-Delta population of the longfin smelt is a valid DPS, due to discreteness and significance (**77 FR 19756**). More complete genetic analyses, published recently, found high genetic structure between major estuaries, with southern estuaries (north to Columbia River) showing evidence for unidirectional (northward) gene flow, suggesting the SFE population is genetically distinct from the northern populations and an important source population to maintain nearby northern estuarine populations (Saglam *et al.*, 2021, p. 1793).

2.2. Distinct Population Segment Range and Distribution

Longfin smelt have been collected throughout the Bay-Delta (see Figure 2.2) and occasionally even upstream of the Delta in some of its tributary river systems. Longfin smelt have been observed in their winter and spring spawning period as far upstream as Colusa State Park in the Sacramento River, the City of Lathrop in the San Joaquin River system, Hog Slough off the South-Fork Mokelumne River, and in the South Delta near Old River south of Indian Slough (Radtke 1966, pp. 115–119; CDFG 2009a, p. 7; Baxter *et al.* 2010, p. 7; Merz *et al.* 2013, p. 132). They have been detected as far upstream on the San Joaquin River as the confluence with the Tuolumne River (Rosenfield, 2010, p. 6) and also in the Napa and Petaluma Rivers (Merz *et al*. 2013, p. 136), especially during wet winters (Rosenfield 2010, p. 6; Grimaldo *et al.* 2020, pp. 11–12) In recent surveys, longfin smelt were captured in many major and tributary sloughs within the Alviso Marsh Complex salt pond restoration area in the South Bay (Lewis *et al.* 2019, p. 1). Longfin smelt migrate out into the ocean at least as far as the Gulf of the Farallones. Longfin smelt have been detected as far south as Monterey Bay (Garwood 2017, p. 117), which represents the southern-most extent of their range (Eschmeyer and Herald 1983, p. 82), although these detections are generally believed to represent very rare dispersal events.

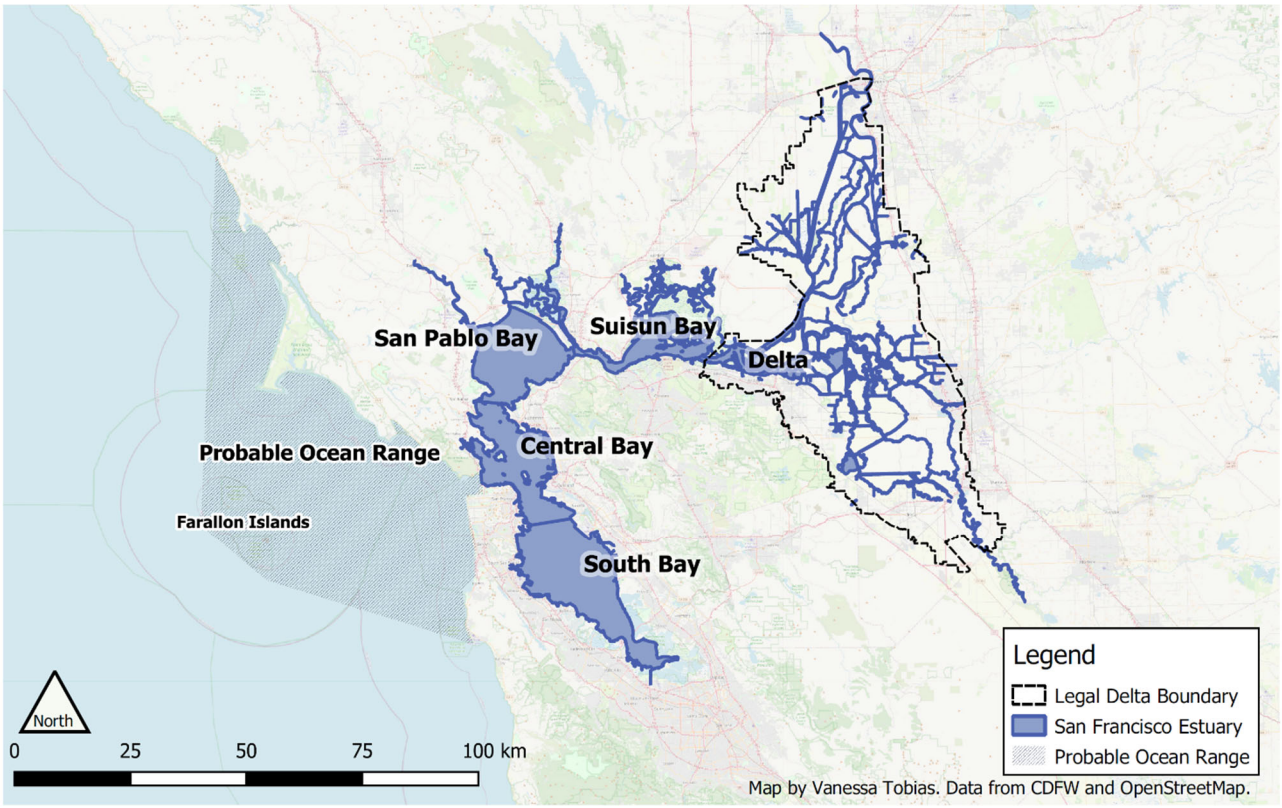


Figure 2.2: Range of the San Francisco Bay-Delta DPS of longfin smelt

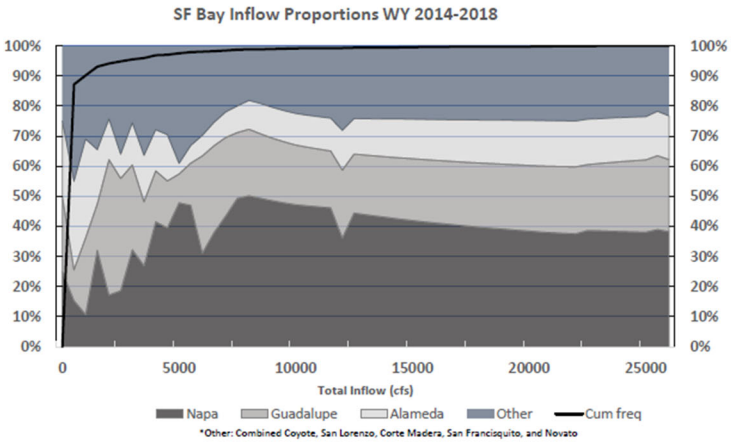
2.3. Ecological Setting

The Bay-Delta is one of the largest estuaries on the West Coast of the continental United States (Sommer *et al*. 2007, p. 271). The modern Bay-Delta bears only a superficial resemblance to the historical Bay-Delta (Whipple *et al.* 2012, pp. 1–28). The region supports an estuary covering approximately 1,235 square kilometers (km²) (477 square miles (mi²)) (Rosenfield and Baxter 2007, p. 1577), receiving almost half of California's freshwater runoff (Lehman 2004, p. 313). The historical island marshes surrounded by low natural levees of the delta are now protected by large, humanmade and rock-reinforced levees, and the islands themselves are intensively farmed (Moyle 2002, p. 32). The watershed, which drains approximately 40 percent of the land area of California, has been heavily altered by multiple dams and water diversions. In the upper Sacramento/San Joaquin estuary, nonnative species now dominate the fish assemblages, both in terms of numbers of species and numbers of individuals (Matern *et al.* 2002, p. 807; Nobriga *et al.* 2005, p. 781; Feyrer and Healy 2003, p. 126). The Bay Institute has estimated that intertidal wetlands in the Delta have been diked and leveed so extensively that approximately 95 percent of the 141,640 ha (350,000 ac) of tidal wetlands that existed in 1850 are gone (The Bay Institute 1998, pp. 4–17).

Broadly, the Bay-Delta consists of several bays and the Delta (Figure 2.2). The San Francisco Bay consists of the San Pablo Bay in the north that receives freshwater input from the Sacramento-San Joaquin system (with additional inputs from the Napa, Sonoma, and Petaluma Rivers), the Central Bay, and the

South Bay in the south that receives little freshwater input (e.g., Alviso Slough) (Largier 1996, p. 69). Dominant fish species in the San Francisco Bay (including San Pablo and South bays) are highly salt-tolerant and include forage fishes such as northern anchovy and Pacific herring. There are also abundant recreationally important fish, including striped bass, white sturgeon, California halibut, leopard shark, and surf perches. In the estuary east of San Pablo Bay, major habitat types include lower salinity embayments, riverine and tidal wetlands, mud flats, and tidal marshes, with substantial areas of diked wetlands managed for waterfowl hunting in Suisun Marsh.

Everywhere freshwater flow enters the San Francisco Estuary, it can generate low-salinity habitats for plants and animals that are adapted to brackish water conditions. Inflow into the Delta represents an average of 90% of the estuary's freshwater and as such, it has the largest influence on estuarine habitat conditions (Jassby *et al.* 1995 p. 275, and Fig. 4, p. 279; Monismith *et al*. 2002, Fig. 7, p. 3010). The southern part of San Francisco Bay, which has very limited freshwater inputs is generally characterized as a lagoonal system, whereas the northern reaches function as a tidal river estuary due to the much larger freshwater flow inputs from the Delta (Kimmerer 2004, p. 7). However, the small tributaries can have important localized effects; for instance, supporting reproduction of longfin smelt during periods of high discharge (Lewis *et al.* 2019, p. 3). For a summary of flow data contributed by various tributaries to the Estuary, see Figure 2.3.



NAME	AGENCY	ID	DRAINAGE AREA (mi²)	Q annual var POR (ac-ft)	% Inflow '14-18
NAPA R NR NAPA CA	USGS	11458000	218	157600	33.7
GUADALUPE R ABV HWY 101 A SAN JOSE CA	USGS	11169025	160	51000	22.6
ALAMEDA C NR FREMONT CA	USGS	11179100	638	21700	16.5
COYOTE C AB HWY 237 A MILPITAS CA	USGS	11172175	319	31900	12.3
SAN LORENZO C A SAN LORENZO CA	USGS	11181040	45	15800	4.7
CORTE MADERA C A ROSS CA	USGS	11460000	18	19700	4.6
SAN FRANCISQUITO C A STANFORD UNIVERSITY CA	USGS	11164500	37	15200	3.9
NOVATO C A NOVATO CA	USGS	11459500	17	8900	1.7

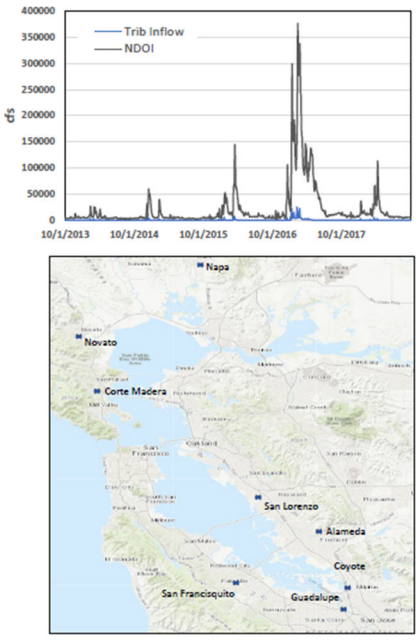


Figure 2.3: Summaries of flow data from gauged tributaries to San Francisco Bay for water years 2014-2018 (see map inset for locations). The upper left panel shows the relative contribution as a function of increasing wetness in a format borrowed from Kimmerer (2004, Fig. 5, p. 16). The lower left panel is a tabular data summary. The upper right panel is a flow time series showing the peak wet year flows in water year 2017 for the net Delta outflow index (black) and the combined Bay Area tributary flow (blue).

Tidal flows that enter the estuary from the Pacific Ocean, and inflowing river water, present opposing forces because these two sources of water have inherent differences in density; saltwater is denser than fresh water. The ocean delivers more force during spring tides than neap tides, and the rivers deliver more force when they run high than when they run low. These opposing hydraulic forces interact with the estuary's bathymetry to create extremely variable and complex hydrodynamic mixing of fresh- and saltwater. Where saltwater and freshwater come into contact, vertical and lateral mixing currents are intermittently generated (Stacey *et al*. 2001, pp. 17026–17035). During periods of high outflow, salinity can be very different at the water surface than at the bottom due to freshwater flowing over the top of saltier water (Mac Williams *et al*. 2015, Fig. 8, p. 20). This stronger vertical salinity gradient when Delta outflow is elevated creates more surface area for mixing of fresh and brackish water and can result in water near the bottom having a net flow landward while overlying water has a net flow toward the ocean (Monismith *et al.* 2002, Fig. 15, p. 3014).

The changes to the bathymetry of the estuary that occurred between 1850 and circa 2008 (marsh reclamation, channel dredging, extensive leveeing) have collectively made the estuary deeper and less hydrodynamically-connected to the surrounding landscape (Andrews *et al.* 2017, Fig 5, p. 64). The deeper waterways create more space for the tides to bring salt landward. Specifically, landscape changes since 1850 are estimated to have resulted in an average landward shift of X2 of 3.23 km (Andrews *et al.* 2017, p. 68). In contrast, the changed freshwater flow hydrograph (discussed in the following paragraph) has caused a winter-spring landward shift of X2 on the order of 10-20 km. Changes to the estuary's landscape also have caused the tide to reach further into Old and Middle rivers (Andrews *et al.* 2017, p. 66) which, as discussed below, is necessary to support water exports from the Delta.

Most wetland conversion had occurred by about 1920 (Whipple *et al*. 2012, Fig 1.16, p. 24). Most large water storage projects were developed from 1910-1970, with the pace of development being especially rapid from 1945-1970 (Cloern and Jassby 2012, Fig 5, p. 6). By the early 1960s, there was more reservoir storage capacity in the estuary's watersheds than the historical average annual runoff from those watersheds. From the 1940s on, most of the larger reservoirs were serving the Federal Central Valley Project (CVP) or California State Water Project (SWP), with some exceptions such as on the Yuba, Merced, and Tuolumne rivers. The CVP and SWP annually decrement (reduce) an average of about 5 million acre-feet (MAF) of Delta outflow whereas non-project reservoirs and diverters annually decrement about 8 MAF of potential Delta outflow (Hutton *et al.* 2017, Fig 4, p. 8). The cumulative development of this annual average of about 13 MAF of surface water supplies has resulted in a long-term decline in Delta outflow during February-June relative to estimates of what flows would have been absent water development (Gross *et al.* 2018, Fig. 6, p. 12; Reis *et al*. 2019, Fig. 3, p. 12). This in turn has increased the frequency of very low outflows that prior to water development would have been very rare and associated only with extreme drought (Reis *et al.* 2019, Fig. 3, p. 12).

Taken together, the landscape changes discussed above, and changes to the estuary's flow regime have changed how mixing processes function, and thus altered ecological opportunities for the estuary's biota to potentially exploit. Studies of suspended sediment concentrations have confirmed the pattern mentioned above--that water mixing is influenced by the bathymetric 'shape' of the estuary, the spring-neap tidal cycle, and wind and wave action over shallower areas (Schoellhamer 2000, Fig. 3, p. 349; Ruhl

et al. 2001, Fig. 7, p. 808; Bever *et al.* 2018, Tables 6-7, p. 1960). These forces all work to aggregate (and disaggregate) sinking particles like sediment and phytoplankton.

A common metric of estuarine mixing and other habitat conditions in the low-salinity zone is X2, the distance in kilometers from the Golden Gate to the place where salinity near the bottom of the water column is 2 practical salinity units (PSU; also parts per thousand, Jassby *et al.* 1995, pp. 274–275). The 2 PSU isohaline was chosen in part because it represents the approximate upstream limit of where surface and bottom salinity diverge, and because historical turbidity and plankton aggregations were broadly associated with it. Estuarine pelagic fishes, including the Bay-Delta DPS, are also associated with this location (Dege and Brown, 2004, Fig 3, p. 57).

The X2 isohaline is always moving, reflecting changes in the size, shape, and ecological function of the low-salinity zone (Mac Williams *et al*. 2015, Figs. 11-12, p. 22). Tidal flows affect X2 most strongly over short time scales (hours to weeks; Kimmerer 2004, Fig. 2, p. 12). Over longer time scales, Delta outflow has the dominant influence on X2 (Jassby *et al.* 1995 p. 275, and Fig. 4, p. 279; Monismith *et al.* 2002, Fig. 7, p. 3010). The surface area of the low salinity zone increases very rapidly as it begins to include a large part of San Pablo Bay (X2 ≤ 55 km), resulting in peak low salinity zone areas of 150 to 250 square kilometers (Mac Williams *et al.* 2015, Fig. 12, p. 22). In the contemporary estuary, X2 moves seaward of 55 km only very rarely (Gross *et al.* 2018, Fig. 6, p. 12). However, in the pre-development estuary, Gross *et al.* (2018, Fig 6, p. 12) estimated that X2 reached monthly averages from February through May of ≤ 55 km in about half of all years. As described in more detail in section 3.1.1 of this SSA, Delta outflow, X2, and other co-linear indicators of wet versus dry conditions during the winter and spring have been statistically associated with first year recruitment and survival of longfin smelt.

2.4. Life History and Biology

Longfin smelt in the Bay-Delta are pelagic (most frequently occurring in open water habitats) forage fish that exhibit a facultatively anadromous life history whereby migration to sea is not required to complete the lifecycle (Moyle 2002, p. 236). Longfin smelt are generally adapted to cold- and cool-water habitats so elements of their facultatively anadromous life cycle within the SFE are influenced by seasonal water temperature variation (e.g., Jeffries *et al.* 2016, p. 1712; Yanagitsuru *et al.* 2021, Fig. 1). The adults reproduce in low-salinity to freshwater habitats beginning in early winter and extending into the spring as water temperature allows. The larvae rear during the spring in locations near where they were spawned. As water temperatures warm each spring into early summer, the young fish move seaward and many individuals move into the Pacific Ocean during the summer months. It is speculated that some of these fish may spend extended periods of time at sea, but many individuals return to the estuary beginning in the fall and continuing into the early winter. These returning fish appear to be a combination of fish getting ready to spawn and younger individuals that are unlikely to do so.

2.4.1. Spawning

The longfin smelt life cycle is at least two years in duration (Rosenfield and Baxter 2007, Fig 6). It is believed that spawning is semelparous (occurring only once). Although some individuals may spawn as 1- or 3-year-old fish before dying (Moyle 2002, p. 236), this has not been confirmed in the San Francisco

Estuary based on age-validated individuals. Longfin smelt migrate from the ocean and bay habitats to low-salinity habitats beginning in the fall, and spawn in fresh and low-salinity water in the winter and spring.

pp. 38–39

p. 18;

p. 529

(Grimaldo *et al.* 2017, p. 6; CDFG 2009a, p. 16; Meng and Matern 2001, p. 755; Wang 1986, pp. 6-10). There has been recent confirmation of persistent and occasionally dense aggregations of adult longfin smelt in the sloughs of Coyote Creek in the South Bay, although recruitment success was confirmed during wet years only (Lewis *et al*. 2020, p. 1)

The spawning period of longfin smelt in the Bay-Delta may begin as early as November and historically lasted until as late as June, although spawning more typically occurs from January to April (CDFG 2009a, p. 10; Moyle 2002, p. 236). Observations from studies of longfin smelt in the San Francisco Estuary identify the longfin smelt spawning period as December through April based on ripe females (Radtke 1966, p. 116), and November through April based on a roughly month-long incubation time period combined with when the presence of yolk-sac larvae have been observed in the environment (Hieb and Baxter 1993, p. 110). Using incubation duration information and monthly densities of newly hatched yolk-sac larvae, Baxter (2021, pers. comm.) estimates infrequent spawning in November, increasing and regular spawning in December, a peak in January or February followed by a sharp decline through April. Within the Bay-Delta, adult longfin smelt spawning starts when water temperature drops below ~14 °C (57.2 °F) and becomes consistent when water temperatures remain below 13 °C (55.4 °F) (CDFG 2009a, p. 11). However, recent studies indicate successful spawning may require temperatures of 13 °C (55.4 °F) or lower (Baxter 2016, pers. comm.). A minimum spawning temperature of 5.6 °C (41 °F) was reported in lab studies (Wang 1986, pp. 6–9). This lengthy spawning period may confer a degree of ecological resilience, as it may allow the Bay-Delta DPS to spread reproductive effort across the winter and into the spring, particularly during wet years.

Spawning areas have been inferred based on collection of recently hatched yolk-sac larvae (Grimaldo *et al*. 2017, Fig 9; Lewis *et al*. 2019, p. 3). They are believed to vary from year to year, depending primarily on the distribution of fresh and low-salinity water at the time of spawning (Grimaldo *et al.* 2020, pp. 11-12) and likely on where and how long the adult fish can find suitable salinity water in a temperature range conducive to egg survival (Yanagitsuru *et al.* 2021, Fig 1).

While the spawning behavior of longfin smelt has not been observed in the SFE, it has been hypothesized that they make short runs upstream into fresh water and low salinity < 2 psu habitats, possibly at night (CDFG 2009a, p. 12; Rosenfield 2010, p. 8). This pattern is similar to documented spawning behavior for the Lake Washington population, in which spawners make overnight runs into tributaries of the lake then return to the lake before dawn (Dryfoos 1965, 61; Moulton 1974, pp. 49-50). Catch patterns of mature longfin smelt in Coyote Creek, in the South San Francisco Bay, have suggested that maturing fish stage downstream in brackish waters, males move upstream before females, likely to

prepare spawning substrates and await ripe females to make spawning runs upstream (Lewis *et al.* 2019, p. 101). Longfin smelt spawn negatively buoyant (demersal), adhesive eggs that are about 1 mm in diameter (Dryfoos 1965, p. 42; Chigbu and Sibley 1994, p. 12; Wang 2007, p. 39). The specific spawning substrates selected by longfin smelt in the San Francisco Estuary remain unknown.

2.4.2. Larval Hatching, Development, and Behavior

In the San Francisco Estuary, longfin smelt larvae hatch between December and May, with rare observations outside this range (Baxter 1999, p. 180). Peaks in abundance of recently hatched yolk-sac larvae occurred most commonly in February (during 8 of 10 years) and March otherwise (in 2 of 10 years; Baxter *et al.* 1999, p. 183). More recent larval fish sampling exhibited a similar pattern, where peak hatching occurred most commonly in February, but also occasionally in late January or March (Baxter 2021, pers. comm.).

Hatch timing is determined by when fish spawn and the temperature at which embryos incubate with incubation time decreasing with increasing water temperatures (see Figure 2.4). At 7 °C (44.6 °F), embryos hatch in 40 days (Dryfoos 1965, p. 42). Sibley and Brocksmith (1995, p. 38) reported an average incubation duration of 29 days at water temperatures ranging from 8 to 9.5 °C. Similarly, Moulton (1970, p. 50) noted that incubation time averaged 25 days at temperatures ranging between 9.6 and 10.6 °C. Hobbs *et al.* (2013, p. 49) incubated eggs at warmer temperatures than any of the studies mentioned above (12 ± 1 °C) and found the shortest mean incubation duration (16 days). More recently, Yanagitsuru *et al.* 2021 found that incubation took an average of 23.7 days at 9 °C, 19.3 days at 12 °C, and 16.5 days at 15 °C.

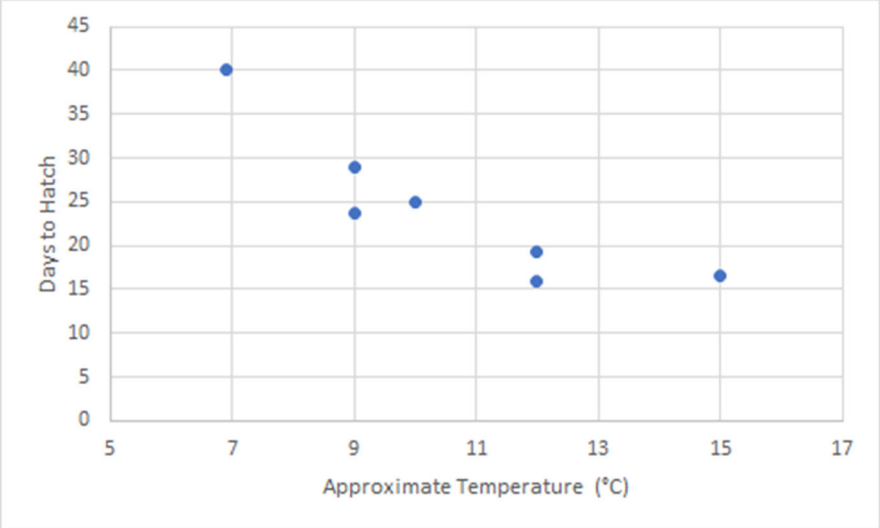


Figure 2.4: Relationship between temperature and days to hatch for longfin smelt embryos (Source: Yanagitsuru et al. 2021, Supplementary Table 2).

There are two prevailing theories about how longfin smelt use hydrodynamics to facilitate the transport and retention of their larvae in the low-salinity zone. An older hypothesis suggests that once hatched

from spawning areas largely upstream of the low-salinity zone, longfin smelt larvae are predominantly surface-oriented and dispersed widely by river flows and tidal currents, particularly when Delta outflow is high (CDFG 2009a, p. 8). A newer hypothesis is that most eggs are spawned in the low-salinity zone as well as the smaller low-salinity zones of the Bay Area tributaries (e.g., Lewis *et al.* 2019, p. 3), such that the larvae hatch directly into suitable rearing habitats (Grimaldo *et al.* 2020, p. 12). Larval retention is then due to a combination of swimming behaviors (Bennett *et al.* 2002; Kimmerer *et al.* 2014) and vertical and lateral mixing of fresh- and brackish water that has been well-described in the estuary (see Section 2.2). Regardless of how correct or dominant each reproductive conceptual model is, there is good data support for the importance of the low-salinity zone as a critical rearing habitat for the early life stages of longfin smelt (Dege and Brown 2004, Fig. 3, p. 57; Bennett *et al.* 2002, p. 1502; Hobbs *et al*. 2010, Fig. 5 p. 565; Grimaldo *et al.* 2020, Fig. 6, p. 10).

2.4.3. Maturity and Fecundity

Longfin smelt maturation begins in the fall with mature fish observed as late as May of the following year (Tempel and Burns 2021, slide 3). Longfin smelt are sexually dimorphic, where males darken in color and the base of their anal fin hardens and elongates, presumably for sweeping fine sediments from spawning sites (Wang 1986, pp. 6-10). Most longfin smelt that exhibit onset of maturation are > 90 fork length (FL) (Baxter 2021, pers. comm.), while fecundity increases exponentially as a function of female size, and ranges from about 1,900 eggs in a 73 mm female to over 16,000 in a 132 mm female (CDFG 2009a, Fig. 3, p. 11). Studies of longfin smelt fecundity for the Lake Washington and Harrison Lake populations also yielded similar results, with fecundity tending to be a function of both size and feeding success (Dryfoos 1965, p. 120; Chigbu and Sibley 1994, pp. 7-8). Since hatching is primarily observed annually from January onward, we use January 1 as the anniversary "hatch date" for incrementing age in this SSA. Thus, subadult fish in their first year of life (defined as age-0) will all become age 1 on January 1 and enter their second year of life. And age-1 fish will increment to age 2 on January 1.

2.4.4 Environmental Parameters Influencing Bay-Delta DPS Growth Rates and Length at Age

There is little published information about the growth rates of longfin smelt, but most of the published information available comes from studies on the Bay-Delta DPS. Souza *et al.* (2006, Fig. 1, p. 4) estimated that larval longfin smelt grew at a mean apparent rate between 0.12 and 0.23 mm/day as they reached the juvenile stage. Growth rates of very young larvae (3-5 days post-hatch) have also been studied in the lab (at various temperatures and salinities) for the purpose of identifying ideal conditions for captive rearing (Yanagitsuru *et al.* 2021, entire). In an otolith study, longfin smelt were found to grow slowly up to about 20 days post-hatch, and at a slightly faster rate up to about 120 days of age (Hobbs et al. 2013, Fig. 7, p. 79).

To assess LFS apparent growth, we modified the Lester *et al.* (2004, entire) biphasic growth model to estimate the impacts of factors on somatic growth patterns using a similar model developed by the author (Matthias *et al*. 2016, entire). Our primary objective was to quantify the strength, direction, and potential interactions of abiotic and biotic factors influencing Bay-Delta DPS growth. Our secondary objective was to assess for systematic or long-term changes or patterns in Bay-Delta DPS growth. Our model was designed to assess variation in apparent growth of longfin smelt using a combination of

observed environmental conditions (e.g., water temperature, conductivity, Delta outflow, density, and prey availability) and random effects. Details on data used, methods, and application, as well as more detailed results are available in Appendices C and D.

Our growth model results support the conclusion that environmental conditions do influence Bay-Delta growth patterns. The best model included abiotic, biotic, and random effects on the growth increment (see Figures 2.5 and 2.6) and abiotic effects on the length-at-age-0. Abiotic effects on growth were temperature squared, Delta outflow, and a temperature-outflow interaction. Biotic effects represented density using abundance indices and time with associated interactions and prey availability (*Eurytemora Afinnis* and *mysis* shrimp indices). For both abiotic and biotic effects, the main effects and interactions were all significant (except for the mysis index effects during Phase-1 growth) and the effects were different between the two growth phases (Fig. 2.5). The average length of each age class of longfin smelt varies from year to year, but we did not detect systematic trends in length-at-age over time (Fig. 2.6).

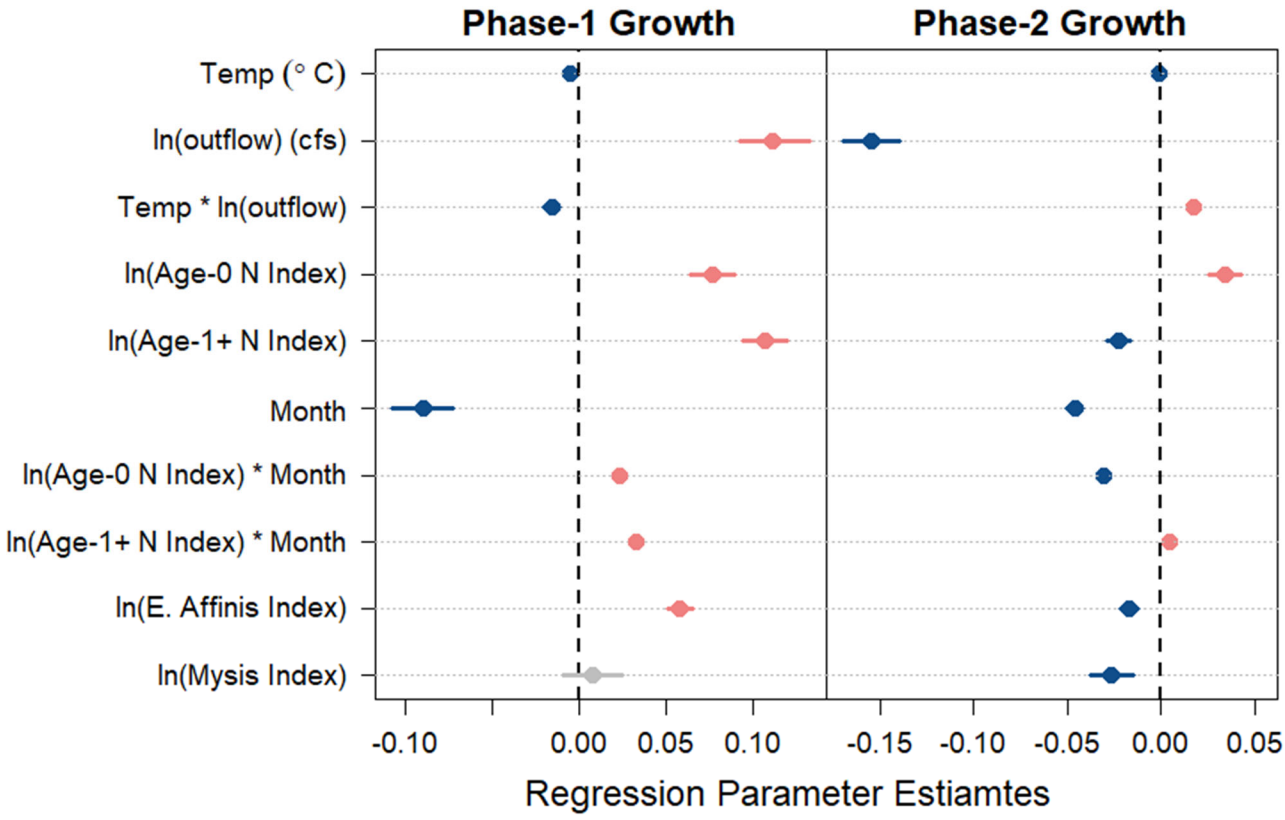


Figure 2.5: Parameter estimates for relationships between environmental variables and growth rates in a biphasic growth model. Colors highlight the direction of relationships: blue indicates a negative relationship, pink indicates a positive relationship, and grey indicates a relationship where the direction may be flat.

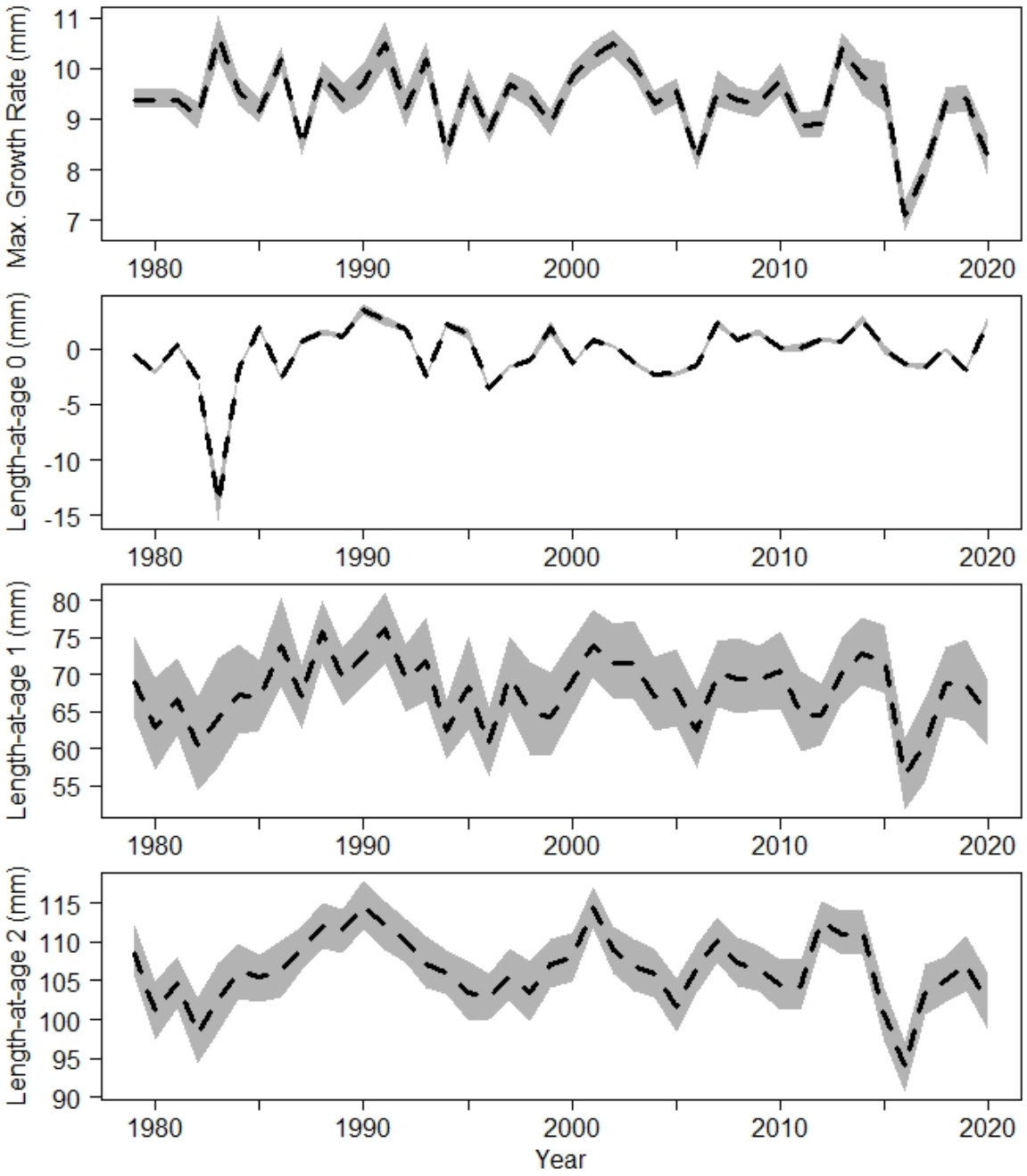


Figure 2.6: Estimated temporal trends in growth patterns from 1979 to 2020. The maximum growth rate (mm; top panel) accounts for variation in the predicted Phase-1 growth and a year-specific random effect predicting the unobserved annual variation in growth patterns. Panels for predicted length-at-age 0, 1, and 2 represent age in years (i.e., 0, 12, and 24 months).

2.4.5. Geographic Distribution and Habitat

Larvae

The spatial distribution of larvae (<20 mm length) within the San Francisco Bay-Delta has not been fully resolved due to lack of adequate coverage by monitoring programs (Grimaldo *et al*. 2017, Fig. 5, p. 1777; 2020, Fig. 6, p. 10). The majority of larvae are affiliated with the estuary's major low-salinity zone generated by the mixing of freshwater outflow from the Delta with the brackish waters of the estuary (see Section 2.2). However, larvae can also be found in tributaries when flows from those tributaries are high enough and temperatures low enough to support egg survival and hatching (Lewis *et al.* 2019, p. 3). The spatial distribution of these larvae reflects the year-to-year variation in the geographic location of the low-salinity zone (Dege and Brown 2004, Fig. 3, p. 57; Grimaldo *et al.* 2020, Fig. 6, p. 10).

Within the low-salinity zone and adjacent waters, larvae have been commonly collected in both littoral (nearshore) and pelagic (offshore) habitats. Upon hatching, the larvae may swim toward the water surface which would facilitate relatively rapid seaward transport (CDFG 2009a, p. 8). However, it is not clear that such a behavior would facilitate retention in the low-salinity zone, especially when Delta outflow is high (Kimmerer *et al*. 2014, Fig. 5, p. 910). Using a 3D hydrodynamic modeling framework, Kimmerer *et al*. (2014, Fig 5, p. 910 and Fig. 6, p. 911) applied the relatively modest swimming capabilities of copepods to show how well simple behaviors could help planktonic animals avoid being washed out to sea and keep them loosely associated within particular salinity ranges. Copepods are considerably smaller than larval fishes, and if they are able to influence their own location in the estuary, it follows that longfin smelt larvae may possess this capacity as well (Bennett *et al.* 2002, p. 1502). The recent findings of larval densities in tidal marsh channels and other edge habitats in densities comparable to offshore waters provides another potential low salinity zone retention mechanism since tidal currents are slower over shallow shoals and associated marsh channels (Bever *et al*. 2016, Fig 8b, p. 15).

Juveniles through Adults

Aggregated survey data have been used to show that juveniles (>20 mm in length) have been detected at one time or another throughout the estuary and into some tributaries to the Delta above tidal influence (Merz *et al.* 2013, Fig. 2, p. 132). However, the spatial distribution of juveniles shows a distinct seaward migration as water temperatures warm in the late spring and early summer (Rosenfield and Baxter 2007, p. 1590).

Juveniles have been collected most frequently from deep water habitats as opposed to shoals (Rosenfield and Baxter 2007, p. 1586). In Lake Washington, age-0 and age-1 longfin smelt favor deep water during daylight and move closer to the surface at night (Quinn *et al.* 2012, p. 342), likely moving in relation to their major source of food, mysid shrimp (Chigbu *et al.* 1998, p. 180). It is possible that the Bay-Delta DPS does so as well, but this has not been evaluated for post-larval fish. Selection for deep water and a general shift to marine habitat were hypothesized to be behavioral responses to seasonally increasing water temperatures (Rosenfield and Baxter 2007, p. 1590).

Phillis *et al.* (2021, entire) utilized boosted regression trees and concluded that the strongest predictors of juvenile longfin smelt catch in the 20-mm Survey were bottom salinity, Secchi depth, Julian Day,

temperature, surface salinity, and the 7-day average position of X2. The same study predicted larval habitat availability during March through July under low and high spawner abundance in dry, moderate, and wet years (see Figure 2.7). These authors also predicted that, in dry years, habitat distributions shifted to Suisun Bay and north San Pablo Bay. Whereas in moderate flow years, their analysis predicted that higher freshwater flows resulted in lower salinity into areas of San Pablo Bay, and habitat suitability was predicted to increase in the South San Francisco Bay. In wet years, they predicted high suitability habitat is available in Suisun Bay, San Pablo Bay, and some of the South San Francisco Bay.

Based on otter trawl survey data, juvenile longfin smelt rapidly adapt to and inhabit increased salinities because about half the juveniles captured by the larval net came from the salinity range 8 to 24 parts per thousand (ppt) (Baxter *et al.* 1999, pp. 189-190), well seaward of X2. This increase in salinity distribution represents both seasonal increases in upper estuary salinity as outflow declines and downstream movement of some individuals (Baxter *et al.* 1999, p. 191). By their first summer of life, juvenile longfin smelt inhabit salinities up to and including marine water (i.e., 32-33 psu; Baxter *et al.* 1999, p. 191; Rosenfield and Baxter 2007, p. 1590; Kimmerer *et al.* 2009, p. 385).

By May of most years, young-of-the-year longfin smelt begin to reach 40 mm FL (Rosenfield and Baxter, 2007, p. 1581). At this size, and regardless of outflow, these approximately 40 mm young of the year are typically distributed throughout the estuary (Baxter *et al.* 1999, p. 189; Merz *et al.* 2013, pp. 136–139). They are found from low salinity (and occasionally freshwater) on the upstream end of the Bay-Delta DPS' range, to marine conditions on the downstream end. Distributions of older age-0 and age-1 fish have only been described coarsely into densities across shoal (<7m depth) and channel (≥7m depth) habitats. For both age groups, density was almost always higher in the deeper channel habitats, and significantly higher from the first fall through the second spring of life, and between the second fall and second winter of life (Rosenfield and Baxter 2007, p. 1586).

In any given month, Bay Study data indicate that some fraction of the LFS population remain in the Bay, but an unknown fraction may be found in the ocean (Rosenfield and Baxter 2007, p. 1590; Merz *et al.* 2013, p. 142). Longfin smelt have been detected in the nearshore ocean off of San Francisco (Garwood 2017; City of San Francisco and CH2M Hill 1984 and 1985, entire). In addition, Feyrer *et al.* (2015) found a statistical association between the North Pacific Gyre Oscillation (an index of) and age-0 longfin smelt catch in the Bay Study. For this correlation to have any mechanistic basis, longfin smelt would need to be present in the ocean. These observations all support the hypothesis that at least partial anadromy is a life history strategy used by the Bay-Delta DPS, which is consistent with the pattern observed in other populations range-wide (Rosenfield and Baxter, 2007, p. 1590). Recent longfin smelt otolith analyses have supported the conclusion; Lewis *et al.* (2019, p. 63) used isotope ratios in otoliths and indicated that longfin smelt may exhibit at least four unique life history strategies. Another perspective is that the fish may be displaying a single life history strategy within a continuum, spawning in waters that are fresh to slightly brackish and then consistently transitioning into waters too saline to be discerned using strontium. The important indication is that component life stages of the DPS display variable spatio-temporal distribution as part of its life history strategy.

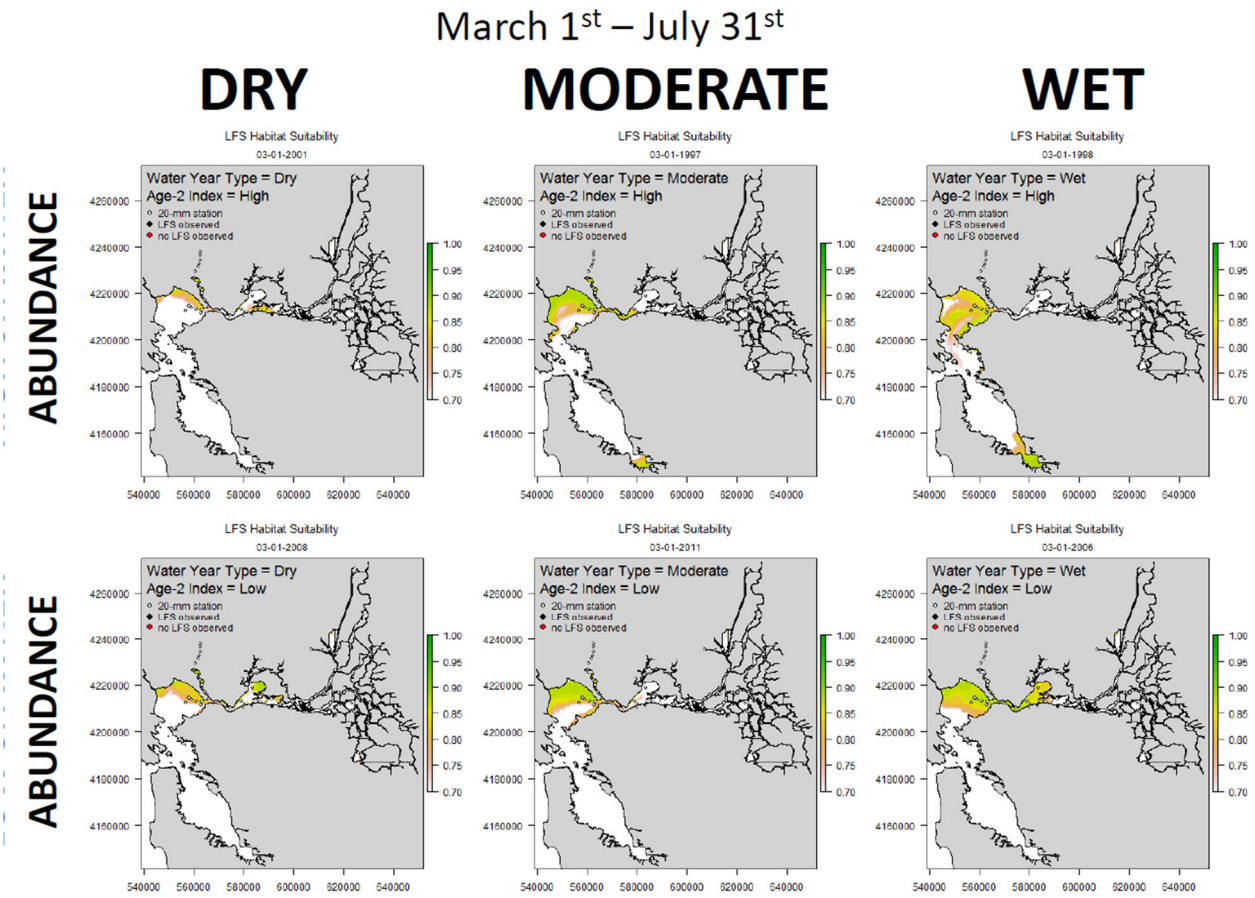


Figure 2.7: Predicted juvenile habitat availability under various scenarios based on boosted regression tree models. Source: Phillis *et al.* 2021, unpublished data

2.5. Diet

Larval longfin smelt select strongly for the calanoid copepod *Eurytemora affinis* as prey; all other prey types combined account for only about 10% of the diet (Barros *et al*. 2022, Fig 6a and 6c, p. 1775). Although, dietary DNA analysis indicates *Acanthocyclops* spp. may also be a substantial dietary component at times (Jungbluth et al. 2021, p. 10). When longfin smelt reach about 1 inch in length, the diet switches and is nearly all mysids, a taxonomic group of larger crustaceans commonly called opossum shrimp (Barros *et al.* 2022, Fig 6b, p. 1775). This finding of a highly specified diet applies to fresh- and brackish-water habitats throughout the estuary (Barros *et al.* 2022, Fig. 2. p. 1768). The currently dominant mysids in the estuary's low-salinity habitats are nonnative species introduced in the 1990s that are smaller than the historically dominant native species, *Neomysis mercedis*, though they are still considerably larger than *E. affinis*. The non-native mysids are also less abundant than *N. mercedis* was in the 1970s and 1980s (Winder and Jassby 2011, Fig 9, p. 685). Both of longfin smelt's primary prey taxa had historical peaks in abundance in the estuary's largest low-salinity zone generated by outflow from the Delta (Kimmerer 2002a, Fig. 2, p. 45). This meant these peak abundances of prey were spatially associated with similar peaks of larval and small juvenile longfin smelt. The abundance of young longfin smelt's primary prey taxa and their historical relationships to X2 variation were disrupted

by food web changes associated with the overbite clam (*Potamocorbula amurensis*) (see Table 2.1). Longfin smelt must switch prey again when they occupy mesohaline to marine habitats as brackish water prey become unavailable. However, what taxa constitute their marine prey has not been evaluated.

Table 2.1. Changes to abundance and X2 relationships for *Eurytemora affinis* and *Neomysis mercedis* following the invasion of the San Francisco Estuary by overbite clam (OC), *Potamocorbula amurensis*. Summarized from Kimmerer (2002, Fig. 7, p. 47). Note that Kimmerer also evaluated the trends for E. affinis for June-October, but by this time of year longfin smelt have outgrown copepod-sized prey (Barros *et al.* 2022, Fig 6b, p. 10).

Season analyzed	Prey taxon	Pre-OC relationship to X2	Post-OC relationship to X2
March-May	*E. affinis*	No significant relationship	Lower mean density; significant inverse relationship (higher abundance when X2 moves seaward in response to high Delta outflow)
June-October	*N. mercedis*	Significant inverse relationship (higher abundance when X2 moves seaward in response to high Delta outflow)	Lower mean density; significant positive relationship (higher abundance when X2 moves landward in response to low Delta outflow)

2.6. Environmental Requirements of the Distinct Population Segment

This section summarizes the key known ecological requirements for the survival and reproduction of longfin smelt at the individual level at each life stage.

2.6.1 Temperature Range

Longfin smelt larvae appear to be adapted to cool water conditions. An experiment testing embryo hatching success at 9, 12, and 15 °C found hatching success generally decreased as temperature increased, but so did variation in hatching success (Yanagitsuru *et al.*, 2021, Fig 1, p. 5). Embryos also tended to hatch at smaller size as experimental temperature treatments increased (Yanagitsuru *et al.* 2021, Fig 2a, p. 6). Yolk-sac larvae (notochord length 5-7 mm) in the 15 °C treatment grew more slowly than larvae in the 9 and 12 °C treatments (Yanagitsuru *et al.*, 2021, Fig 3a, p. 7). Collectively these results indicate that temperatures approaching and exceeding 15 °C impair larval viability, and hence this benchmark may be ecologically significant. It is logical to presume that the Bay-Delta DPS, inhabiting as it does the most southern portion of the species range, has historically been (and is) at the uppermost temperature tolerance range of the species.

Field surveys have detected peak yolk-sac larval (4-9 mm total length) numbers in water temperatures between 8 and 12 °C, corresponding to the early rearing months (Grimaldo *et al.* 2017, p. 8). At later larval stages, longfin smelt are still likely restricted to water temperatures below 20 °C. Jeffries *et al.* 2016 (p. 1709) found that larger larvae (standard length 15-19 mm, 40 days post-hatch) exhibit cellular stress at 20 °C, suggesting that this may be nearing the upper limit of their extended temperature tolerance. In general, age-1 fish inhabit lower temperature water than age-0 fish, although both age classes inhabit 16 -18 °C water in summer and fall (Baxter, 2009, fig 8., p 191). Adults are thought to be limited by water temperature >22 °C during the summer, and likely spend the majority of this time in cooler Bay habitats and the open ocean. As no standardized field surveys currently exist beyond the Golden Gate Bridge, the oceanic distribution and habitat use of longfin smelt remains largely unknown.

After returning from the ocean, and subsequently maturing in the estuary, the fish return to spawn where water quality conditions are favorable for egg survival. These conditions vary in location depending on delta outflow and the location of the LSZ, as well as flows from Bay Area tributaries. Although spawning can start once water temperatures drop below 16 °C (60.8 °F) (CDFG 2009a, p. 11), other information suggests temperatures of 13 °C and maybe lower are more ideal (Baxter 2016, pers. comm.; Tempel and Burns 2021, slide 12).

2.6.2 Salinity tolerance

In laboratory tests evaluating salinity tolerance, yolk-sac larvae (notochord length 5-6 mm) survived the longest and grew the largest at 5 and 10 ppt (Yanagitsuru *et al.* 2022, p. 6). Yolk-sac larvae were also able to maintain water balance equally between 0.4 and 10 ppt, but were unable to survive at 32 ppt, which is the equivalent to marine salinity. On the other hand, yolk-sac larvae experienced stalled yolk resorption in freshwater. In a separate laboratory study examining fertilization and embryo incubation

salinities, fertilization and incubation of embryos was successful at 0.4 and 5 ppt, but fertilization success was significantly greater at 0.4 ppt (Rahman *et al*. 2023, p. 5).

In field surveys, peak yolk-sac larval (4-9 mm total length) densities have been found at 2-4 ppt (Grimaldo *et al.* 2017, p. 8), which is also concordant with the early life salinities that produce highest survival to later life stages (Hobbs *et al.* 2010, p. 564). Longfin smelt lab studies, however, did not test larval tolerance to 2-4 ppt salinity. CDFW's 20-mm Surveys have also shown peak larval (mean length of 20.2 mm) distribution near X2 (Dege and Brown 2004, pp. 57–58), although larvae have been detected in salinities as high as 12 ppt, likely the result of tidal dispersion of larvae around a core habitat.

2.6.3 Turbidity

The role of turbidity in providing simultaneous feeding and predator avoidance advantages for larval fishes has been well established (Utne-Palm 2002, p. 115; Pangle *et al.* 2012, pp. 10-11). The small size of most fish larvae means that the distances over which they react to prey are very short (e.g., a scale of millimeters). This implies relatively few sediment and algal particles that tend to dominate aquatic turbidity interfere with prey detection. A helpful analogy provided by Utne-Palm (2002, p. 115) is that turbidity in aquatic environments works like fog does in terrestrial environments; at short distances, it has little effect on our ability to see, but at longer distances, it can increasingly obscure objects from sight. Further, the sediment and algal particles often backlight relatively translucent zooplankton, helping larval fishes see these prey more easily (Utne-Palm 2002, p. 119). In contrast, larger fishes that may prey on fish larvae have longer search and reactive distances so more sediment and algal particles are in between these larger fish and their potential prey (Utne-Palm 2002, pp. 122–123). This can visually confound efforts by larger animals that hunt primarily using eyesight to see potential prey, making it more difficult for them to forage effectively.

We assume that these turbidity mechanisms that apply broadly to larval fishes also apply to longfin smelt in the San Francisco Estuary. In laboratory turbidity tests, larvae had higher survival at 40 NTU[1] and grew larger at 20 NTU and 40 NTU as opposed to 10 NTU (Yanagitsuru 2020, pers. comm.). In addition, larvae under the 10 NTU treatment group did not successfully transition from eating rotifers to the larger *Artemia*. Taken together, these recent laboratory experiments combined with the field results described above provide an assessment of early-life habitat needs.

2.6.4 Dietary

As reviewed in Section 2.5, during the time that longfin smelt larvae and small juveniles are feeding in low-salinity habitats, they appear to focus on only two prey taxa. Larvae less than about 2.5 cm (1 in) in length appear to primarily use the copepod *Eurytemora affinis* as prey so it is logical to assume it is a dietary need. The same is true for larvae and small juveniles over about 2.5 cm in length which appear to require mysids as prey. Longfin smelt adults that had returned to Suisun Marsh have shown a strong

[1] NTU stands for Nephelometric Turbidity Units, i.e. the unit used to measure the turbidity of a fluid or the presence of suspended particles in water. The higher the concentration of suspended solids in the water is, the higher the turbidity and corresponding NTU value is.

dietary preference for mysids while relying on copepods and amphipods when mysids are scarce (Burdi 2022, pers. comm.; CDFW unpubl. Diet Study Data; Feyrer *et al.* 2003, p. 281).

Both prey taxa have declined and their relationships with X2 were changed by the overbite clam's impact on the estuarine food web (see Table 2.1). *E. affinis* did not have a significant interannual association with X2 until after the overbite clam began rearranging the food web. It has been assumed that this loss of larval prey could have contributed to the decline of longfin smelt as well as other POD species (Sommer et al. 2007, p. 274). If that were a correct hypothesis, a decline in recruits per spawner would be expected starting in the late 1980s. However, that change was not observed in either of the two studies that have evaluated it (Maunder *et al*. 2015, p. 107; Nobriga and Rosenfield 2016, pp. 54–55). Thus, it does not appear that declining abundance of *E. affinis* has generated a change in early life stage survival large enough to be discerned with available data.

The situation is the opposite for mysids which had higher abundance in wet versus dry years before the overbite clam, but not after (see Table 2.1). This indicates that, prior to the overbite clam invasion, a prey abundance mechanism for juveniles was difficult to statistically distinguish from other recruitment mechanisms associated with years of high Delta outflow. If mysid abundance was important to longfin smelt recruitment, then the expectation following the overbite clam invasion would be a substantial decline, even after accounting for the influence of outflow (or X2). This was observed (Kimmerer 2002a, Fig. 5, p. 46 and Fig. 8, p. 48; Thomson *et al.* 2010, Fig. 6b, p. 1442), which lends support to a food limitation hypothesis. However, if the decline of *E. affinis* cannot be linked to longfin smelt decline, then the only other meaningful early life prey is mysids. Interestingly, several studies that have attempted to link time series of mysid abundance with time series of longfin smelt abundance or survival have not been able to do so. Mysid abundance was not able to predict Fall Midwater Trawl (FMWT) catches of longfin smelt with the strength of statistical inference desired by one study (Thomson *et al.* 2010, Fig. 6c, p. 1442).

The presence versus absence of overbite clam in the estuary was not retained in the longfin smelt model described by Mac Nally *et al.* (2010, Fig. 3, p. 1425) meaning that there was not a consistent change in longfin smelt abundance before and after the overbite clam invasion that couldn't be better explained by other covariates. Further, this model predicted longfin smelt had a top-down (predatory) influence on mysid abundance, rather than a bottom-up influence of food limitation. Similarly, the mysid covariate tested by Maunder *et al*. (2015, Table 2, p. 108) was not retained in any of these authors' alternative life cycle model constructs (Table 3, p. 108).

Chapter 3 – Current Condition

For the Bay-Delta DPS to maintain viability, a significant portion of its population must be resilient to the variable environmental conditions. In this chapter, we start with describing the stressors that the species has historically faced to date, and how the species has responded to these stressors over time. We then show the estimates of species relative abundance based on long-term monitoring program data to document the current condition of the species. Then, we present the results of our quantitative modeling estimating probability of extinction on the current trajectory, and close with a discussion of the 3R's relative to the ecological viability of the DPS.

3.1. Stressors

Here we discuss the most probable stressors and environmental variables that have likely led to the Bay-Delta DPS's current condition, which include the following: habitat loss and degradation via reduced freshwater flow, food web effects from reduced flows and invasive species, increasing temperatures. Additionally, we discuss other potential stressors that are less certain to be influencing abundance patterns or are thought to be influencing population dynamics to a lesser degree. These include: conversion of tidal marsh and environmental contaminants; and more so historically than presently, entrainment via water diversions. Based on our knowledge of the life history of the Bay-Delta DPS, we assess how changes in the system may have affected recruitment and subsequent survival among life stages, resulting in the observed decline in abundance through time, which is presented following this treatment of historical and extant stressors.

3.1.1. Reduced freshwater flow

A significant stressor to the Bay-Delta longfin smelt is reduced flows and alteration of the natural hydrograph—reduction in the magnitude and duration of freshwater flows into and through the Delta and to a lesser extent from the various Bay Area tributaries. In the Bay-Delta, the natural hydrologic cycles of drought and flood would yield a different outflow regime (i.e., "unimpaired flow") if upstream dams and diversions were not in existence, as water project management has decreased springtime flows resulting from upstream storage, and increased summer inflows that are subsequently diverted for urban and agricultural beneficial uses (Kimmerer 2004, p. 15).

Demand for water supplies and flood protection have substantially increased over time. In response, local, State and Federal agencies have built dams and canals, and captured water in reservoirs, to increase water storage during the wet season and convey water to farms and cities during the dry season, resulting in one of the largest man-made water systems in the world (Nichols *et al.* 1986, p. 569). Operation of this system has thus altered the timing, magnitude, and duration of freshwater flows into the Bay-Delta (Andrews *et al.* 2017, p. 72; Gross *et al.* 2018, p. 8). Storage in the upper watershed of peak runoff, and release of the captured water for irrigation and urban needs during subsequent low flow periods, resulted in a broader, flatter hydrograph with less seasonal variability (Kimmerer 2004, p. 15).

In addition to the system of dams and canals built throughout the Sacramento and San Joaquin River basins, the Bay-Delta is unique in having the largest water diversion system on the west coast. The State Water Project (SWP) and Central Valley Project (CVP) each operate water export facilities in the south Delta (Kimmerer and Nobriga 2008, p. 2). Project operation is dependent upon upstream water supply and export area demands, both of which are strongly affected by the interannual variability in precipitation within the broader hydrologic basin draining the Sacramento and upper San Joaquin Valleys. From 1956 to the 1990s, water exports increased, rising from approximately five percent of the Delta inflow to approximately 30 percent of the Delta inflow (Cloern and Jassby 2012, p. 7). By 2012, an estimated 39 percent of the estuary's unimpaired flow in total was either consumed upstream or diverted from the estuary (Cloern and Jassby 2012, p. 8).

Although alterations in the Delta and California's water diversions/storage have dated back to the latter 1800s, the most substantial parts of California's water infrastructure were constructed during the 1940s-1970s (see Section 2.3). It is noted that water year type classifications should not necessarily be conflated with realized estuarine hydrological conditions, as water operations modify these parameters based on supply considerations and extant regulatory requirements. For example, Reis *et al.* (2019, Table 5, p. 12) quantified a statistically-significant reduction in the frequency of realized delta outflow as a proportion of unimpaired flow reaching the Bay. Hutton *et al*. (2017, p. 2500) confirmed statistically-significant decreases in outflow during four months (February, April, May, and November), and increases in July and August—with follow-up analysis indicating primary attribution for these changes associated with the State and Federal water projects (Hutton *et al.* 2017b, Table 3, p. 2523). Further, the magnitude of impairment has changed over time (Hutton *et al.* 2017a, Fig. 4, pp. 2507-2508; Reis *et al*. 2019, Fig. 3, p. 12). It is probable that the magnitude (along with frequency) of realized delta outflow conditions are critical to this underlying relationship between flow variables, ecological conditions, and historical abundance trends, but only more detailed analyses and modeling will help characterize this risk quantitatively *and* mechanistically.

Numerous studies have shown the positive correlation between longfin smelt juvenile abundance and freshwater flow (Stevens and Miller 1983, pp. 431–432; Jassby *et al.* 1995, p. 285; Sommer *et al.* 2007, p. 274; Thomson *et al.* 2010, pp. 1439–1440; Kimmerer 2002a, p. 47; Rosenfield and Baxter 2007, p. 1585; Kimmerer *et al.* 2009, p. 381; Mac Nally *et al.* 2010, p. 1422; Maunder *et al.* 2015, p. 108; Nobriga and Rosenfield 2016, p. 53; Kimmerer and Gross 2022, p. 2734). Longfin smelt indices have exhibited a persistent correlation with freshwater flow. The survival of longfin smelt through their early life-stages is lower during dry conditions and higher during wet conditions--the evidence for this is that longfin smelt abundance indices nearly always decline sharply during dry periods then rebound when wet weather returns (Mahardja *et al.* 2021, pp. 9–10). In addition, multiple and consecutive dry conditions or prolonged drought are expected to be significant pinch points putting downward pressure on the Bay-Delta DPS' trajectory, and the population may no longer possess the resilience to be able to recover from a string of consecutive dry years, or years in which water operations are unfavorable during important intervals within their lifestage.

Biologically speaking, one of the most important aspects of the various estuarine circulation mechanisms is that they can help animals avoid being transported seaward even though the net flow averaged over the entire water column is flowing toward the ocean. The same mixing currents that can aggregate sinking particles can assist in aggregating motile plankton. However, swimming organisms like young longfin smelt (Bennett *et al*. 2002, p. 1502) and their prey (Kimmerer *et al.* 2002, Fig. 9, p. 365; 2014, Fig. 6, p. 911) combine changes in their vertical and lateral position with these variable mixing currents to help maintain themselves in spatial association with particular salinity ranges and other habitat attributes.

Changes in Delta outflow affect both the location and the function of the low-salinity zone. When Delta outflow is low, X2 encroaches into the legal Delta (i.e., east of Chipps Island; Jassby *et al.* 1995, Fig. 2a, p. 274). When this occurs in the winter and spring, it affects where longfin smelt spawn and their larvae and young juveniles rear. Upstream of X2, surface and bottom salinities are always about the same (Jassby *et al.* 1995, Fig. 2c, p. 274). This means that, upstream of X2, there is little if any stratification of the water column and therefore the behaviors that planktonic organisms employ to stay associated with particular salinity ranges become ineffective. As a result, landward (upstream) of X2, these animals are less likely to avoid movement with net flows because the net flow direction is uniform across the water column. This can be beneficial if they occupy water in which the net flow direction is seaward (toward the low-salinity zone), but detrimental if they occupy water in which the net flow direction is moving toward the CVP and SWP water diversions in the southern Delta (Kimmerer 2008, Fig. 16, p. 23). In contrast, fish and invertebrates that are strongly affiliated with in-water structures and vegetation, as well as benthic fish and invertebrates, can resist the net flows in the Delta. However, these are not behaviors that have been attributed to the early life stages of longfin smelt (Bennett *et al.* 2002, Fig. 4, p. 1502; Grimaldo *et al.* 2020, Fig. 6, p. 10).

In terms of flow changes within the legal Delta, the CVP and SWP have a more outsized influence on hydrodynamics than non-project diverters (Hutton *et al.* 2019, Fig. 7, p. 11). Water exports from the Delta increased from the 1950s into the late 1980s (Cloern and Jassby 2012, Fig 7a, p. 7; Service 2019, Fig 5.7). Thereafter, the increasing trend began to level off, but the year-to-year variability increased. The export of water from the southern Delta tends to draw a 'net' flow of water from north to south (Kimmerer and Nobriga 2008, Fig. 7, p. 12). This flow is referred to as 'net' because it expresses a tidally-filtered flow, meaning that the tidal flows that routinely cycle negative (toward the SWP and CVP diversions) and positive (seaward) over hourly time scales have been mathematically removed to characterize the portion influenced by exports as net flow direction. Nonetheless, the transport of water toward the SWP and CVP diversions is fundamentally an instantaneous to tidal time-scale process.
The CVP pumping plant diverts water off of Old River during all phases of the tide. In contrast, the SWP includes Clifton Court Forebay (CCF) in front of its diversions. The CCF is a gated, regulating reservoir that helps manage the highly variable water levels in Old River for the CVP pumps and local irrigators. During flood tides, the CCF intake gates are opened to allow the tide (and gravity) to bring water into the forebay. The water that is being exported at the CVP, and the water flowing into CCF, are backfilled by the tide and gravity flow of water from elsewhere in the Delta. Over time scales of weeks to months, this backfilling, which is the hydraulic mechanism for the net flow, dominates the transport of river water in the Delta (Kimmerer and Nobriga 2008, Fig 3, p. 10).

It has long been recognized that abundance indices for age-0 longfin smelt tend to increase in wet years that have followed dry years (Stevens and Miller 1983, Table 8, p. 433; Jassby *et al.* 1995, Fig. 5, p. 280; Rosenfield and Baxter 2007, Table 3, p. 1585); but longfin smelt abundance indices have also declined over time in a range of water year types (Thomson *et al.* 2010, Fig. 6, p. 1442). Freshwater flows entering the estuary change a wide variety of physical, chemical and biological parameters that can influence the distribution and abundance of aquatic species (see Section 2.3). Thus, it is unlikely that a single mechanism associated with or co-varying with wet and dry years is responsible for generating observed patterns in longfin smelt recruitment. Recent analyses that have attempted to consider the influence of wet and dry years in the context of longfin smelt's life cycle have found evidence that the freshwater flow-, or flow-associated mechanisms, influence the number of juvenile recruits produced per spawner (Maunder *et al*. 2015, pp. 105–106; Nobriga and Rosenfield 2016, Fig. 2, p. 53).

Population dynamics of longfin smelt surviving a sufficient duration into the juvenile life stage to begin migrating seaward do not appear to be influenced by freshwater flow variation (Nobriga and Rosenfield 2016, p. 55). These two pieces of evidence imply that the reproductive success of the adult spawners, the survival of eggs and early life stages, or both of these factors, are the endpoints being affected by the freshwater flow- or flow-associated mechanisms. Figure 3.1 shows hypothesized, staggered, but overlapping, timing of flow- or flow-associated mechanisms that might have an important quantitative influence on longfin smelt spawning success, on egg survival, or survival from the larval to the juvenile life stage. Each of these mechanisms is known to covary with flow, or to have done so historically (citations provided below). It is not the existence of these mechanisms, but their *quantitative influence* on the viable egg production and survival of longfin smelt's early life stages that remains a clear science priority in support of longfin smelt conservation.

While the overall pattern relating freshwater flows to abundance indices for the Bay-Delta DPS is widely accepted, the mechanisms driving this correlation are not fully quantified or resolved. Following are the enumerated summaries of mechanistic drivers of Bay-Delta DPS abundance dynamics as related to freshwater flow, or flow-associated physical or biological parameters.

Mechanism 1, spawning locations: High flows during the spawning season likely led to more total spawning effort seaward of the legal Delta (Grimaldo *et al*. 2020, Fig. 6, p. 10) and higher spawning success in Bay Area tributaries (Lewis *et al*. 2019, p. 3). If egg survival tends to be higher seaward of the Delta, then variation in spawning locations may be a recruitment mechanism that covaries with freshwater flow. This is speculation that warrants focused research efforts to better inform longfin smelt conservation.

Mechanism 2, spawning season duration: The duration of the longfin smelt spawning season each year is fundamentally driven by climatic influences on the estuary's water temperatures. Wet weather increases tributary outflows from Bay Area streams where overlying air temperatures remain cool longer into the season than is typical in the Delta. This might lead to longer spawning seasons in these smaller systems. Wet weather also increases Delta outflow, which likely leads to more total spawning effort seaward of the legal Delta (Grimaldo *et al.* 2020, Fig. 6, p. 10). If elevated outflow is maintained into the spring, one expected result would be successful spawning

opportunities not available in warmer, lower outflow years when the low-salinity zone is encroaching on the Delta and overlying air temperatures during May typically begin to exceed Bay Area air temperatures (Conomos *et al*. 1985, Fig. 4, p. 5).

Mechanism 3, low-salinity zone location and retention for larvae and young juveniles: The hydrodynamic phenomena reviewed in section 2.3 may create bioenergetically-favorable transport and retention opportunities for larval and juvenile longfin smelt. For instance, the Delta does not have as much shoal habitat as seaward parts of the estuary. When outflow is high enough to place the low-salinity zone over larger shoal and marsh habitats, the lower water velocities and estuarine mixing currents may enhance opportunities for early life stage longfin smelt to maintain position and find food in higher quality habitat (Hobbs *et al.* 2006, p. 916; Grimaldo *et al.* 2017, p. 7).

Mechanism 4, entrainment of larvae and young juveniles: High flows during the spawning season likely lead to more total spawning effort seaward of the legal Delta (Grimaldo *et al.* 2020, Fig. 6, p. 10). This leads to smaller fractions of larvae vulnerable to entrainment in exported water. Higher net flows out of the Delta also better facilitate the seaward transport of larvae that are spawned in the Delta and lessen their risk of entrainment in water diversions (Kimmerer and Nobriga 2008, Fig 7, p. 12; Rosenfield 2010, Fig. 9, p. 40; Grimaldo *et al.* 2009, Table 2, p. 1261).

Mechanism 5, prey availability for larvae (*Eurytemora affinis*): If food limits the survival of longfin smelt during their larval stage, it would logically be associated with declines in *E. affinis*—the predominant component of larval longfin smelt diet (Barros *et al*. 2022, Fig. 6a, p. 1775). The influence of Delta outflow on *E. affinis* production has changed over time, due in large part to the strong influence of overbite clam grazing (Table 2.1). The abundance of *E. affinis* has declined over time (Winder and Jassby 2011, Fig. 6, p. 682), but as of 2011, longfin smelt recruits per spawner had not (Nobriga and Rosenfield 2016, Fig. 2, p. 53). This dissociation of trends makes it unlikely that the decline of *E. affinis* had a major quantitative impact on the survival of larval longfin smelt (Nobriga and Rosenfield 2016, pp. 54–55), but a comparable analysis for the last decade is lacking.

Mechanism 6, prey availability for young juveniles (mysids): If food limits the survival of longfin smelt exceeding about an inch in length, it would logically be associated with mysids—the predominant component of the diet at this lifestage (Barros *et al*. 2022, Fig 6b, p. 1775). If mysid abundance was important to longfin smelt recruitment, then the expectation following the overbite clam invasion would be a substantial decline, even after accounting for the influence of outflow (or X2). This is what was observed (Kimmerer 2002a, Fig. 5, p. 46 and Fig. 8, p. 48; Thomson *et al*. 2010, Fig. 6b, p. 1442), which lends support to a juvenile longfin smelt food limitation hypothesis. We recognize that direct statistical support for a bottom-up longfin smelt-mysid link has not been reported in several recent statistical evaluations (Mac Nally *et al*. 2010, Fig. 3, p. 1425; Thomson *et al*. 2010, Fig. 6c, p. 1442; Maunder *et al.* 2015, Tables 2-3, p. 108). However, longfin smelt may continue to rely on mysids as prey until they leave the estuary, and the survival of these older fish may have declined over time (Nobriga and Rosenfield 2016, Table 5, p. 54). Therefore, it remains possible that changes in mysid abundance have been an important

contributor to the observed decline. This is speculation that warrants focused research efforts to better inform longfin smelt conservation.

Mechanism 7, prey delivery from the Yolo Bypass: As outflow from the Delta increases, the proportion of the total flow entering from the Yolo Bypass increases as well (Kimmerer 2004, Fig. 5, p. 16). As the bypass drains, its relatively shallow depth, long residence times, and warmer temperatures can help deliver phytoplankton and invertebrates into the Delta (Sommer *et al*. 2004, Fig. 3 p. 254). However, in the wetter years when Yolo Bypass floods and drains, many, if not most, longfin smelt are rearing in fairly distant locations. Therefore, it is uncertain whether these winter-spring floodplain food subsidies are substantial contributions to larval longfin smelt at the population scale.

Mechanism 8, turbidity of the low-salinity zone: The potential importance of turbidity as a moderator of feeding success and predation risk for young longfin smelt was reviewed in section 2.6.3. Turbidity has declined in the estuary for a combination of reasons. However, the high winds that accompany wet weather can resuspend sediment and increase low-salinity zone turbidity, especially when the low-salinity zone is flowing back and forth over major shoal areas like Suisun Bay (Bever *et al*. 2018, Fig. 11, p. 1957). Further, high winter-spring flows deliver new sediment and particulate matter that can increase turbidity (Kimmerer 2004, Fig. 27, p. 41; Cloern and Jassby 2012, Fig. 10, p. 10).

In summary, during high outflow years, longfin smelt are believed to benefit from a suite of mechanisms that can extend the spawning season and increase the cumulative survival of the early life stages. Conversely, during low outflow years, fewer of these benefits are accrued and survival is reduced. For these reasons, the interannual variation in Delta outflow and to lesser extent, flows in Bay Area tributaries (which can also be represented by correlates like X2) mechanistically represent a primary population need from December through May or June each year.

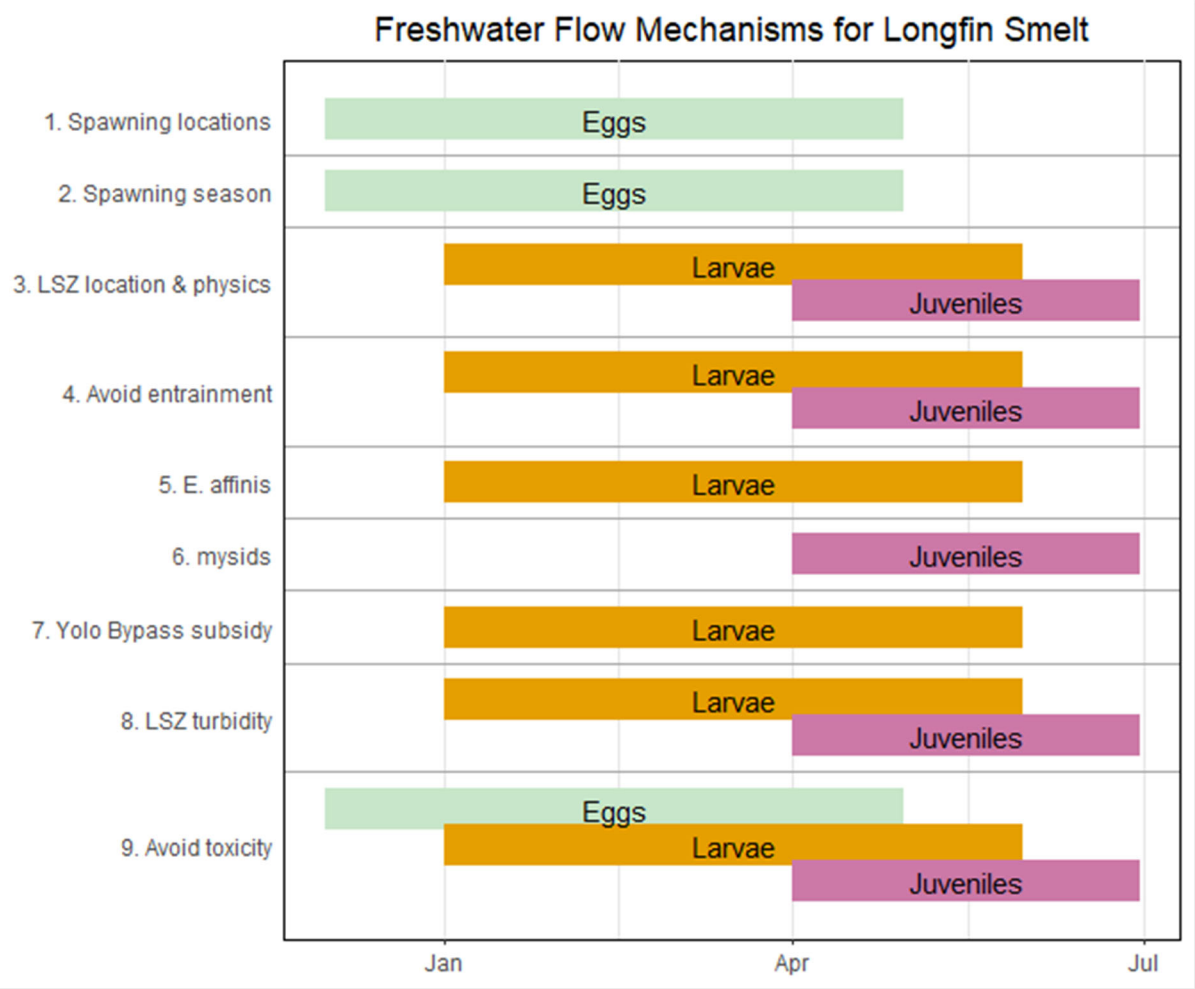


Figure 3.1: Conceptual timeline representing the multiple potential mechanisms by which interannual variation in Delta outflow may mechanistically contribute to recruitment of Age-0 longfin smelt. The life stage durations are approximate in that they start and end on the first and last day of particular months. The reader should recognize that exact stage durations will vary from year to year.

The extended durations of elevated freshwater flow out of the Delta and other estuary tributaries during wetter water years increases the number of beneficial environmental conditions that spawning longfin smelt and their progeny experience. Contrasting this to the pattern in dry years, as X2 and, by extension, the low-salinity zone, moves upstream, longfin smelt migrate farther upstream to reach their spawning habitats (CDFG 2009a, p. 17). Spawning further upstream in the Bay-Delta makes longfin smelt more susceptible to entrainment at the State and Federal water diversions in the south Delta, particularly during late-winter and early spring when larvae are present (Rosenfield 2010, p 13). Studies of hydrodynamics and fish entrainment have found that predicted losses increased with reverse flows in the southern Delta, which are a function of export rates and Delta inflows (Grimaldo *et al.* 2009, p. 1266; Rosenfield 2010, p. 13). In addition, as X2 moves upstream (landward) in the estuary, likely consequences are less spawning habitat available, weakening or absent gravitational circulation (which may have energetic, physiological, and predator avoidance consequences for young longfin smelt) and

decreased water turbidity (which also has survival implications for young longfin smelt based on their role as both predator and prey) (Kimmerer 2002b, p. 1279).

Finally, as the spawning range of the species is significantly concentrated in the west Delta and Suisun Bay during dry years instead of the entire San Francisco Estuary, the Bay-Delta DPS is especially vulnerable to local catastrophic events. For example, if a large oil spill occurred in the Delta during a dry year, the entire spawning class and its progeny could be wiped out. But during wet years where spawning can occur in San Pablo Bay, the South San Francisco Bay, and their tributaries (during exceptionally wet years), a large oil spill in the Delta may not affect spawning occurring in San Pablo Bay and the South San Francisco Bay. Since 1971, large petroleum spills that have demonstrable impacts in the San Francisco Estuary have occurred approximately once every 8 years (CDFW 2021b).

3.1.2. Food limitation

The available information indicates that in the Bay-Delta DPS rely on a relatively small number of crustacean meso- and macrozooplankton taxa. Longfin smelt larvae have diets dominated by a copepod, *Eurytemora affinis,* that is common in the low-salinity zone during the spring (CDFW, unpublished data; see Figure 3.2A) indicates that *Limnoithona tetraspina* are more dominant in January and February larval diets, while other inverts are important at times in the Napa River and San Pablo Bay (Bouley and Kimmerer 2006, p. 221). Like most carnivorous fishes, as longfin smelt larvae increase in body size and mouth gape, they begin taking larger prey that provide more calories per unit of effort expended catching them (i.e., optimal foraging theory). The two most common prey taxa of these larger longfin smelt are epibenthic mysids and amphipods (Burdi 2022, pers. comm.; CDFW unpubl. Diet Study Data). The copepod *Eurytemora affinis* was also at one time an important prey item for a now much depleted mysid species, *Neomysis mercedis* (Knutson and Orsi 1983, p. 478).

The *Eurytemora affinis* population has been in decline since the 1970s, but beginning in the late 1980s, the zooplankton community started undergoing about a decade of rapid change in species composition, trophic structure, and utility for fish production (Winder and Jassby 2011, pp. 683–685; Kratina *et al.* 2014, p. 1070; Brown *et al.* 2016a, p. 8). As these food web changes were beginning to take place, the production of longfin smelt per unit of outflow or X2 began to decline (Kimmerer 2002a, p. 47; Thomson *et al.* 2010, p. 1442c) coincidental with the invasion of the estuary by the overbite clam (*Potamocorbula amurensis*) (Carlton *et al.* 1990, p. 81) and with extended drought in the Central Valley (Rosenfield and Baxter 2007, p. 1589). This decline in the abundance indices was sharp and substantial (generally recognized as a “step decline”). The decline of longfin smelt’s historical prey has not been accompanied by a large change in prey use (Feyrer *et al.* 2003, p. 285; Barros *et al.* 2019, p. 15). This suggests that longfin smelt had formed strong predator-prey interactions with their primary prey, a hypothesis supported by empirical data (Mac Nally *et al.* 2010, p. 1426).

As described under Life History, approximately 90% of juvenile and, when they return to the estuary, adult longfin smelt diets are comprised of predominantly mysids and, to a lesser extent, amphipods (Burdi 2022, pers. comm.; CDFW unpubl. Diet Study Data). *Neomysis mercedis*, which was once a

dominant contributor to the low-salinity zone food web, has dropped in numbers by over tenfold and accounted for <4% of total zooplankton biomass after 1994 (Winder and Jassby 2011, p. 684). In addition to lower abundance, CDFW's Zooplankton Study has shown that the average individual sizes of mysids in the San Francisco Estuary have decreased over time (Hennessey 2011, unpublished data), with a species composition shift towards *Hyperacanthomysis longirostris*, an invasive species that reaches maturity at a smaller mass than *Neomysis*. Despite these changes, longfin smelt are one of the few species that continue to rely heavily on *Neomysis* and other mysids as their primary prey (see Figure 3.2B). As longfin smelt exhibit very little variation in prey use, they are considered more susceptible to food web changes than some other fishes (Feyrer *et al.* 2003, p. 281).

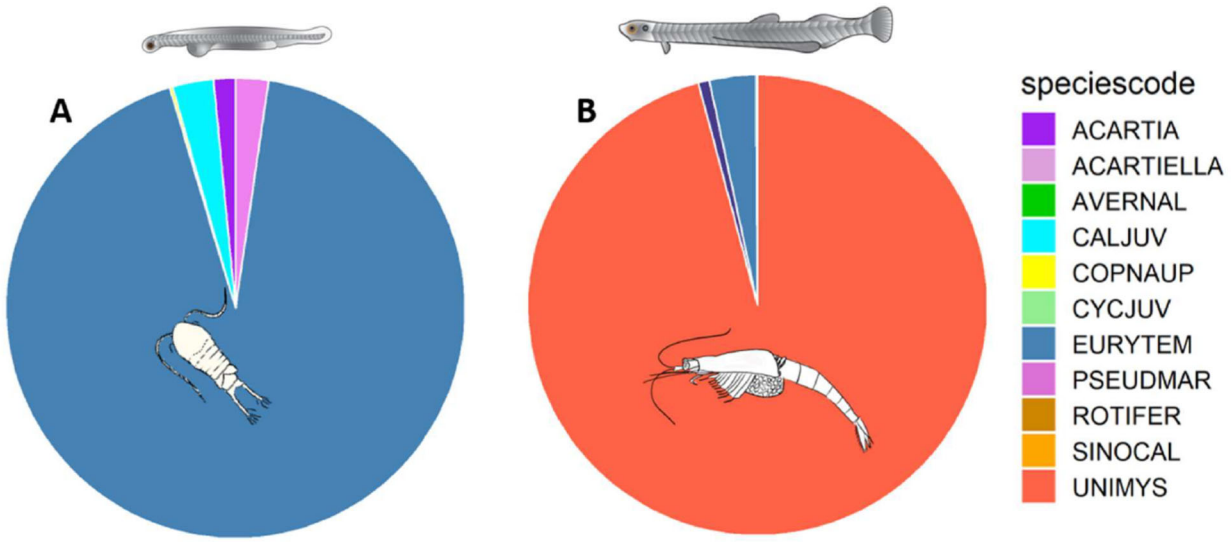


Figure 3.2: Proportion of diet biomass for all longfin smelt broken up between A) larval (<25mm TL) and B) juvenile fish (>25mm TL). Copepod and mysid art by Arthur Barros. Larval and juvenile fish art by Adi Khen. Source: Barros *et al.* 2019, p. 17.

The general decline in diatom phytoplankton and zooplankton is likely affecting juvenile and adult longfin smelt by decreasing food supply for historically important prey species, such as *Neomysis mercedis* (Kimmerer and Orsi 1996, pp. 418–419; Feyrer *et al.* 2003 p. 281). In addition to phytoplankton suppression, laboratory and field experiments have shown that overbite clams feed directly on the nauplii of common calanoid copepods (Kimmerer *et al.* 1994, p. 87), which are the primary prey source for larval longfin smelt.

The Pelagic Organism Decline (POD)

Around 2002-2004, abundance indices for multiple species declined in what was known at the time as the Pelagic Organism Decline or POD. The POD referred to a coincident drop in the FMWT catches of four species (delta smelt, longfin smelt, striped bass, and threadfin shad) (Sommer *et al.* 2007, p. 273), and the event is generally recognized as another step decline. The coincident declines of multiple

species suggested a possible common cause, but a mechanism for decline that applied to all four fish was not forthcoming (Mac Nally *et al.* 2010, p. 1426; Thomson *et al.* 2010, pp. 1442–1443). Latour (2016, Fig. 7, p. 244 and p. 245) later hypothesized that changes to fish catchability due to increasing estuary water clarity might be an explanation for the POD.

Fish population dynamics theory is important to interpreting analyses of longfin smelt data from the literature because most of the trend analyses cited above have ignored it. The Bay-Delta DPS has plausibly been declining for over 50 years and that decline is presently at circa 3-4 orders of magnitude below initial observations. The FMWT index is the most commonly used metric of longfin smelt recruitment in the scientific literature. It is generally an index of age-0 fish but the adult longfin smelt population is also declining (Rosenfield and Baxter 2007, Table 3, p. 1585; Nobriga and Rosenfield 2016, Table 2, p. 49, which in turn is limiting how many eggs can be produced. The reason the longfin smelt population keeps appearing to have a 'step decline' or 'change in intercept' of its outflow relationship is at least in part because analyses that do not account for the declining abundances of the parental generations are based on an improper population model that ignores the influence of adult egg supply on how many recruits can be produced (see Figure 3.3).

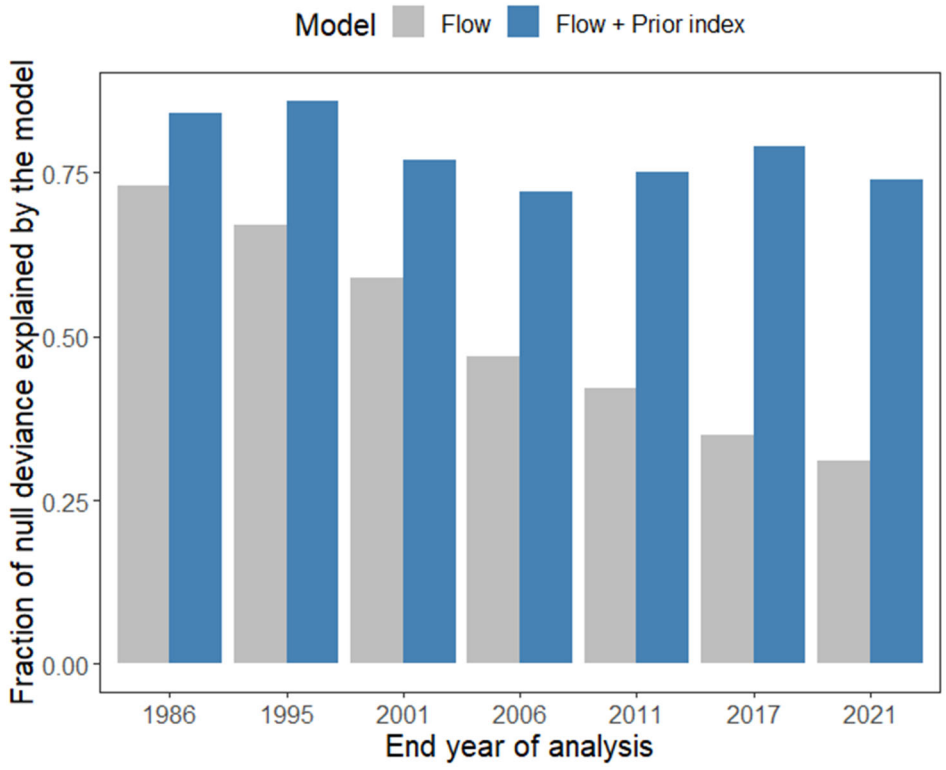


Figure 3.3. Summary of linear regression models predicting the longfin smelt Fall Midwater Trawl (FMWT) indices over time, 1967 through the year listed along the x-axis, i.e., moving from left to right along the x-axis, the length of the data time series is increasing. The gray bars show the fraction of null deviance (akin to an r-squared) explained by generalized linear models of log(Delta outflow) averaged for February-June versus log(FMWT index). The ability of this model to explain trends in the indices has

declined substantially over time (see also Tamburello *et al.* 2018). The blue bars depict the explanatory power of models that also include the FMWT index from two years prior as a predictor of the current year index (following Mac Nally *et al*. 2010). This latter model is at least coarsely accounting for the potential of declining adult abundance to limit egg supply which in turn limits how many juveniles can be produced. The predictive capability of this model formulation has not trended downward like the flow-only models have.

3.1.3. Temperature

As described in the Life History and Biology section, longfin smelt are most abundantly detected within a narrow temperature range relative to the range that occurs in the upper estuary. Subadult longfin smelt habitation in the San Francisco Estuary is limited to when water temperatures are below 22 °C (Baxter *et al.* 2010, p. 68), and, based on field surveys of ripe and post-spawning females (Wang *et al.* 1986, p. 9; Tempel and Burns 2021, slide 12), successful spawning may require water temperatures below 14 °C, while larvae and young juveniles show a preference for temperatures below 12 °C and 20 °C, respectively, for successful rearing--particularly in food-limiting environments like the San Francisco Estuary, where bioenergetic metabolic demands for caloric intake increase with increasing water temperatures. As the southern-most population, the Bay-Delta DPS likely already experiences the warmest temperatures in the full species range. Water temperature in the Delta commonly exceeds 22 °C during the summer (Vroom *et al.* 2017, p. 9904; data from CDEC, CENCOOS, and USGS), and has been predicted to exceed this threshold for longer periods due to climate change, which would limit habitat occupancy across affected areas, and stress life-stages with limited adaptive motility (i.e., in early spring for young juveniles). It remains unknown to what extent the ocean acts as a thermal refugium for the DPS; however, recent otolith data has shown that adults do exhibit anadromy that had been hypothesized previously by Rosenfield and Baxter (2007, p. 1590). Bay Study Otter trawl age-0 captures in July exhibited mean salinity conditions of 24 ppt, so some juveniles in July (perhaps sooner) can withstand marine salinities (Baxter, 1999, p. 189). Summaries of City of San Francisco outfalls catches also confirm age-0 and age-1 individuals present in the Gulf of Farallones (CDFG, 2009, page 6). The potential effects of climate change are described in detail under the Future Condition chapter.

3.1.4. Loss of suitable spawning habitat

The only fairly well demonstrated aspect of longfin smelt spawning behavior is that the fish appear to find spawning locations in and near the low-salinity zone (Grimaldo *et al*. 2020, Fig. 6, p. 10) and other smaller low salinity habitats in Bay Area tributaries (Lewis *et al*. 2019, p. 3). Just-hatched longfin smelt larvae have been found in pelagic waters and adjacent shoals and wetland areas (Grimaldo *et al*. 2017, p. 1776; Lewis *et al.* 2019, Fig. 2, p. 3). Thus, what other attributes of spawning habitat selection are important to successful reproduction are unknown and would benefit from additional research in support of longfin smelt conservation.

3.1.5. Predation

There is no research from the San Francisco Estuary available from which to develop a complete list of important predators of longfin smelt or to estimate predation rates on longfin smelt to determine how

they have trended over time. There have been two recent predator diet studies spatially limited to the legal Delta that tested for prey using DNA extracted from predator stomachs. Michel *et al.* (2018, p. 9) sampled the San Joaquin River between Head of Old River and Stockton, and they did not detect any longfin smelt in any of four species of predators' stomachs. Brandl *et al.* (2021) collected six species of predators from the North Delta. The predator with the most longfin smelt detections was Sacramento pikeminnow (Brandl *et al.* 2021, Table 4). The only other predator from which longfin smelt DNA was detected was striped bass. No longfin smelt DNA was detected from channel catfish, white catfish, largemouth bass or smallmouth bass. Because of this paucity of data, the following discussion is limited to generalities about the role of predators in fish food webs that are broadly applicable and form the basis of major, widely used modeling tools.

The early life stages of fish are often subject to high rates of predation that play important roles in modulating abundance and amplifying the consequences of food limitation (Ahrens *et al.* 2012, Fig. 2, p. 46, and throughout; Pangle *et al.* 2012, pp. 5–6). Thus, changes in vulnerability to predation of eggs, larvae, and small juvenile longfin smelt are a plausible hypothesis for why survival is higher in wetter years than drier years. If predation rates covary with the freshwater flow influence on longfin smelt recruits produced per spawner, they are likely modulated through several other mechanisms like turbidity, temperature, access to zooplankton prey, or outcomes of differences in wet versus dry year hydrodynamics (Fig. 2.8).

Chronic food limitation and predation risk are often tightly linked in fish food webs (Ahrens *et al*. 2012, pp. 47–48). The reason is that prey organisms do their best to limit potential contact with predators. One key way they do this is by limiting foraging times, which are often relatively risky because small fishes have to behave in ways that increase their vulnerability to predators when they are actively foraging, e.g., by leaving shelter habitats or simply moving around more actively in surface waters (Ahrens *et al*. 2012, Fig. 1, p. 43). Thus, when food densities decline, prey fishes have two choices. They can either eat less and grow more slowly or they can increase foraging times to compensate for the lower prey densities. Depending on the context of the ecosystem a fish finds itself in, there are pros and cons to either of these choices. How longfin smelt have behaviorally responded to declining prey density has not been studied and would be valuable information from a conservation perspective.

3.1.6. Contaminants

Due to the extensive range of the Bay-Delta DPS, there is potential for exposure to a variety of contaminants. Potential exposure and effects will likely vary based on location and habitat characteristics, which change drastically from fresh to salt water, as well as differing anthropogenic sources (e.g., agricultural practices, industrial and urban wastewater discharge). The Delta is largely an agricultural area with a multitude of tributaries conveying agricultural and urban discharge. The Suisun Bay watershed consists of a mixture of agriculture, managed duck ponds, urbanization, and industry. The San Francisco Bay is encircled by largely urbanized and industrial land uses, with some significant inputs from agriculture in the San Pablo Bay.

Contaminant sources include discharge from municipal wastewater treatment plants, agricultural outfalls, stormwater runoff, and direct application such as for pest control or anti-fouling. The types of contaminants can change throughout the year with changes in agricultural use and pest control efforts

and storm loading. “Legacy” contaminants in the Bay–Delta, those from historic loading that persist in the environment, such as organochlorine chemicals from past agricultural use and mercury from past mining activity, can bioconcentrate through the food web, posing additional health risks. Human population growth has increased the urban footprint over time but there is still a significant presence of agriculture adjacent to Suisun Bay and the Delta. Regulation has reduced the use of some contaminants, only to be replaced by other more potent alternative water-soluble chemicals. In addition, there are insufficient data to know the full exposure risk of most constituents (Johnson *et al.* 2010, p. viii). Kuivila and Hladik (2008, p. 14) found that of the 163 pesticides being applied, over half were not being monitored.

The Delta and all regions of the Bay have been listed in the 2018 California Integrated Report (Clean Water Act Section 303(d) List and 305(b) Report) as an impaired water for several contaminant compounds (SWRCB June 2018, Appendix A). This list included: elemental contaminants, or ‘metals’ (mercury and selenium), toxic organics (dioxins, furans, polychlorinated biphenyls), and pesticides (chlordane dieldrin, DDT). Additional emerging contaminants of concern include: newer pesticides, flame retardants, nutrients, naturally occurring toxins, micro-plastics (e.g., from synthetic clothing), and pharmaceuticals and personal care products. The U.S. Geological Survey’s (USGS) on-going analysis of water in the Delta suggest that on average 10 new synthetic organic pesticides chemicals are detected every year (CDPR 2020, dataset). Moschet *et al.* (2017, p. 1558) have indicated the presence of in excess of 50 chemical compounds from a single, 1L, grab sample.

Contaminants can have a variety of direct and indirect impacts on fish and their supporting food web. Direct effects involve impacts to growth, vitality, immunocompetence and disease condition, behavior, reproductive impairment, mutagenicity and direct acute lethality. It is also possible that indirect effects from certain contaminants may manifest via changes in the estuarine community, that consequently impair growth, survival or reproduction of the consumer species.

To date, there are no data documenting the impact of contaminants to longfin smelt in the SFE. Field-based toxicity is difficult to determine, as impacted fish are not recovered in order to be examined (i.e., fish either die from direct exposure and resulting disease, or are eaten). Risk of exposure and effect, as determined by comparison to other species (e.g. delta smelt and inland silverside) potentially include: direct effects on development, growth and reproduction; impacts resulting from impairments to bioenergetic demands, impaired locomotion, reducing feeding success and leading to increased susceptibility to predation, disease, and entrainment (Brander *et al.* 2012, p. 2854; 2016; Connon *et al.* 2009, p. 12; 2011, p. 299; Hasenbein *et al.* 2014, p. 696; Jeffries *et al.* 2015a, p. 17407; Jeffries *et al.* 2015b, p. 55; Cole *et al.* 2016, p. 219; DeCourten and Brander 2017, p. 2). Fong *et al.* (2016, pp. 20–21) suggested that a weight of evidence approach indicates numerous detected contaminants in the Bay-Delta have detrimentally affected the ecosystem, and these authors concluded that contaminants likely played a significant role in the POD.

3.1.7. Entrainment

When water is diverted from the estuary, the opportunity is created for fish to follow the flow of water and become "entrained" by the hydrodynamic footprint of those diversions. There are several sources of entrainment for longfin smelt. These are discussed here in turn.

In-Delta Agricultural Diversions: Water is diverted at numerous sites throughout the Delta for irrigation of crops, particularly during the summer months (Siegfried *et al.* 2014, Figs. 10-11, p. 11) when most longfin smelt are rearing seaward of the legal Delta. Herren and Kawasaki (2001, p. 347) reported over 2,200 such water diversions within the Delta, but CDFG (2009, p. 25) notes that number may be high because Herren and Kawasaki (2001) did not distinguish intake siphons and pumps from discharge pipes. Given the temporal mismatch between seasonal peaks in water diversions to supply farms in the Delta and use of the Delta waterways by longfin smelt, seasonal irrigation of Delta farms does not seem like a major conservation concern for longfin smelt.

Other Diversions: The Barker Slough pumping plant is located in the north Delta and serves as the inlet to the North Bay Aqueduct, which is a municipal diversion that serves Solano County. This diversion is owned and operated by the California Department of Water Resources (DWR) and has positive barrier fish screens that likely limit entrainment of longfin smelt spawned in the vicinity of the diversion.

Downstream in Suisun Marsh, the Roaring River and Morrow Island Distribution Systems (RRDS and MIDS) are additional DWR facilities that divert water from Montezuma and Goodyear sloughs, respectively. The water is distributed to waterfowl management wetlands in Suisun Marsh and eventually returned to the marsh channels (minus what evaporates). The RRDS is screened while MIDS is not. Both diversions have been observed to entrain or impinge longfin smelt (CDFG 2009b, p. 41). Longfin smelt catches in Suisun Marsh declined earlier in time than they did in other surveys (Rosenfield and Baxter 2007, p. 1589). These authors recommended to evaluate that observation more carefully. The RRDS and MIDS came online in 1979-1980. A key question is whether longfin smelt decline in Suisun Marsh was already underway due to land and water management practices, or might the distribution systems have affected hydrodynamic conditions in the marsh in a way that increased entrainment of longfin smelt or otherwise inhibited their use of the marsh as a spawning habitat?
Entrainment into several power plants may have been an historically important source of longfin smelt mortality (particularly larvae), but these plants have since been decommissioned.

State and Federal Water Export Facilities: The CVP and SWP each include pumping plants in the south Delta. These pumping plants are used to export water to users throughout much of the State. The operation of these facilities can exert a strong influence on regional hydrodynamics (Kimmerer and Nobriga 2008, Fig. 7, p. 12; Hutton *et al.* 2019, Fig. 7, p.11). That hydrodynamic influence can result in the entrainment of fish, sometimes from considerable distances (e.g., Kimmerer 2008, p. 2, Fig. 1, p. 3). In most years, longfin smelt have been collected ("salvaged") in the fish facilities that sit in front of each pumping plant. The salvage of fish is an indicator that individuals are being entrained in exported water.

Historically, the salvage of age-1 and older longfin smelt peaked in January (Grimaldo *et al.* 2009, Fig. 5, p. 1262). These fish likely represented individuals searching for spawning habitats and immature

individuals comingling with adults. The salvage of age-0 fish peaked in April-May as larvae reached sizes at which they could be retained on the fish screens of the CVP and SWP fish facilities. In all likelihood, however, some larvae began to be entrained once they started hatching in December or January, but remained undetected until about March, with salvage efficiency increasing in April-May as the fish grew larger.

Age-0 longfin smelt salvage has historically been a function of larval abundance and Old and Middle River (OMR) flow (Grimaldo *et al.* 2009, Table 2, p. 1260). This pattern reflects two things: (1) the fish facilities are sampling devices so like other fishing gear, they can catch more fish when there are more to catch, and (2) OMR flow indexes the 'sampling effort' of the fish facilities by reflecting how much and how quickly water is being moved toward the pumping plants. When net OMR flow is positive, San Joaquin River water is generally moving seaward through the Delta. The more net negative OMR is flowing, the more water in the Delta is moving toward the pumping plants, the more of that water has come from the Sacramento River, and the faster that water is moving south.

OMR is a metric that represents a net direction of water flow, meaning that the tidal flow that moves in the positive and negative directions over hourly to spring-neap time scales has been mathematically removed. Nonetheless, the water transport processes OMR represents are instantaneous to tidal time scale variations in gravity flow of water interacting with river and tidal currents via spatial differences in water surface elevation (Andrews *et al*. 2016, equation 5, p. 6).

Longfin smelt can be entrained in exported water when adults and comingling age-1 individuals enter the south Delta, and as larvae and small juveniles that are either rearing or being tidally dispersed landward of X2 (CDFW 2020, Fig. 13, p. 53). During periods of high Delta outflow adult longfin smelt and their progeny are much less likely to be entrained in exported water because the low salinity zone is not encroaching on the Delta and more individuals may be cued to spawn in Bay Area tributaries if they are likewise flowing high.

It is possible that in the past entrainment of larval longfin smelt may have reached levels of concern (e.g., 2002 per CDFW 2020, Fig. 10, p. 47; see Figure 3.4). However, since 2009, the entrainment of longfin smelt has not been substantial enough to affect the species population dynamics. The results of two different analytical approaches to the Smelt Larval Survey (SLS) data suggest that it is not likely population-level entrainment of larvae has exceeded 3% since 2009 (Kimmerer and Gross 2022, Table 1, p. 2736; Gross et al. 2022, p. 187). Gross et al. (2022, entire) coupled particle tracking modeling with the SLS data set and found an upper 95% credible interval of proportional entrainment was 2.9% in the critically dry winter of 2013 and nearly zero in the wet winter of 2017. A second analysis similar in approach to Kimmerer (2008, entire) analyzed all of the SLS data from 2009-2020. Similarly, this approach also found proportional entrainment was unlikely to have exceeded 3% (range = 0.3% to 2.9%) (Kimmerer and Gross 2022, Table 1, p. 2736). We interpret these findings to indicate that the OMR management strategies in place since 2009 have been an effective conservation strategy for longfin smelt.

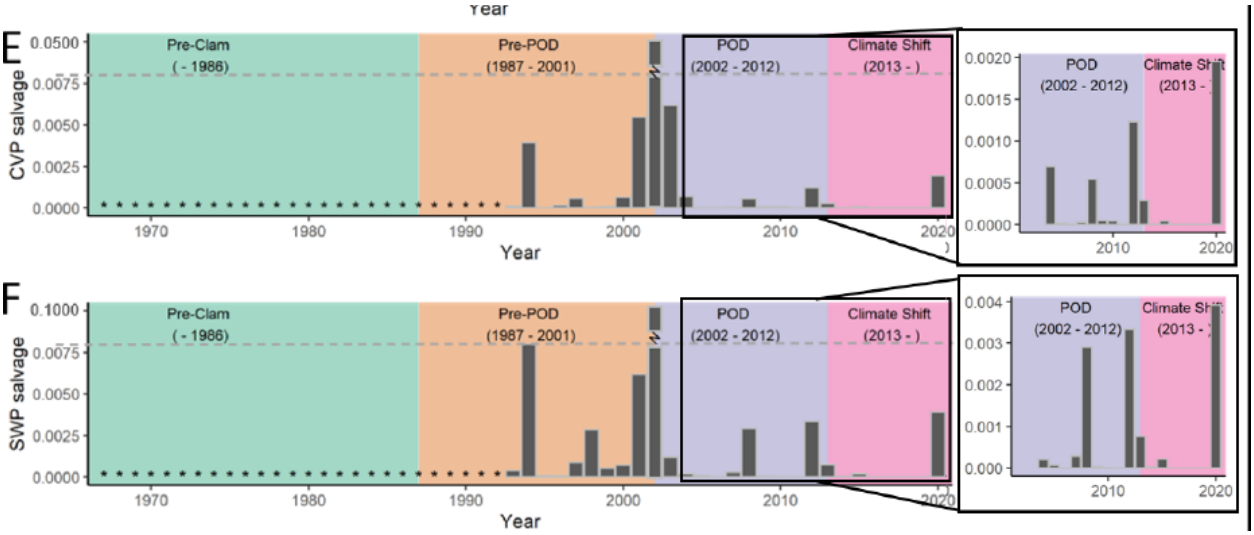


Figure 3.4: Longfin smelt salvage numbers at the SWP and CVP facilities, before and since OMR management by CDFW.

3.2. Current DPS Survey Indices

In the previous section, we detailed the stressors acting on the species. Here we summarize abundance indices showing the DPS' decline over recorded time to its current condition as a result of these stressors. These data begin with simple plots of relative abundance over time, but also include more sophisticated population modeling and viability metrics (summarized herein, and fully documented in a series of technical appendices).

3.2.1 Species relative abundance

Collectively, available survey data indicate a marked, and significant decline for the SF Bay-Delta DPS of longfin smelt throughout the estuary, and across all life stages. To show how the species' status over time to its current condition, we first utilize field survey data from three established surveys--the 20-mm Survey, Bay Study, and FMWT. As described earlier, each of these surveys routinely catches longfin smelt. SLS, although designed for longfin smelt, is not used here as it was designed to assess entrainment risk and not relative abundance, and it has a much shorter survey history (2009-present). Further information about specific surveys including purpose, timing, and geographic coverage are enumerated in Technical Appendix A, following.

The 20-mm Survey (https://wildlife.ca.gov/Conservation/Delta/20mm-Survey) has been conducted since 1995. The CDFW does not produce an index of longfin smelt abundance from the 20-mm Survey, so we adapted the methods for the delta smelt index for longfin smelt. The code and method used to produce the 20-mm index values can be found in Technical Appendix B. As noted in Technical Appendix B, its sampling grid does not include San Pablo Bay, Central San Francisco Bay, or the South San Francisco Bay, and as such the survey underestimates relative abundance in wet years. However, the 20-mm Survey encompasses a significant portion of the distributions of larval and small juvenile longfin smelt, including

nearly the entire range during dry years. Since 2002, 14 of 20 years have been classified by DWR as Critical, Dry, or Below-Normal (https://cdec.water.ca.gov/reportapp/javareports?name=WSIHIST). During that period, the trend in longfin smelt catches from the 20-mm Survey has been generally downward, and recently, remained relatively low (see Figure 3.5).

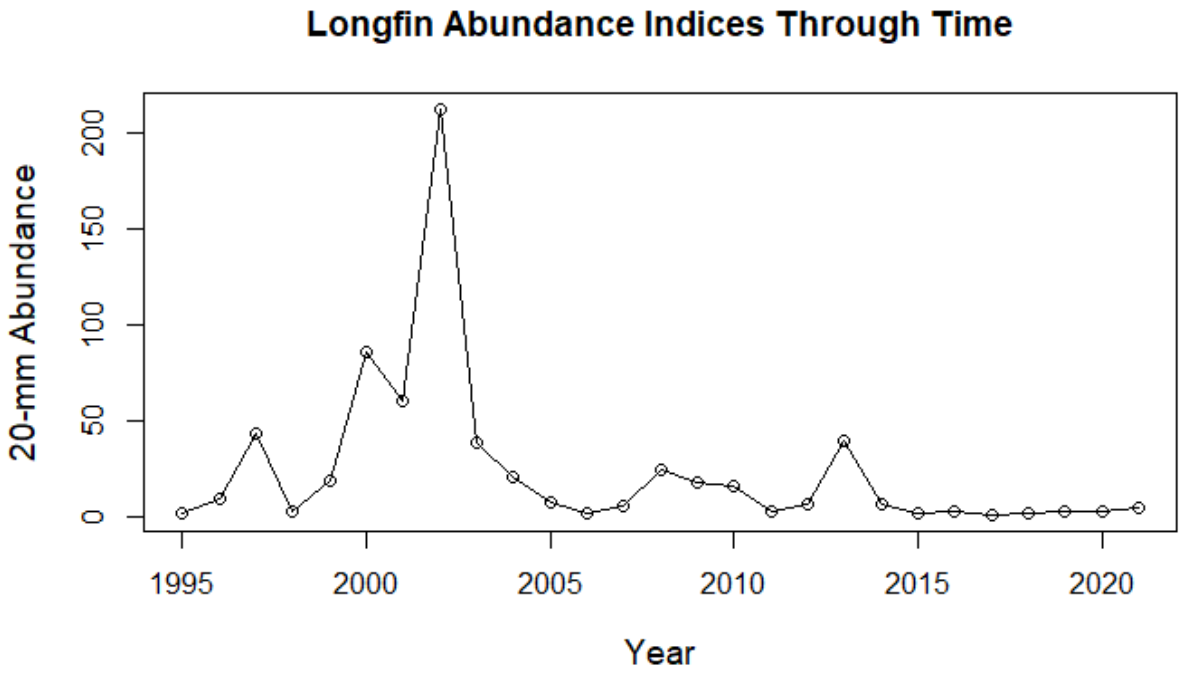


Figure 3.5: Longfin smelt abundance indices from the CDFW 20-mm Survey, 1995-2018 Source: CDFW 2021a

The FMWT, which captures both juvenile and adult longfin smelt, has the longest history of any survey that is considered effective for the Bay-Delta DPS and has been used to determine the status of the species since the late 1970s (Stevens and Miller 1983, pp. 431–432). Since that time, its peak (unitless) index value recorded was approximately 63,000 in 1982, although the highest ever index value reported was over 80,000 in its first year of completion (1967). The FMWT abundance index for longfin smelt has not exceeded 10,000 since 1983, has not exceeded 1,000 since 2006, fell below 100 for the first time in 2007, and has registered values less than 100 eight more times since then (see Figure 3.6). The 2021 FMWT index for longfin smelt was 323, which was the highest since 2011 (https://www.dfg.ca.gov/delta/data/fmwt/indices.asp).

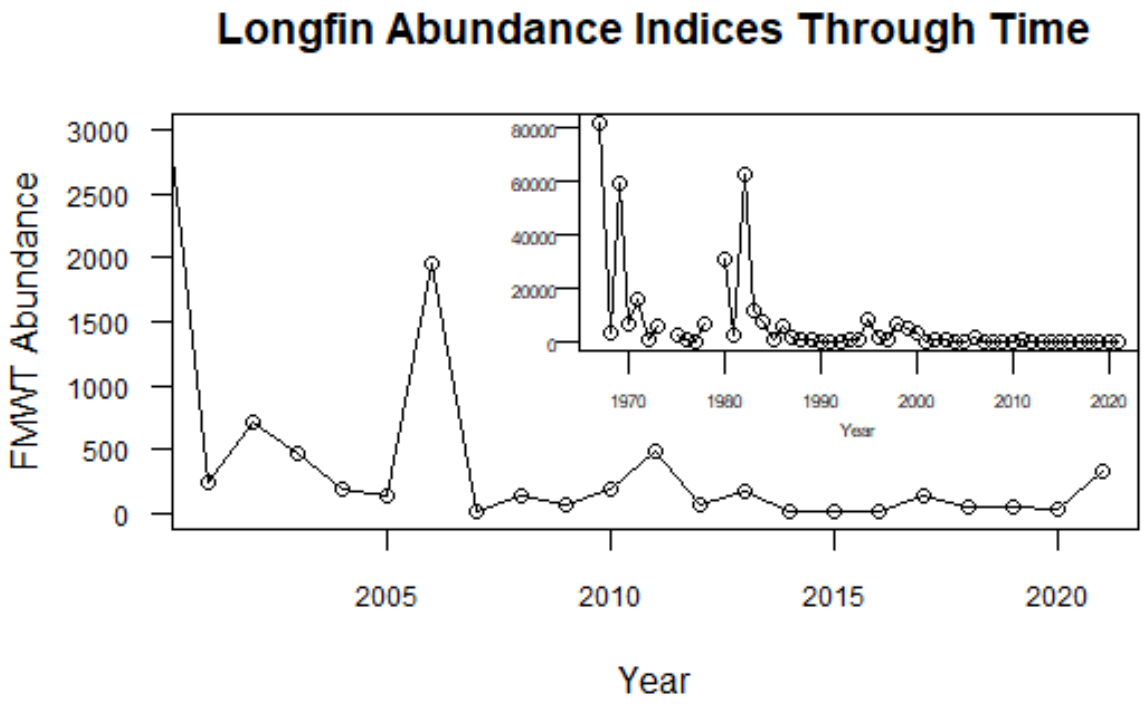


Figure 3.6: Longfin Smelt abundance indices for 2000-2020 from the FMWT (Inset displays time series since 1967). Source: CDFW 2021a

The San Francisco Bay Study (https://wildlife.ca.gov/Conservation/Delta/Bay-Study) samples low-salinity to fully marine waters of the estuary, utilizing both otter (OT) and midwater trawls (MWT); standardized sampling has occurred in most years since 1980. However, sampling was more sporadic in the 1990s and again in several recent years. By deploying two gears, the Bay Study samples near bottom as well as midwater to surface-oriented fishes (Feyrer *et al*. 2015, Fig. 5, p. 3614).

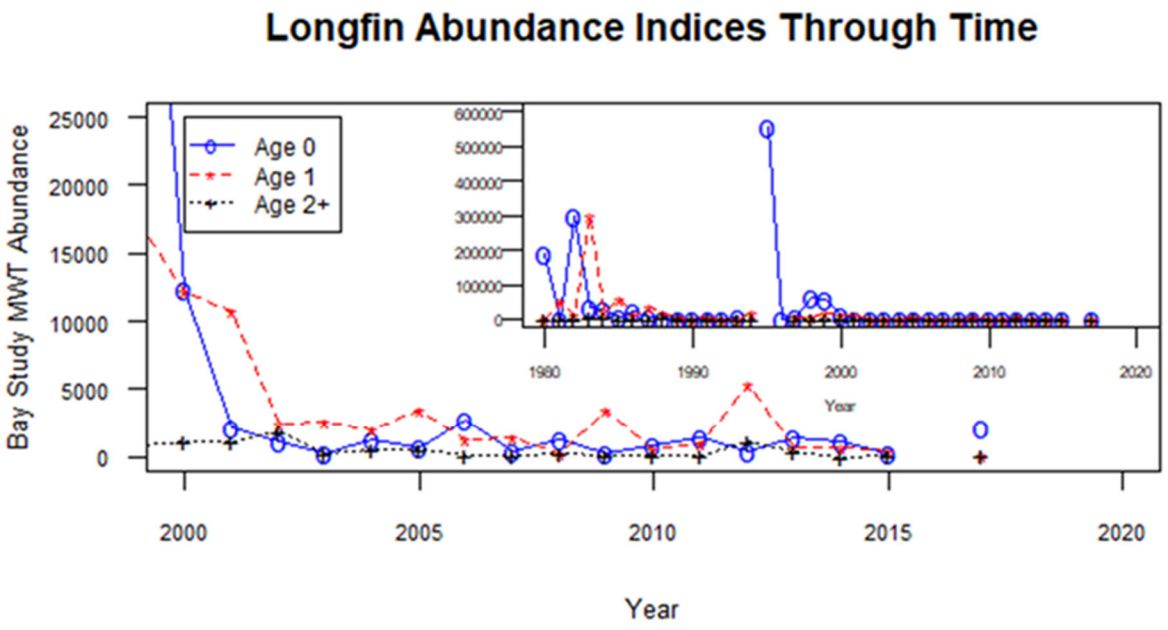


Figure 3.7: Age 0, Age 1, and Age 2+ longfin smelt abundance indices over time from the Bay Study Midwater Trawl, 1999-2017 (Inset displays time series since 1980). Source: CDFW 2021a.

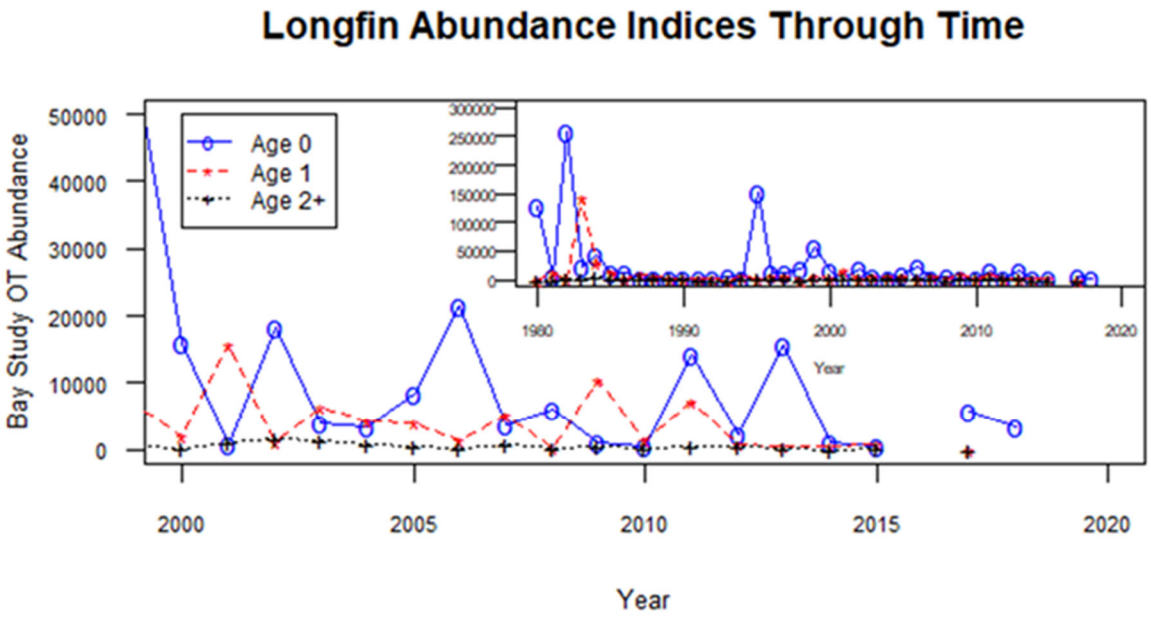


Figure 3.8: Age 0, Age 1, and Age 2+ Longfin Smelt abundance indices over time from the Bay Study Otter Trawl, 1999-2018 (Inset displays time series since 1980). Source: CDFW 2021a

Longfin smelt densities calculated from SLS monitoring data also indicate a decline in abundance. Prior to 2014, larval longfin smelt were generally detected at nearly every SLS sampling station in Suisun Bay,

the confluence of the Sacramento and San Joaquin Rivers, and Northern Delta at densities numbering in the multiple hundreds of individuals per 1,000 m^3 (see Figure 3.9). After 2014, detection frequencies declined and became more variable. Extended dry years compound the negative impacts to longfin smelt as the fish have not shown an ability to quickly recover and reoccupy upstream spawning habitats following drought. Successive years with such unfavorable conditions are likely to continuously suppress longfin smelt abundance which propagates through to the next spawning generation.

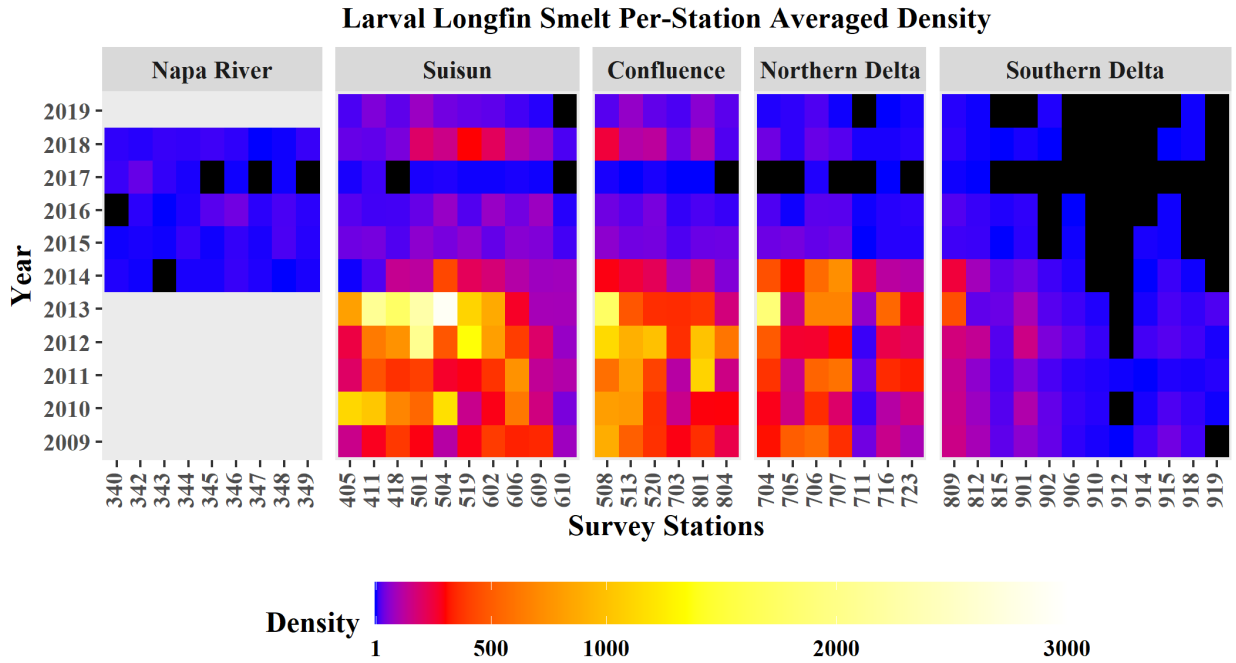


Figure 3.9: Per-station averaged densities for larval longfin smelt across four regions of the Upper Estuary. Densities were averaged at each station for each year of the Smelt Larval Survey. Legend presents average densities per 1,000 m3. Black boxes represent sampling stations that did not detect longfin smelt across all tows for that year. (Eakin 2021, p. 196).

Considering the consistently low abundance indices across all monitoring survey programs and life stages it is apparent that the DPS lacks the ability to recover to the higher abundance numbers seen in the past. All the best available field surveys for documenting long-term abundance trends suggest longfin smelt numbers have substantially declined over time, with current relative abundance reflecting small fractions of the species historical relative abundance. Even considering the small upticks in the most recent survey results, the general trend over time has been lower highs and lower lows in the indices. This supports the conclusion that abundance of all life stages has declined substantially over the course of several decades and that decline has generally continued in recent years. This decline has spanned three to four orders of magnitude in the overall abundance indices over the course of the historical record. In the next section, we summarize these trends numerically as estimates of average annual population growth and consider the consequences of such rates on the long-term viability of the population.

3.2.2. Baseline Scenario and Population Viability Analyses

While descriptive statistics summarizing the historical time series of established (and derived) official relative abundance indices can be informative, they do have limitations (as described herein and in Technical Appendix B) when it comes to inference about population viability, and certainly for ascribing causation relative to potentially important covariates. Therefore, we have performed additional modeling efforts to frame the available information in a context relevant to the purposes of this SSA.

We present a baseline scenario analysis, where we assume that there are no additional changes in the system over time; specifically, this approach assumes that current stressors remain the same and that they have the same magnitude and intensity of impacts on the Bay-Delta DPS into the future. We describe two population viability analyses (PVA; Morris and Doak 2002, Ch. 1 pp. 1–14), which use two different population models to summarize population growth rates and use them to forecast the risk of extinction into the future. These two PVAs, termed count-based and age-structured, use two different model structures: a broad view of the weight of evidence presented by the suite of monitoring surveys and a relatively detailed view of life stage specific vital rates. Details for these modeling approaches are described in Technical Appendix B & Technical Appendix E.

Count-Based PVA

In the first analysis, each abundance index is treated as an independent time series describing the pattern in abundance of longfin smelt. This analysis is done using a count-based PVA framework, which is mathematically and biologically simple as it summarizes the patterns of decline in each of the abundance indices and the impact that has on the probability of quasi-extinction in the future. Quasi-extinction was defined as 1% of the mean of index values since 2008 or lower (≤ 16). Estimates of population growth rates and variability were derived from all available years for each survey. The summaries of the abundance indices and associated count-based PVA represents a quantitative summary of the information that many biologists use to form their intuition about the status of the species.

Values for each of the indices of longfin smelt abundance have decreased substantially over the available time series (Figures 3.5-3.7). Even though this pattern is consistent across each of the indices, it can be difficult to interpret what changes in the indices indicate for population viability because each is constructed differently. When expressed as year over year changes to the indices, we derive estimates of annual population growth rates (lambda) that can be summarized in terms of their central tendency and spread. When population growth rates are less than one, the abundance declines and when population growth rates are greater than one, the abundance grows. Growth rates are multiplicative, so a growth rate of 0.5 indicates that abundance dropped by half, whereas a growth rate of 2 indicates a doubling population.

Mean population growth rates for individual surveys were less than one for most of the abundance index data series, which indicates that population size is declining over time (see Figure 3.10). Only the 20 mm Survey index produced an estimate greater than one. However, variability was high for all

surveys, and confidence limits on each of the estimates included one. This indicates that even though abundance tends to decline on average, in some years abundance will be stable or increasing.

To summarize the population growth rates from multiple surveys, we performed a meta-analysis on the mean growth rates for each of the surveys. Like most of the individual surveys, the estimated mean population growth value from the meta-analysis was less than one, but the variability around the mean was much lower. It is noteworthy that for some of the surveys it is not possible to calculate an abundance estimate when zero Longfin Smelt are caught. This has become an issue for some recent years and the inclusion of only index values from years with non-zero catches may bias estimates of mean population growth rates upward.

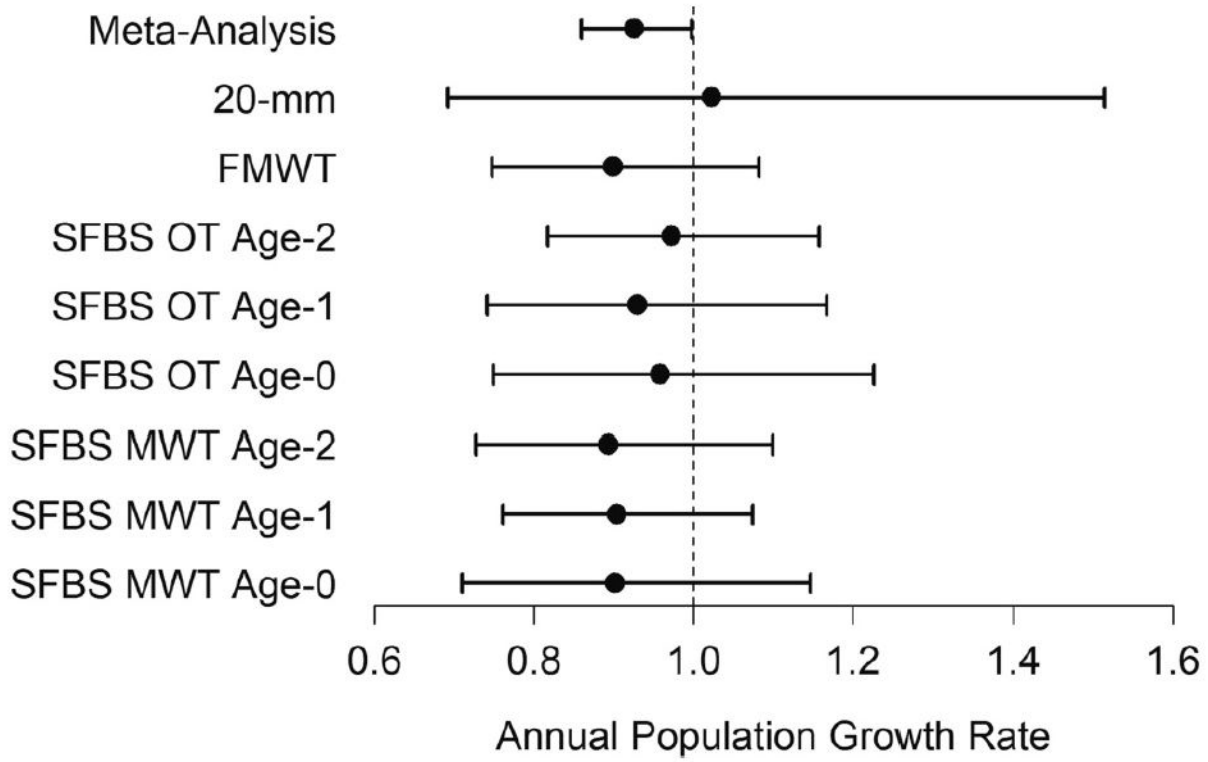


Figure 3.10: Mean population growth rates from several monitoring programs and a meta-analysis of the mean. Calculations were made using all available years for each survey, based on a count PVA framework. Error bars are 95 % confidence limits.

Based on the mean population growth rate values and applying the simplifying assumption that the rate of population growth is constant over time (i.e., applying the historically observed average rate of decline forward through time), we would expect the population to decline between roughly 42% and 89% over the next 20 years. The large variability observed in population growth estimates for each survey makes a sustained increase in population size unlikely, as indicated by the probability of quasi-extinction over time. Populations with highly variable growth rates tend to have low viability over time because adding variation tends to make a population grow more slowly over the long term and the

population size is more likely to fall below the quasi-extinction threshold than it is for populations with less variable growth rates (Morris and Doak 2002). Without intervention, populations with negative growth rates are expected to go extinct, regardless of the initial population size or the variability in their growth rates. An important question to investigate becomes when extinction is likely to happen because this determines the timeframe for implementing management actions to increase the growth rate.

To illustrate a timeline for extinction risk, we calculated a cumulative distribution function for the time to extinction based on the mean population growth from the meta-analysis, using the mean of the most recent non-missing abundance index values as an initial abundance, and the mean of quasi-extinction thresholds that we calculated for each of the indices. Mean probability of extinction (solid line) and 95% confidence intervals (dashed lines) were calculated using a modified extCDFfunction from the popbio package (Stubben and Milligan, 2007). The starting population size was set to the mean of the initial abundance index for the surveys. Quasi-extinction was defined as 2.6% of the initial abundance index, resulting in a quasi-extinction threshold of 50. The mean population growth rate exhibits a probability of quasi-extinction exceeding 20% when carried forward for two decades (see Figure 3.11), which was proposed by Lindley *et al*. (2007, Table 1, p. 4) as a criterion ascribing high extinction risk to Central Valley salmonid populations.

The shape of the quasi-extinction curve was influenced by the choice of initial abundance, with lower values producing a steeper slope in the near-term and a higher asymptote in the long-term than was observed for quasi-extinction curves associated with higher initial abundance values. The mean growth rates for each other individual surveys produce similar levels of extinction risk (Technical Appendix B).Based on the meta-analysis, the probability of quasi-extinction is 33% over the next twenty years and reaches 50% in 30 years (Fig. 3.11). Applying the same assumptions over a longer time horizon (i.e., 2050-2065), the probability of extinction for the Bay-Delta DPS under current conditions continues to increase. Here we present the results of a meta-analysis using values from the official abundance indices associated with various long-term monitoring surveys, but similar results were obtained using annual summaries of catch per tow as an alternative index of abundance for the surveys.

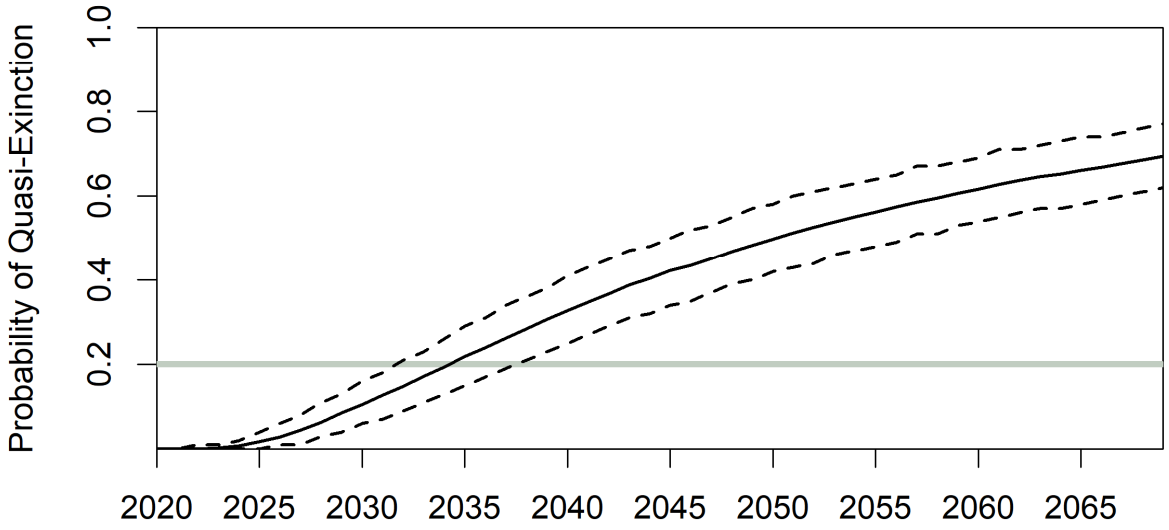


Figure 3.11: Mean probability of quasi-extinction (solid line), as calculated from the values in the meta-analysis, with bootstrapped 95% confidence bands (dashed lines). The wide, horizontal line highlights a 20% probability of quasi-extinction.

Age-Structured PVA

The second population viability analysis recognizes the need for a model that takes the age structure of the population and density dependence into account. In this analysis, we developed an age-structured state-space population dynamics model and estimated changes in vital rates for the population over time (Technical Appendix E). The population viability analysis that is based on this model explicitly draws on estimates of vital rates (survival and reproduction) that link the age classes. In this model, survival rates from one age class to the next were structured as a proportion, while reproduction rate was structured used the Beverton-Holt equation to represent density dependence. Data on the abundance of the age classes came from Bay Study's abundance indices for both of its sampling gears (midwater trawl and otter trawl). These were incorporated into the model as two observations of each time point.

The age-structured PVA evaluated four age-specific abundance thresholds for quasi-extinction risk: the estimated abundance index associated with the lowest 1st, 5th, 10th, and 15th lower percentiles of the estimated age-specific abundance indices. Specific values associated with each percentile are given in Technical Appendix E, but as a relative index of abundance, the values only have meaning in relation to each other. The population was considered quasi-extinct if the abundance estimate for any age class fell below the specified threshold. Populations. that reach quasi-extinction are considered unable to

recover, so if the abundance falls below the quasi-extinction threshold in any year, the population remains "extinct" for all subsequent years in that run of the model.

For each simulation, the initial abundance was drawn from the distribution of estimated age-specific abundance values for 2015, which was the last year of the continuous time series of indices for Bay Study. Vital rates were drawn from the distribution of parameters for all years in the data set (1980-2015). As such, the full historical dataset informed the simulations of future abundance, and the analysis assumes that the future will follow patterns similar to the past. The simulation was run 1000 times and the probability of quasi-extinction is the proportion of times the population reached quasi-extinction at or before the year specified.

The probability of quasi-extinction was sensitive to the choice of quasi-extinction threshold and, as we might expect, setting higher thresholds results in higher probabilities that the population will become extinct (see Figure 3.12). Even with this variability, the model results show a substantial risk of extinction even at low threshold values. For example, using the 10 percent quantile of abundance estimates as the threshold, approximately a 60% of simulations reached extinction by 2050. The 10 percent quantile line also meets Lindley *et al*.'s (2007, Table 1, p. 4) criteria for high risk of extinction (20% probability over two decades). All thresholds shown in Fig. 3.12 meet Lindley *et al*.'s criteria for moderate risk of extinction (>5% over 100 years).

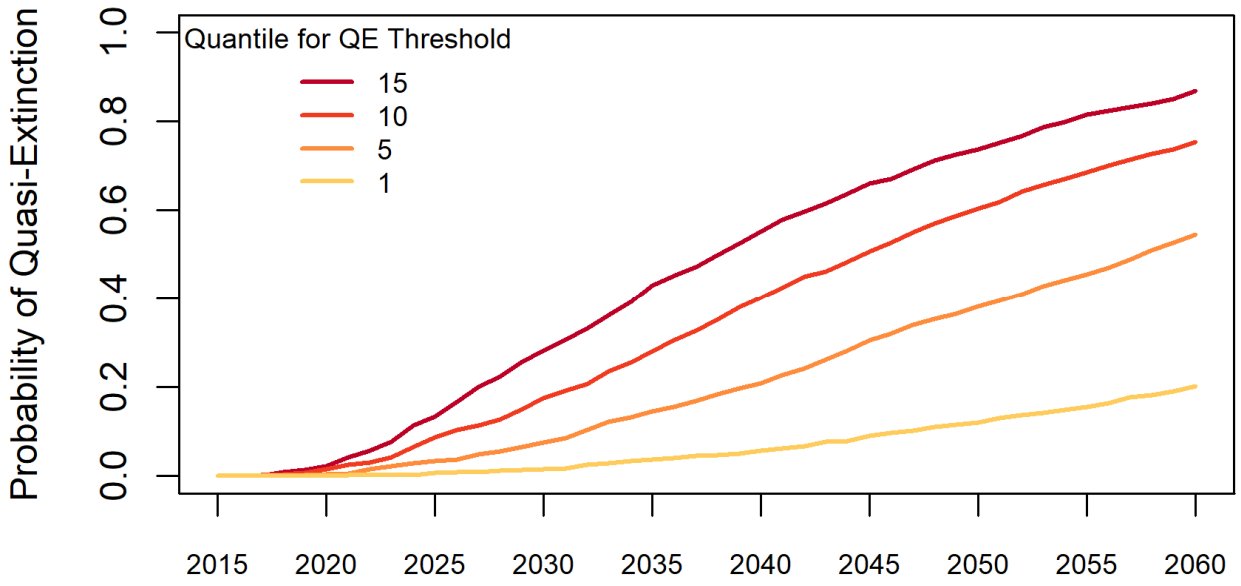


Figure 3.12: Estimated probability of quasi-extinction for four sets of age-specific abundance values, based on the percentiles at the lower end of the distributions of relative abundance estimates.

3.3. Summary of the 3 R's

Here, we conclude the current ecological viability by summarizing the species response to these stressors with an assessment of the 3 R's as they apply to the Bay-Delta DPS.

3.3.1. Redundancy

The San Francisco Bay-Delta DPS is a single, genetically-indistinguishable population with basically unidirectional gene flow to northern populations (I.e., a 'source' population), as evidenced by genetic data (Saglam *et al*. 2021, p. 1797). Additionally, nearby California estuary populations likely do not provide meta-population benefits to Bay-Delta longfin smelt because these populations appear to have declined in abundance and may not be permanently inhabited (Brennan *et al.* 2022, entire). Age structure of the population (more than one age class extant in the system) may yield some buffer against a year with a catastrophic event. However, it is not presently known how many spawning age-classes in any given generation of the Bay-Delta DPS exist. The spawners may: 1) be all or nearly all age-2 or, 2) include meaningful numbers of age-1 and, 3) possibly even age-3 fish. If there is only one spawner age-class in each generation as is the case in Lake Washington, the Bay-Delta DPS has less inherent resilience to poor conditions than if there are multiple spawning age classes. This is one of the most important uncertainties that limits our ability to model the DPS' population dynamics and by extension, evaluate the three Rs. Nonetheless, it is apparent in the trend data that sequential dry years and droughts have strongly contributed to population decline. Inhabiting a hydrologically-connected estuarine system, with a capability of facultative anadromy, the fate of the DPS is reliant upon the ecological condition of the San Francisco Bay-Delta and, to an extent, the DPS' ability to exploit the nearshore Pacific Ocean niche.

3.3.2. Representation

We assess the species representation (adaptive capacity) within the Bay-Delta based on the observed phenotypic plasticity in its life history (to the extent we can infer it from available data) and the current and future conditions within the system that may afford some capacity for behavioral adaptations that enhance growth, survival and reproduction. To the extent that lifestages can tolerate higher salinities (a capacity that increases over developmental time), the species does possess adaptive capability to move seaward, thus seeking cooler thermal refugia and opportunities to sustain its survival, growth, and development until their return to the Delta for spawning. However, like our rationale above, as evidenced by the species relative abundance and the trends observed over time, even with plasticity in its life history, the species has not been able to withstand the cumulative changes to the Bay-Delta ecosystem and has fared poorly amidst the suite of stressors acting upon it. In short, recent and current conditions in the SFE have constrained adaptive options, impaired fitness, and reduced survival. Therefore, we do not anticipate that either genetic or environmental diversity confers sufficient adaptive capacity to the Bay-Delta DPS to overcome the detrimental environmental changes (especially into the anticipated future—see Future Condition chapter, following).

3.3.3. Resilience

Given the biology, life history, and interconnectedness of the San Francisco Estuary that the Bay-Delta DPS inhabits, its lack of population redundancy, and our estimate that it possesses limited inherent adaptive capacity (low representation) within these ecological constraints, the factor most contributing to the Bay-Delta DPS' extinction risk centers on its population resilience (I.e., the interaction of extant stressors on the current standing population). The key stressors impacting the Bay-Delta DPS are inadequate frequency and magnitude of favorable freshwater flow conditions for survival, growth, and reproduction; limited food supplies; and small population size. It is acknowledged that these factors are all ecologically interrelated.

Reduced freshwater flow

The biggest stressor impairing production of longfin smelt is decades of accumulating impairment of freshwater flow caused by the development and redistribution of water supplies in California. Freshwater flow to the estuary, especially as Delta outflow, is the most important species need. This DPS has a long-standing relationship with Delta outflow. With the exception of 2021, abundance indices during Critical, Dry, and Below-normal year types were never higher than the preceding abundance index of the immediately preceding Wet or Above-normal year type. Further, no FMWT index for longfin smelt has ever surpassed the initial 1967 index and in the overbite clam era (1987-present), only one index (1995) has even gotten within ten percent of the 1967 index. Thus, the decline of the Bay-Delta DPS, which is now in its sixth (observed) decade, has led to very low abundance indices.

Loss of food

The invasion of the overbite clam was a catastrophic event as it had an immediate and permanent impact on the pelagic food web in the San Francisco Estuary. The overbite clam both reduces the base primary producers of longfin smelt's food web and competes directly with larval longfin smelt by consuming some of the same prey. The suspected role of food limitation is a lower carrying capacity for longfin smelt, even in wet years. This stronger limit on juvenile fish production has interacted with insufficient frequency of wet years to contribute to persistent species decline. An apparent lack of trophic adaptability provides longfin smelt with little representation to withstand accumulating changes to the food web. This is further evidenced with the species inability to recover from the catastrophic overbite clam invasion.

Small population size

Though not a stressor in itself, small populations are more vulnerable to demographic, stochastic, and allele effects (Foden *et al.* 2016, p. 40). Based on field survey indices, the abundance of the Bay-Delta DPS has been at, or near, record lows for the past decade. The current small population size thus lowers the species resilience, and limits opportunities for higher recruitment during favorable years. Small populations also have reduced capacity to adapt to environmental changes (representation) due to limited genetic variation (Willi *et al.* 2006, p. 440). Mechanisms for this include: lower fitness owing to

environmental stress (Willi *et al*. 2006, p. 447), inbreeding depression, genetic load, or nongenetic allele effects (Reed 2005, p. 556). However, we lack the information needed to quantitatively assess whether the small population size of the Bay-Delta DPS affects the species representation.

Other Potential (and/or Historical) Stressors

Temperature: Laboratory studies have suggested that longfin smelt are more physiologically sensitive to elevated water temperature than the delta smelt (Jeffries et al. 2016, p. 1714). The 20 °C threshold suggested for juvenile longfin smelt is commonly exceeded over large areas of the estuary, particularly the Delta and the lower Sacramento and San Joaquin rivers during the summer months (Vroom et al. 2017, p. 9904; data from CDEC, CENCOOS, and USGS), possibly coaxing the fish to seek cooler oceanic waters. Currently, longfin smelt are able to use facultative anadromy, albeit with a strongly declining population trend in the Bay-Delta—suggesting this strategy may be insufficient to mitigate temperature stress. The Bay-Delta DPS' ability to continue to survive in the Bay-Delta given anticipated warming is discussed in the Future Condition chapter, below.

Entrainment: The best currently available information suggests that if OMR management comparable to what has occurred since 2009 continues, the entrainment of longfin smelt will not have a strong influence on the viability of the Bay-Delta DPS.

Loss of Tidal Marshes: It is unknown how important historical tidal marshes were to longfin smelt, but they currently use a mixture of tidal marsh and open-water habitats. Tidal wetlands and marshes in the San Francisco Estuary were largely converted to other uses by about 1920 and have recently begun to be restored. The loss of tidal marsh habitats may have hampered species productivity, but to date, there are no indications that restoration has been sufficient to stem the decline. Therefore, we cannot conclude whether or not the species has lost resilience due to landscape changes that occurred in the 19th and 20th centuries. The quantitative contributions of restored estuarine marshes to larval growth and rearing remains a potentially important science question in support of longfin smelt conservation.

In Table 3.1, following, we summarize how current drivers are understood to be impacting the Bay-Delta DPS under our evaluation within this SSA.

Table 3.1: Summary of Threats under Current Condition for the San Francisco Bay Delta DPS Longfin Smelt

Stressor/Threat	Life Stage(s)	Mechanism	Magnitude of Threat
Reduced freshwater flow	Larvae and juveniles	Productivity reduction leading to recruitment loss	
Loss of food	All	Productivity reduction due to food web collapse and increased competition	
Small Population Size	All	Reduced stock for reproduction through succeeding cohorts, greater vulnerability to stochastic events, Allee effects	
Temperature Stress (Habitat Constriction)	Adults and late juveniles	Physiological stress: Escape to cooler environments as adults is an option	
Increasing temperature	Larvae and younger juveniles	Lethal mortality to early lifestages: Unable to avoid as larvae	
Entrainment	Larval and younger juveniles	Entrainment to the South Delta export facilities and other diversions	
Contaminants	All	Direct mortality through reduced fitness or indirect through food web	
Tidal habitat loss	Larvae and juveniles	Reduced rearing area lowering growth and recruitment	

In summary, the Bay-Delta DPS of longfin smelt is not resilient; more than five decades of declining abundance indices have documented an inability of this fish to sustain itself in the contemporary estuary. The Bay-Delta DPS also has extremely limited redundancy, since it effectively represents a single population inhabiting the San Francisco Bay-Delta and nearshore ocean environment. The representation of the Bay-Delta DPS is thought to be limited as well, reflecting that same declining abundance trend and no discernable and quantifiable compensatory adaptation to historical ecological conditions. Based on our evaluation of the 3 Rs, we conclude the San Francisco Bay-Delta DPS of the longfin smelt is highly vulnerable in its current condition.

Chapter 4 – Future Condition

In the previous chapter, we evaluated the current condition of longfin smelt and described how several stressors have affected the DPS and its habitat. In this chapter we provide information related to the stressors most likely to act on the longfin smelt and its habitat in the future and how those stressors may influence the future condition of the Bay-Delta DPS. It is unlikely that the conditions for the baseline scenario in the count-based PVA presented in Chapter 3 will indefinitely persist owing to known and potential changes anticipated in the future. In order to provide some insight into the future conditions for a particular species, we construct plausible scenarios to assist in articulating how the species needs, threats, and habitat conditions may change in the future.

In the Current Condition chapter, we presented a baseline scenario, where no changes are experienced in the system. In our approach to the species future condition, we present future scenarios of changes in the San Francisco Estuary. We then assess how the species has responded in the past and use that information to help predict the species' response in the future—the most plausible outcomes we can capture in our scenario planning. We finally close with a summary discussion projecting the future ecological scenario the Bay-Delta DPS will most likely face and discussing how that might be anticipated to either increase, decrease, or leave unchanged the risk profile for specific life stages of the Bay-Delta DPS.

4.1. Scenario Planning and its Application

Scenario planning is a comprehensive exercise that involves the development of scenarios to capture a range of plausible future conditions. That development is then followed by an assessment of the potential effects of those scenarios on a given species. Scenarios are not predictions or forecasts of what *will* happen in the future for a species but are projections or explorations into the range of conditions that may exist based on current information (see Figure 4.1). The scenarios are intended to provide the "upper" and "lower" bounds of plausible conditions (see Figure 4.2), outline uncertainties, and provide decision makers with a means for managing risk and maintaining flexibility in current and future decisions.

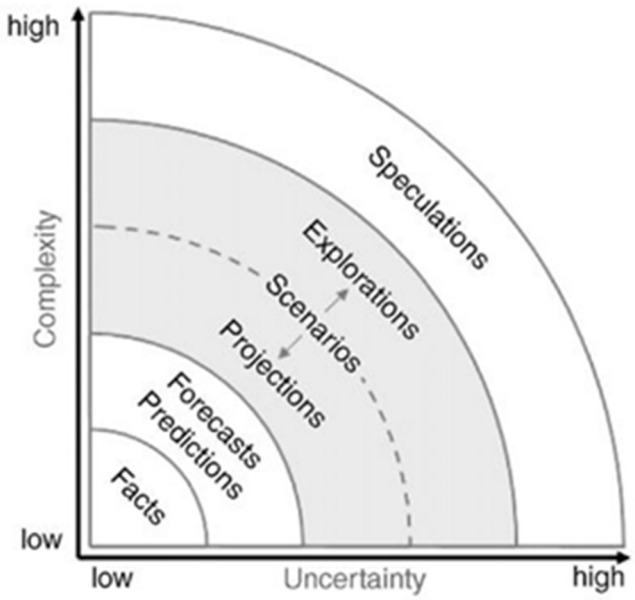


Figure 4.1. The levels of uncertainty and complexity in situations for which scenarios can be useful in considering future possibilities (adapted from Rowland *et al*. 2014). Gray areas represent speculative reasoning that have a higher attendant uncertainty associated with the scenario planning, as these are based upon forecasting that includes its own attendant uncertainty, as well as conceptual risk assessment founded on admittedly imperfect mechanistic understanding of causation (or multifactorial processes and emergent processes that are even more difficult to predict).

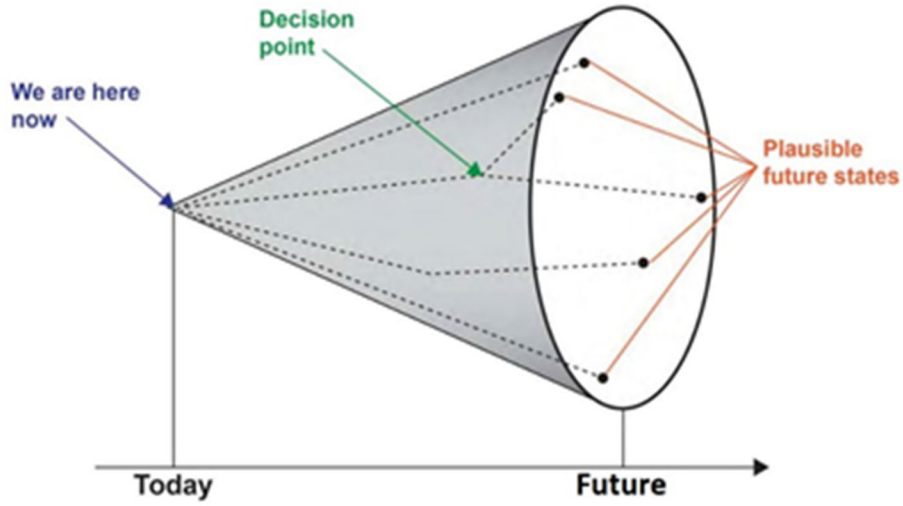


Figure 4.2. Conceptual diagram of the broadening range of plausible alternative futures as one moves farther away from the present and different events and decision points shift trajectories. (Rowland *et al*. 2014 (p. 6) adapted from Timpe and Scheepers 2003, entire). This conical projection is a visual presentation of the SFB DPS longfin smelt "risk profile" we attempted to capture in this chapter of the SSA.

A range of time frames with multiple possible scenarios allows us to create a "risk profile" for the longfin smelt Bay-Delta DPS and its viability into the future. While we do not expect every condition for each scenario to be fully realized, we are using these scenarios as examples of the range of possibility. For each scenario, we describe the stressors that are reasonably anticipated to occur for the DPS and how they may change in the future. We used the best available science to predict trends in future threats facing the Bay-Delta DPS. Data availability varies across the range of the DPS. Where data on future threats or trends were not available, we looked to past threats and their trends, and evaluate if it is reasonable to assume these trends will continue into future and to what degree.

To assess the future viability of the longfin smelt Bay-Delta DPS, we used published information related to the varying conditions of the San Francisco Estuary, including future climate change information and projected increases in water demand, and how these changes may impact how well the estuary can support longfin smelt in the future.

For this SSA, the foreseeable future was assessed out through the year 2050. This period represents our best understanding of the projected future conditions related to climate change and water demands based on anticipated population and agricultural growth. However, we also provide scenarios and data projections through 2100, as they are scientifically available and published, and each of our climate change assessment sections may have its own respective suitable foreseeable future timeframe. As described under the stressors in the Current Condition chapter, reduced freshwater flow, temperature, and invasive species as a source of food limitation have had the largest impact on the DPS, so here we specifically analyze how these threats may change in the foreseeable future.

4.2. Introduction to climate change

Our analysis includes consideration of ongoing and projected changes in climate. The terms "climate" and "climate change" are defined by the Intergovernmental Panel on Climate Change (IPCC). The term "climate" refers to the mean (average) and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements (IPCC 2013a, p. 1450). The term "climate change" thus refers to a change in the mean or variability of one or more measures of climate (for example, temperature or precipitation) that persists for an extended period (IPCC 2013a, p. 1450).

Measurements spanning several decades demonstrate that changes in climate are occurring, and that the rate of change has increased since the 1950s. Examples include warming of the global climate system, substantial increases in precipitation in some regions of the world and decreases in other regions (for these and other examples, see: Solomon et al. 2007, pp. 35–54, 82–85; IPCC 2013b, pp. 3–29; IPCC 2014, pp. 1–32). Specific to the SFE, Bashevkin et al. (2021, p. 2) found a temperature increase averaging 0.017°C per annum, over the last 50 years, with increases most widespread in the late-fall through winter (November to February) and spring (April to June) with warming fastest in the northern regions of the Delta. Climate change models predict increased extremes in winter precipitation frequency and intensity, but these models do not agree on whether the overall impact of these extremes will result in an increase or decrease in annual precipitation (Polade et al. 2017, p. 7).

Scientists use a variety of climate models, which include consideration of natural processes and variability, as well as various scenarios of potential levels and timing of greenhouse gas (GHG) emissions, to evaluate the causes of changes already observed, and to project future changes in temperature and other climate conditions (Meehl *et al*. 2007, entire; Ganguly *et al*. 2009, pp. 11555, 15558; Prinn *et al*. 2011, pp. 527, 529). All combinations of models and emissions scenarios yield similar projections of increases in average global surface temperature (commonly known as global warming), until about 2100 (Wulff *et al.* 2021, entire). Although projections of the magnitude and rate of warming differ after about 2030, the overall trajectory of all the projections is one of increasing global warming through the end of this century. Thus, there is strong scientific support for projections that warming will continue through the 21st century, and that the magnitude and rate of change will be influenced substantially by the extent of GHG emissions (Meehl *et al*. 2007, pp. 760–764, 797–811; Ganguly *et al*. 2009, pp. 15555–15558; Prinn *et al*. 2011, pp. 527, 529; IPCC 2013b, pp. 19–23). See IPCC 2013b (entire), for a summary of other global projections of climate-related changes, such as frequency of heat waves and changes in precipitation.

Global climate projections are informative, and we present the best scientific information available at the time of our analysis. Projected changes in climate and related impacts can vary across and within different regions of the world (IPCC 2013b, pp. 15–16). Therefore, we used "downscaled" projections when they were available and have been developed through appropriate scientific procedures. Downscaled projections provide higher resolution information that is more relevant to the spatial scales used for analyses of a given species (see Glick *et al*. 2011, pp. 58–61, for a discussion of downscaling).

In our climate change analyses, two plausible future scenarios were considered. We selected representative concentration pathways (RCP) 4.5 and 8.5 as the bookends for our analysis. RCPs 4.5 and 8.5 correspond to radiative forcing values in 2100 of 4.5 and 8.5 watts/m^2, which represent scenarios with an effort at abating global GHGs and a business-as-usual scenario, respectively (see van Vuuren *et al.* 2011, pp. 8–10 for RCP model development). Descriptions on how these scenarios were developed and chosen are provided in each of our climate change subsections, and the anticipated effects of each scenario on the risk profile for the longfin smelt DPS are also summarized in each analysis. We used our expert judgment and appropriate analytical approaches to weigh relevant information, including uncertainty, in our consideration of the best scientific information available regarding various aspects of climate change and their most likely impacts on longfin smelt.

4.2.1. San Francisco Bay-Delta Climate Change

There are three climate change drivers that are likely to affect the future condition of the Bay-Delta DPS; warming air temperature, changing precipitation patterns, and sea level rise (see Figure 4.3). Where trends can be reasonably predicted, these drivers of change are expected to have mostly negative impacts on the DPS, both individually and as interactive forces or incentives on the spatial and temporal distribution of longfin smelt and their life history transitions.

Driver		
Warming air temperature	Warmer Bay-Delta water temperature in the fall-winter	• Later Estuary re-entry (?)
	Warmer Bay-Delta water temperature in the spring-early summer	• Compresses spawning season (-) • Compressed estuary rearing (-)
	Less precipitation falling as snow	• Higher winter flow (+) • Lower spring flow (-)
Changing precipitation patterns	More frequent dry years	• Fewer years of strong recruitment (-)
Sea level rise	Higher salinity per unit of freshwater input	• Landward encroachment of the LSZ (-)
	Changes to terrestrial-aquatic ecotones	• Unpredicted habitat changes (?)

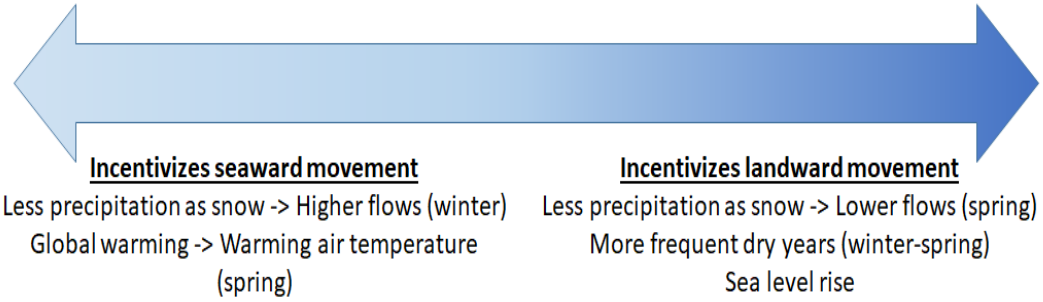


Figure 4.3. Summary of climate change drivers that are affecting and will continue to affect the San Francisco Estuary, the linkages between these drivers and mechanisms affecting longfin smelt, and the anticipated population-dynamic consequences. The direction of impact is shown in parentheses in the outcomes column: (+) = positive effect, (-) = negative effect, and (?) = uncertain direction of effect. The color graded arrow at the bottom depicts interactions among driver-linkages showing how they are expected to interact to create contrasting incentives for where in the estuary longfin smelt will attempt to spawn as adults and where the resulting progeny will rear.

There is greater certainty that sea level will rise leading to greater average saltwater intrusion into the estuary and Delta. The National Academy of Sciences (NAS) projected that sea levels along the California coast south of Cape Mendocino will rise 4–30 cm (2–12 in) by 2030, 12–61 cm (5–24 in) by 2050, and 42–167 cm (16–66 in) by 2100 (NAS 2012, p. 131) compared to year 2000 sea levels. Research indicates that the coastal land area south of Cape Mendocino is sinking at an average rate of about 1 mm (.04 in) per year, although Global Positioning System (GPS)-measured rates vary widely (-3.7–0.6 mm per year) (NAS 2012, p. 93). The NAS committee used output from global ocean models under an IPCC (2007) mid-range greenhouse gas emission scenario (NAS 2012, p. 5). However, carbon dioxide emissions from fossil fuels for the past decade have been at the high end of IPCC scenarios owing to rapid economic growth in developing countries (Le Quéré *et al.* 2009, pp. 831–832).

Climate change could affect longfin smelt due to increases in water temperature. Using projected air temperature global climate models (GCMs), Brown *et al.* 2016b (pp. 5–6) created water temperature projection models spanning 20 areas in Suisun Bay and the Delta through 2100. Although these temperature projections were applied to delta smelt, the spawning window metric included an upper temperature limit of 20 °C which is an approximate upper thermal limit for rearing post-larval and juvenile longfin smelt (Jeffries *et al.* 2016, p. 1712). Brown *et al.* (2016b, Fig. 4, p. 13) predicted the delta smelt spawning window would shift earlier in the year. The predictions had substantial spatial and temporal variability, but the general conclusion is that climate warming will affect the estuary faster in the spring as the century progresses. This will increase bioenergetic stress on young longfin smelt and may force them seaward earlier in the year where water temperatures will be cooler. Their capacity to adapt to these changes at the time scale they are projected to occur was questioned by Jeffries *et al.* (2016, p. 1714).

In addition, the spawning window for longfin smelt is earlier than that of delta smelt; longfin smelt typically spawn at temperatures between 7 and 14 °C (Wang 2007, p. 38). Thus, their spawning window will likely be shortened as well, resulting in a smaller average annual number of days available for spawning. Laboratory tests have shown that embryotic and larval development is more successful at 9 and 12 °C than at higher temperature; temperatures of just 15 °C were detrimental for embryo hatching success and larval survival (Yanagitsuru *et al.* 2021, pp. 5–7). Thus, even marginal increases in water temperature could shorten the spawning season and reduce survival of early life stages.

In the following section we provide an in-depth analysis of how climate change may affect the San Francisco Estuary over time, and how the projected changes could in turn impact longfin smelt. We use published datasets that project water temperature increases, shifts in the hydrograph, changes in the frequencies of water year types, and sea level rise specific to the San Francisco Estuary. We then hypothesize how longfin smelt may respond to these projected changes based on the scenario planning exercise. The full dataset is available at https://github.com/USFWS/Longfin-Smelt-Climate.

4.2.2. Temperature trend future condition analysis

To estimate the effect of global warming as it pertains to the effects of rising temperatures on longfin smelt, we use a spatial and temporal approach in Suisun Bay and the Delta, which as described in the species Life History section, are important spawning and rearing habitats for longfin smelt, particularly in low outflow years. Water temperature change projections for San Francisco Bay were not available at time of this writing, so our analysis excludes San Francisco Bay and its tributaries. The embryonic through early juvenile life stages are when longfin smelt are believed to be most vulnerable to warming temperatures because these early life stages do not possess the ability to migrate to the cooler waters of central San Francisco Bay and the coastal ocean.

USGS (Wulff *et al.* 2021, entire) produced regionally downscaled climate change model outputs to forecast water temperatures at fixed locations in the upper San Francisco Estuary, specifically the Sacramento-San Joaquin Delta and Suisun Bay. The downscaled average daily air temperatures from the

climate change scenarios were sub-sampled for the Delta region, and then averaged to produce Delta daily average air temperature for 2010 through 2100. We relate these projected water temperatures to multiple thermal physiological and behavioral thresholds of longfin smelt across life stages to assess future thermal habitat, spawning, and rearing suitability. Our approach is similar to that of Brown *et al.* 2016b (p. 5) for delta smelt, except we reflect how the temperature changes may affect the life history of longfin smelt.

USGS produced water temperatures corresponding to 10 GCMs for RCP 4.5 and 10 GCMs for RCP 8.5. We averaged each of the 10 GCMs to develop ranges and medians for each RCP. The RCP 8.5 scenario represents rapid economic growth with little effort to limit or reduce emissions, reaching atmospheric greenhouse gas (GHG) concentrations exceeding 900 parts per million (ppm) by 2100 (see Figure 4.4). The RCP 4.5 scenario represents a more moderate scenario, with atmospheric GHG concentrations increasing through mid-century, reaching a concentration of 550 ppm, followed by stabilization (van Vuuren *et al.* 2011, p. 17). It should be noted that the State of California Resources Agency focuses on RCP 8.5 for the Delta Climate Change Vulnerability Assessment, as it most closely resembles current emissions trends (Schwalm 2020, p. 19657) and presents a more precautionary approach (DSC 2021, p. 3-3). However, other experts have argued that emissions over the past decade may be closer to median scenarios such as the RCP 4.5 (Hausfather and Peters 2020, no pagination). For our rising water temperatures analysis, we define the foreseeable future as 2050, which is also a precautionary approach as both RCPs show similar trends through mid-century. However, we also present scenarios through 2100 as data are available.

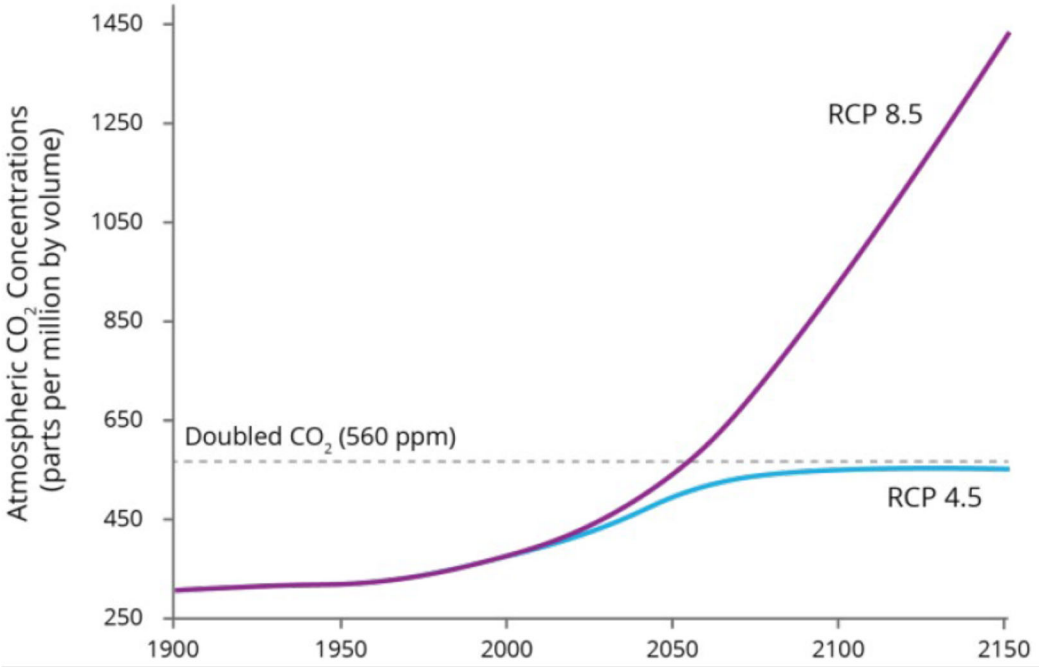


Figure 4.4: Emission of carbon dioxide under the RCP 4.5 and 8.5 scenarios. Source: Modified from van Vuuren et al. 2011, p. 17

As described under Life History, subadult and adult longfin smelt rear in central San Francisco Bay and the coastal ocean to escape warming temperatures that occur in much of the estuary during the summer. This movement is likely part of the DPS' adaptive capability and could be facilitated as water temperatures rise toward 20 °C in the late spring. Likewise, longfin smelt have not been known to return to most of the estuary until temperatures drop below 22 °C in the autumn. To project how hospitable habitat in the Delta may change in the future, we project the number of days water temperatures may be above 22 °C from a decadal median through 2100.

Based on catch data, longfin smelt adults are present under favorable temperature conditions in Suisun Bay and the Delta. For our analysis, we project water temperatures at Martinez, Mallard Island, Decker Island, Jersey Point, and the North Delta in the Sacramento Deepwater Shipping Channel. From the 2010-2020 time period, the annual average number of days above 22 °C was 56, 83, 91, 101, and 91 days, respectively. This suggests that longfin smelt are currently unable to inhabit the upper estuary for up to three months each year, which is broadly consistent with survey data.

By our established foreseeable future of 2050, it is projected that the average number of days above 22 °C will increase by approximately 30 days for most sites under the RCP 8.5 scenario, with the greatest increase projected at Martinez which is projected to see an extra 36 days with temperatures above 22 °C (see Figure 4.5). RCP 4.5 projects increases of 20 additional days in the lower Sacramento and San Joaquin Rivers to 30 additional days in Suisun Bay at Martinez (see Figure 4.6). Both scenarios suggest a shortening of the favorable temperature season for both post-larval/juvenile longfin smelt in the late spring and juvenile/adult longfin smelt in the fall. By 2100, the number of additional days above 22 °C is projected to increase by approximately 57-67 days for most sites under the RCP 8.5 scenario, which would result in substantially less time available for longfin smelt to use the estuary in many years.

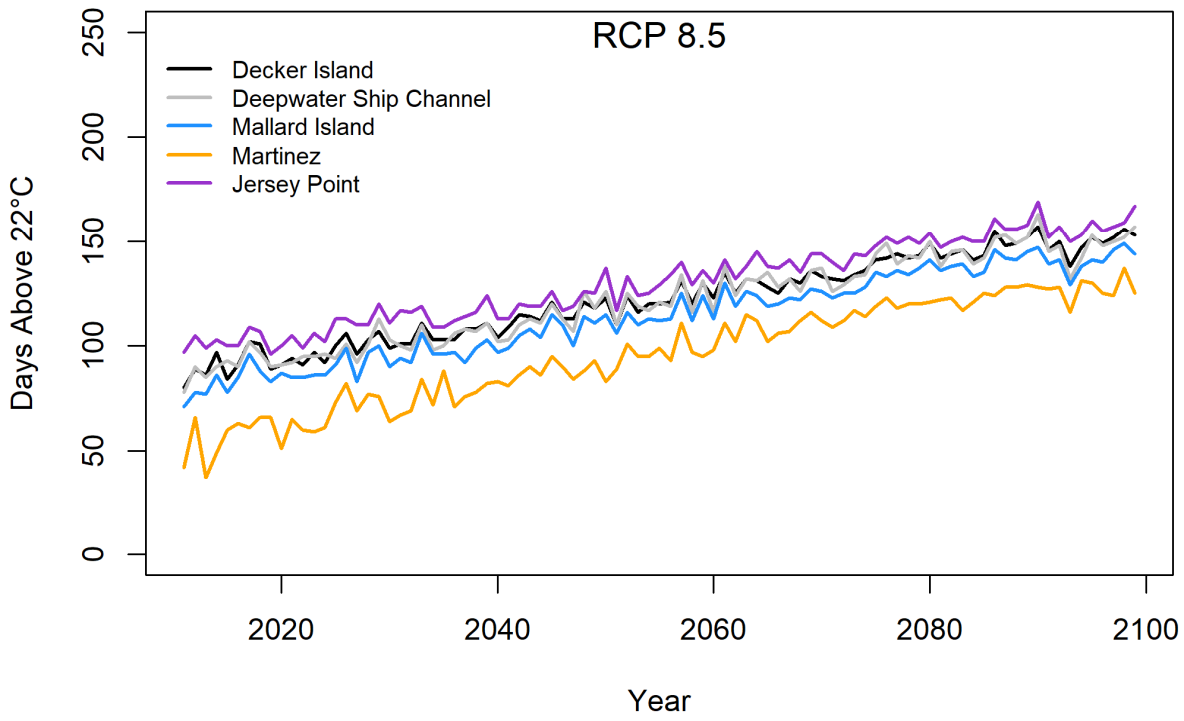


Figure 4.5: Projected number of days above 22 °C at five locations under the RCP 8.5 scenario.

As described in the Species Needs and Life History sections, longfin smelt spawning typically begins in December and continues until temperatures approach 14 °C. Larval catches from survey data peak at 8 °C, and gradually become less frequent at all temperatures warmer than that (Grimaldo et al. 2017, Fig. 6, p. 1778). No longfin smelt of any life stage have been found in water above 22 °C (Jeffries et al. 2016, Fig. 2, p. 1709). Here, we use 14 °C as a threshold to estimate how the spawning window may change in the future due to rising temperatures, and we use the 22 °C threshold to estimate when longfin smelt will seasonally migrate seaward of the Delta. In addition, we use 8-12 °C as the favorable conditions temperature zone for larval longfin smelt.

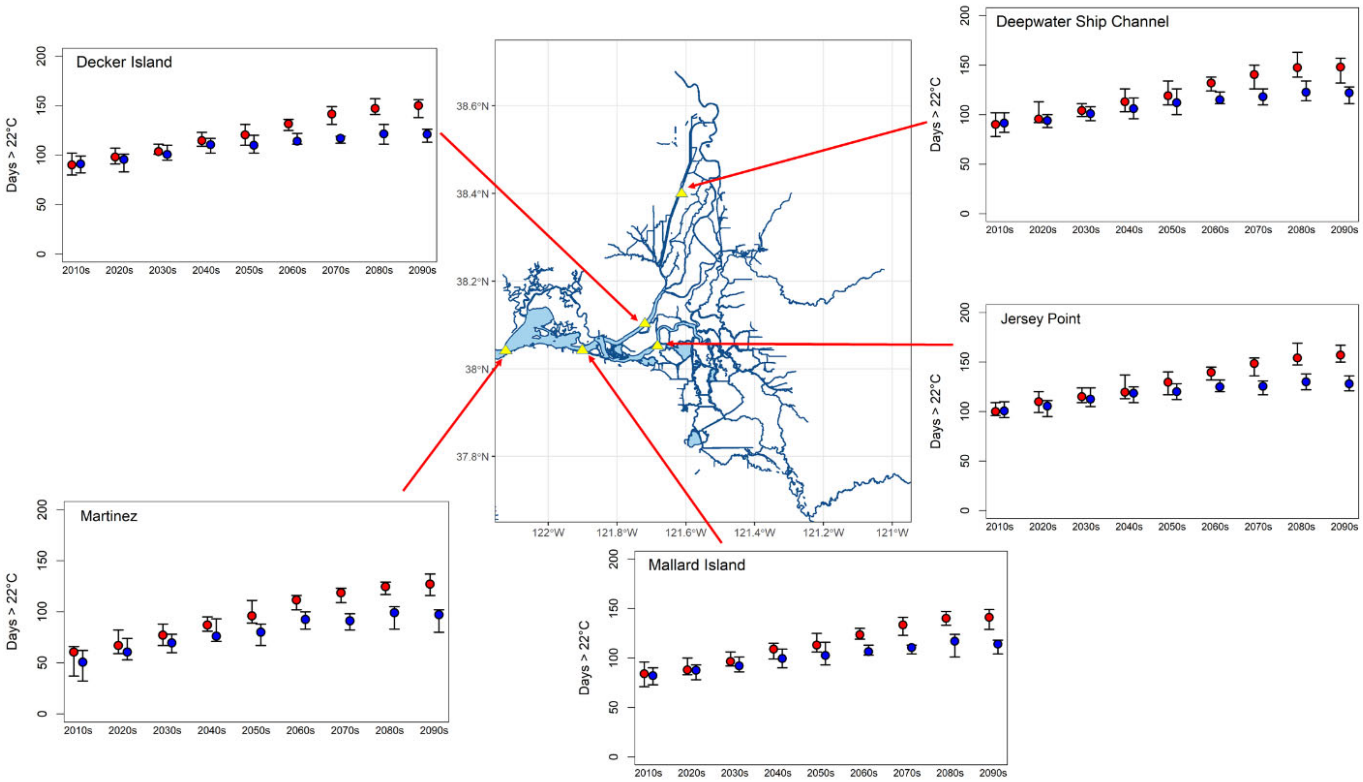


Figure 4.6: Map of the projected number of days above 22 °C at five locations under both the RCP 8.5 (red) and RCP 4.5 (blue) scenarios. Lines extend between the minimum and maximum average projections for the 10 GCMs. (Map source: Bashevkin and Barros 2021, entire.)

Taken together, the Delta and Suisun Bay are projected to become less hospitable for longfin smelt in the future, with a suitability window that could decrease by more than two months in an average year (this would result in cutting the currently recognized habitability window by more than half). If they can, the young-of-the-year fish would need to make an earlier migration towards the Bay and the ocean. Individuals that are not developed enough to physiologically tolerate marine salinity as temperatures warm toward stressful thresholds will likely suffer higher mortality. Fish that survive their first year of life should be able to more easily come and go from the ocean as needed until they make a final return to spawn and die. We anticipate that adult longfin smelt will return to the estuary later in the fall to avoid rising temperatures. There is no information available to predict whether this will delay spawning or not, but higher springtime water temperatures will likely truncate spawning opportunities with increasing frequency.

Based on field survey data, there is evidence that juvenile longfin smelt have been leaving the Delta earlier in the summer and returning later during the autumn by 1-2 months, suggesting that they may be spending more time in the coastal ocean than they did historically (Tobias and Baxter *unpublished preprint available at https://www.preprints.org/manuscript/202101.0512/v1*). This delayed return could be part of the species' response and adaptation to a warmer estuary.

In addition, back calculating and estimating hatch times for longfin smelt has suggested that hatching has been occurring earlier in the spawning season (Lewis and Hobbs 2021, unpublished data; Miller 2021, slide 14). Preliminary analysis suggests that outflow, temperature, and salinity may not be drivers of this early hatching behavior (Miller 2021, slides 15-17). But taken together, the spawning window for longfin smelt may have already shortened over recent history, based on field survey data.

To evaluate how the longfin smelt adult spawning window could change due to rising temperatures, we first assessed the environmental conditions where mature females were caught during field surveys. CDFW has been conducting the Longfin Smelt Fecundity Study since 2008 (Tempel and Burns 2021, slide 3), and the data have shown that mature females were collected between November and February. The majority of mature females have been caught at the confluence of the Sacramento and San Joaquin rivers, although detections occur throughout the estuary (see Figure 4.7). The spawning window has been calculated by subtracting an estimated incubation time of 20 days from the last larval (<8 mm) detection by the 20mm Survey. Putting the data together suggests that although the spawning window can vary, recent existing data shows a consistent threshold of <14 °C (see Figure 4.8).

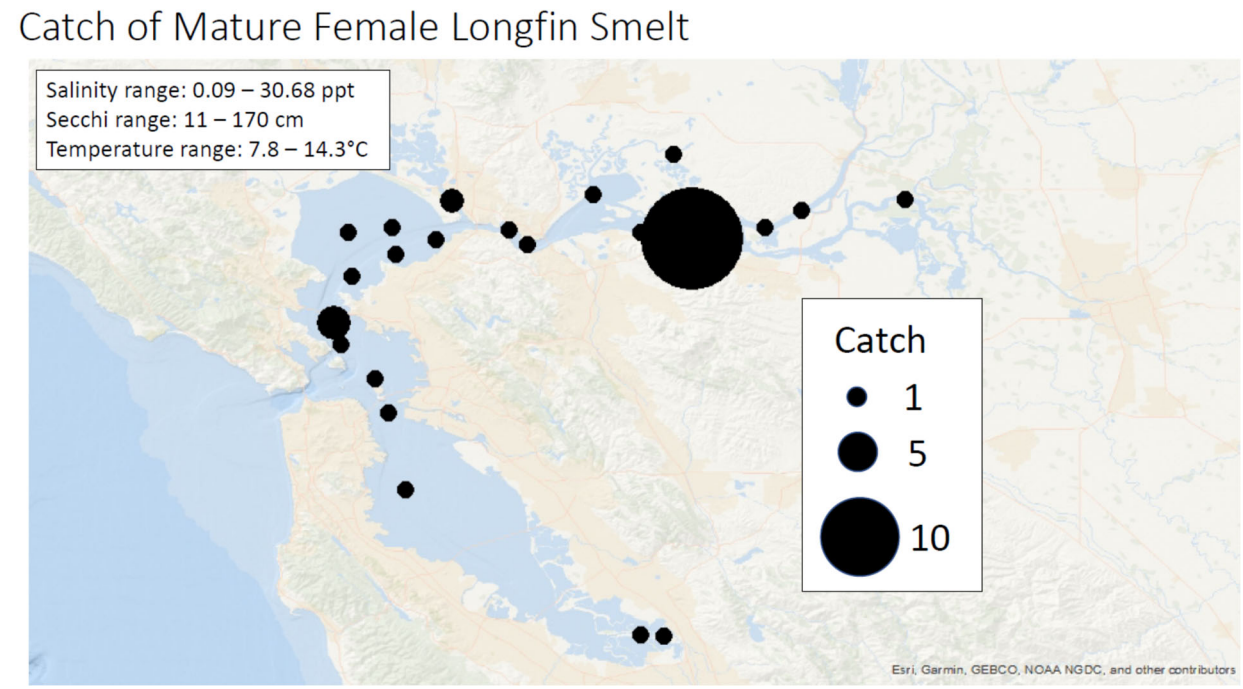


Figure 4.7: Map mature female longfin smelt catch from DFW's Longfin Smelt Fecundity Study. Source: Tempel and Burns 2021, slide 5

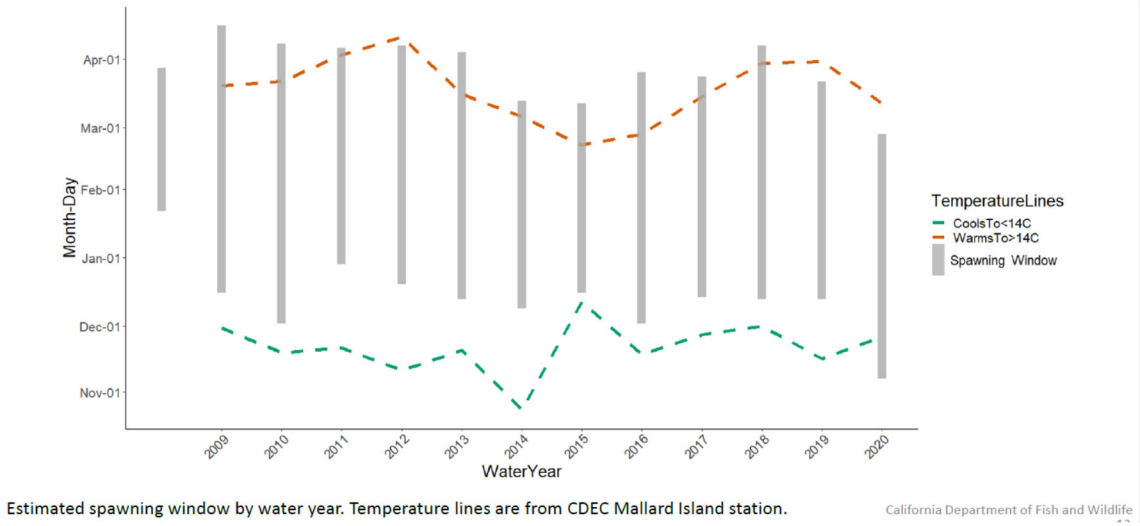


Figure 4.8: The calculated spawning window with temperature lines above (orange dashed lines) and below (green dashed lines) 14 °C. Source: Tempel and Burns 2021, slide 10

Since 2010, the annual average number of days below 14 °C has ranged 117-125 days. This suggests a typical spawning season lasting about 4 months. By 2050, under the RPC 8.5 scenario, it is projected that the spawning season in the upper estuary will decrease by 21-24 days (see Figure 4.9). By 2100, the spawning season is predicted to decrease by about 2 months, which would represent a major reduction in spawning opportunity (see Figure 4.10). The RCP 4.5 scenario projects smaller decreases of 10-11 days by 2050, to 19-24 days by 2100 (see Figure 4.11). Thus, there is high uncertainty about future spawning opportunity for the Bay-Delta DPS of longfin smelt that ranges from unhelpful from a species conservation perspective to potentially catastrophic.

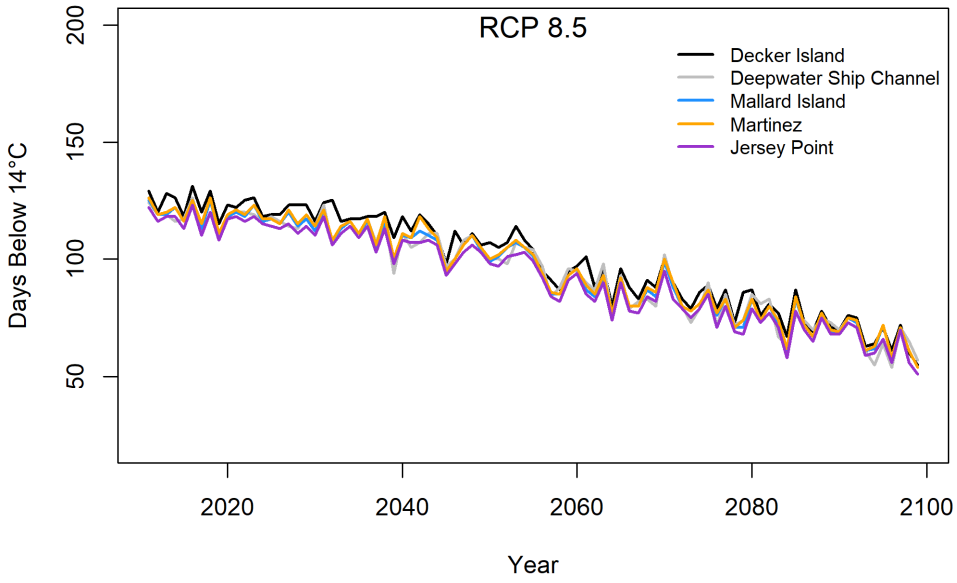


Figure 4.9: Projected total annual days below 14 °C at five locations based on the RCP 8.5 scenario.

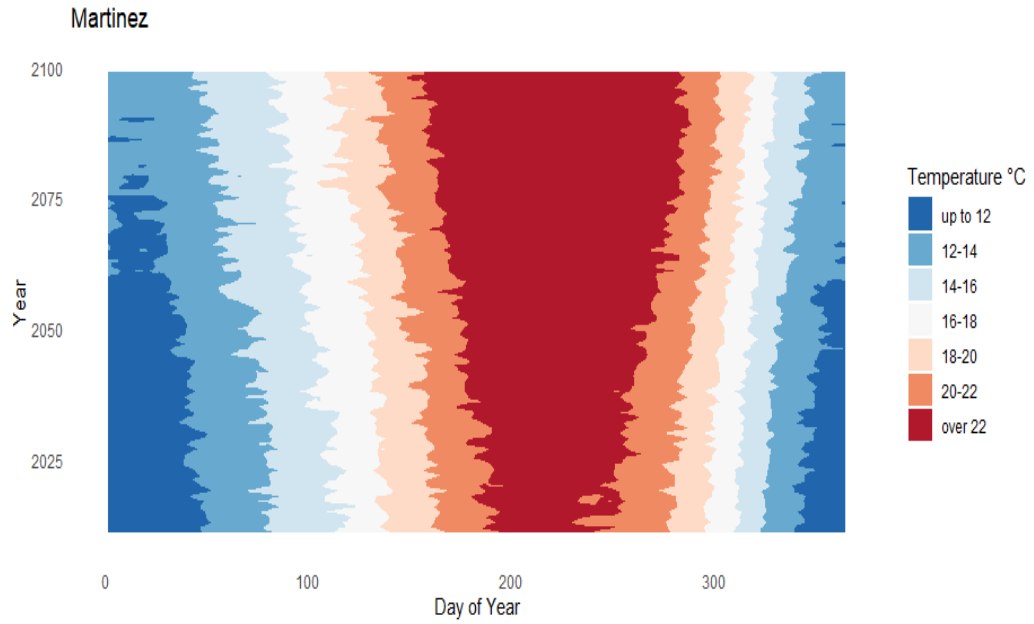


Figure 4.10: Heat map of temperature increases over time at Martinez in Suisun Bay from 2010-2100 under the RCP 8.5 scenario.

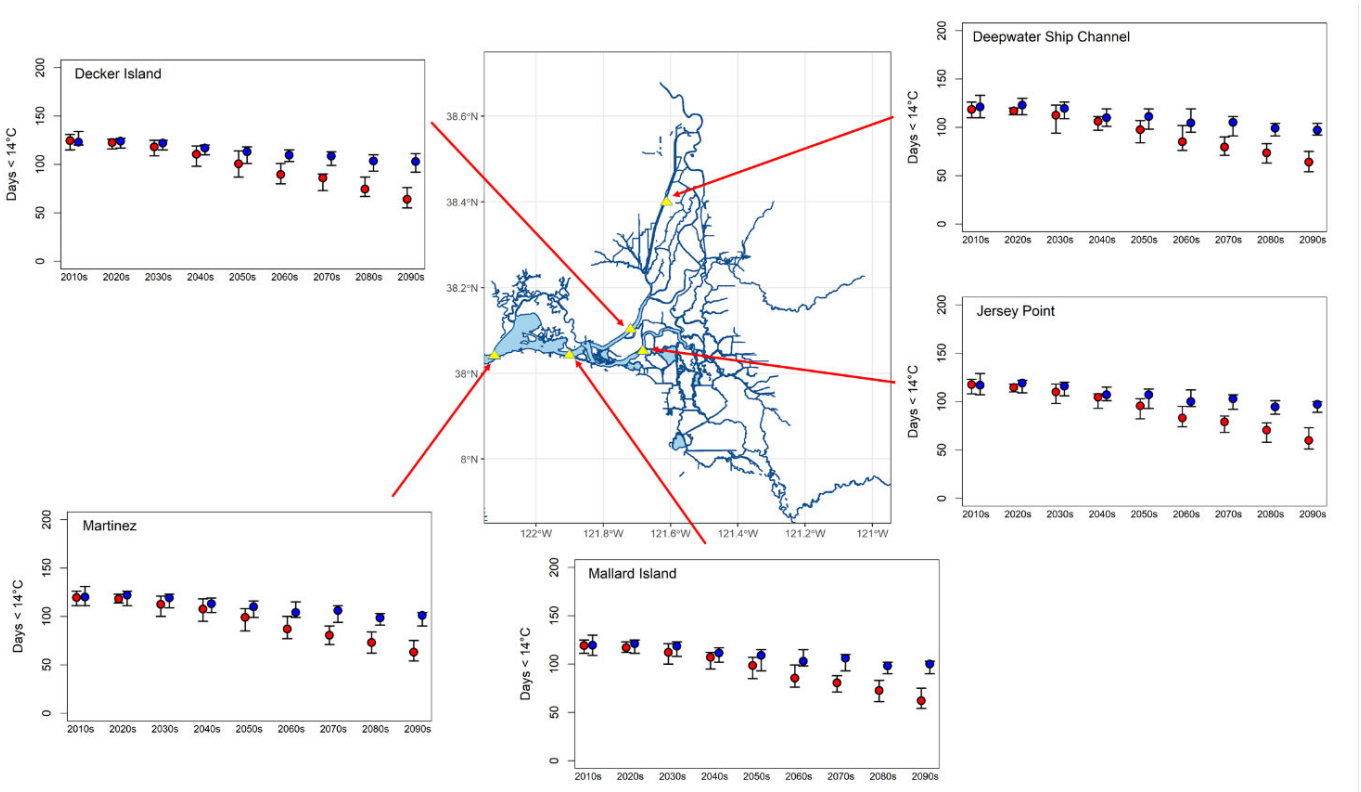


Figure 4.11: Map of the projected number of days below 14 °C at five locations under both the RCP 8.5 (red) and RCP 4.5 (blue) outputted scenarios. The red and blue dots represent the mean across all RCP

8.5 an RCP 4.5 projections, respectively. Lines extend between the min and max average projections for all 10 GCMs). Map source: Bashevkin and Barros 2021.

As described under the Species Needs, longfin smelt larvae development success is highly temperature dependent, and the highest probability of success occurs at temperatures of 9-12 °C. With extended days and periods outside of this favorable range, it is expected that lower fractions of spawned eggs will hatch into free-swimming larvae, and that yolk-sac larvae may experience slower growth and lower survival (Yanagitsuru et al. 2021, pp. 5–6). Compared to juveniles and adults, longfin smelt embryos and yolk-sac larvae are likely the most vulnerable to rising temperatures as they have the lowest tolerance to elevated temperatures (Baxter *et al.* 2010, p. 68; Jeffries *et al.* 2016, p. 1709; Yanagitsuru *et al.* 2021, pp. 5–6). They also do not yet possess the physiological capability to survive in cooler, higher salinity waters such as those found in the Bay and ocean (Yanagitsuru *et al.* 2022, Table 1, p. 5).

In summary, climate change is expected to significantly raise water temperature in the upper estuary. Temperature changes are projected to be most prominent in Suisun Bay, where the number of days above 22 °C (upper limit for all post-larval life stage) could more than double, forcing age-0 longfin smelt to trade estuary rearing time for ocean rearing time if they can do so successfully. There is high uncertainty about future spawning opportunity for the Bay-Delta DPS of longfin smelt that ranges from unhelpful from a species conservation perspective to potentially catastrophic. As a cold-water species at the southernmost portion of its range, the Bay-Delta DPS is likely already needing to avoid water temperatures above its physiological tolerance during the summer months. Further increases in water temperatures could mean extended summer-like conditions in which Suisun Bay and the Delta would be inhospitable for the fish. Longfin smelt are already exhibiting a pattern of exiting the Delta earlier in the summer, and this pattern may accelerate as temperatures rise. It is not known if it is a benefit or a risk for longfin smelt to extend ocean residence; historically many individuals stayed in the estuary in the summer. Likely due to rising temperatures, juvenile and subadult longfin smelt appear to be returning to the estuary later in the autumn, and this shift in estuary occupancy could shift further into the late fall or winter as temperatures continue to rise, potentially impacting spawning opportunities, e.g., leading to a constricted spawning temperature window. A shortened spawning window could hamper spawning success and decrease overall fecundity in the population. Lastly, larval longfin smelt may experience increased heat stress with highly uncertain, but potentially catastrophic population consequences.

4.2.3. Flow scenarios future condition analysis

As discussed under the Species Needs section, freshwater flow is a primary driver of longfin smelt production. As mentioned in the Current Condition chapter, reduced freshwater flow has been a substantial and increasing stressor on the species, likely for many decades. Here we predict how the species may respond to projected further changes of freshwater flow using downscaled global climate models for the Sacramento/San Joaquin watershed. The projections and analyses are based on information published by Knowles *et al.* 2018.

USGS Computational Assessments of Scenarios of Change for the Delta Ecosystem project (CASCaDE2) is an interdisciplinary modeling research effort aimed at projecting responses of the Delta to scenarios of future change. The goals of CASCaDE2 are to provide an improved understanding of, and a new capability to model, the effects of future climate and infrastructure changes on the Delta ecosystem. As part of CASCaDE2, USGS developed an analysis of projections of managed flows in the Sacramento/San Joaquin watershed under future climate change scenarios.

The CASCaDE2 project used outputs from 10 GCMs to produce 20 Coupled Model Intercomparison Project Phase 5 (CMIP5) climate change scenarios. Each GCM was run with 2 scenarios of future GHG concentrations, or representative concentration pathways (RCPs). The two RCPs valuated correspond to radiative forcing values in 2100 of 4.5 and 8.5 W/m2 (RCP 4.5 and RCP 8.5), which represent scenarios with an effort at abating global GHG emissions and a high-end emissions scenario, respectively. These 10 GCMs were selected from a larger ensemble of 31 CMIP5 GCMs as being particularly suitable for California water resources assessments, based on the realism of their simulated climate both globally and in the region of interest. For complete details of the model development, see Knowles et al. (2018 pp. 7634–7638).

The GCM outputs were used to drive models of the Bay-Delta watershed's hydrology and operations. These models produced flow, storage, and meteorological time series reflective of the Sacramento and San Joaquin basins. Water Year (WY) averages were calculated based on trends in basin-averaged precipitation and air temperature. Here, we use WY projections as the primary variable to describe the effect of changing flow conditions on longfin smelt production. We use RCP 4.5 as a more conservative projection of climate change with stricter global emissions standards, and RCP 8.5 as a high-end emissions scenario.

An important caveat for Knowles *et al.* (2018) is that the authors assumed no major changes to management infrastructure or strategy due to the difficulty of anticipating the details of such changes. Projecting freshwater demands in a complex water management network like California's is difficult. Another caveat of the data projections noted from Knowles *et al.* (2018) is that estimated demand scenarios were only available for California's level of development (LOD) in 2005 and for 2030. In CASCaDE2, all modeling runs use the 2005 LOD until the beginning of WY2030, after which they use the 2030 LOD. Projections past 2030 may therefore represent an inaccurate estimate of freshwater demand due to uncertainty in overall demand. Urban water demands are projected to increase from around 8 MAF (million-acre-feet) per year in 2030 to 11-13 MAF/year by 2100 (Christian-Smith *et al.* 2012, p. 42), which is roughly a 50% increase due largely to population growth. On the other hand, agricultural water demand is projected to decrease due to changes in cropland (conversion to urbanization), resulting in a decline of approximately 29 MAF/year in 2012 to 27 MAF/year in 2062 (Wilson *et al.* 2016, p. 8), which is a 7.8% decrease.

Averaged across all scenarios, the total Sacramento basin unimpaired runoff is projected to increase by an average of approximately 9.7% over a wide spread of variability by 2099 (Knowles *et al.* 2018, p. 7641). Historically from 1980-2009, the mean daily hydrograph of actual impaired Delta inflow displayed

rising inflows from mid-December, peaking in February-March, before leveling off in July (see Figure 4.11a). In comparison, the future scenarios project a general tendency leaning towards increased Delta inflow from December-March, followed by a reduction in inflow from April-July (see Figure 4.11b). This is a result of increased projected precipitation falling as rain and reduced snowpack due to rising temperatures that would likely cause higher peak flows in the winter followed by a steep drop off in the spring through mid-summer. Although a similar monthly pattern is seen between the RCP 4.5 and CP 8.5 scenarios, RCP 8.5 projects much higher peak inflows followed by a steeper drop off in April. Historically, approximately 60-65% of unimpaired flow contributions from the Sacramento basin occurred from October 1st to March 31st, but averaged across all scenarios, that percentage is projected to increase to approximately 83%.

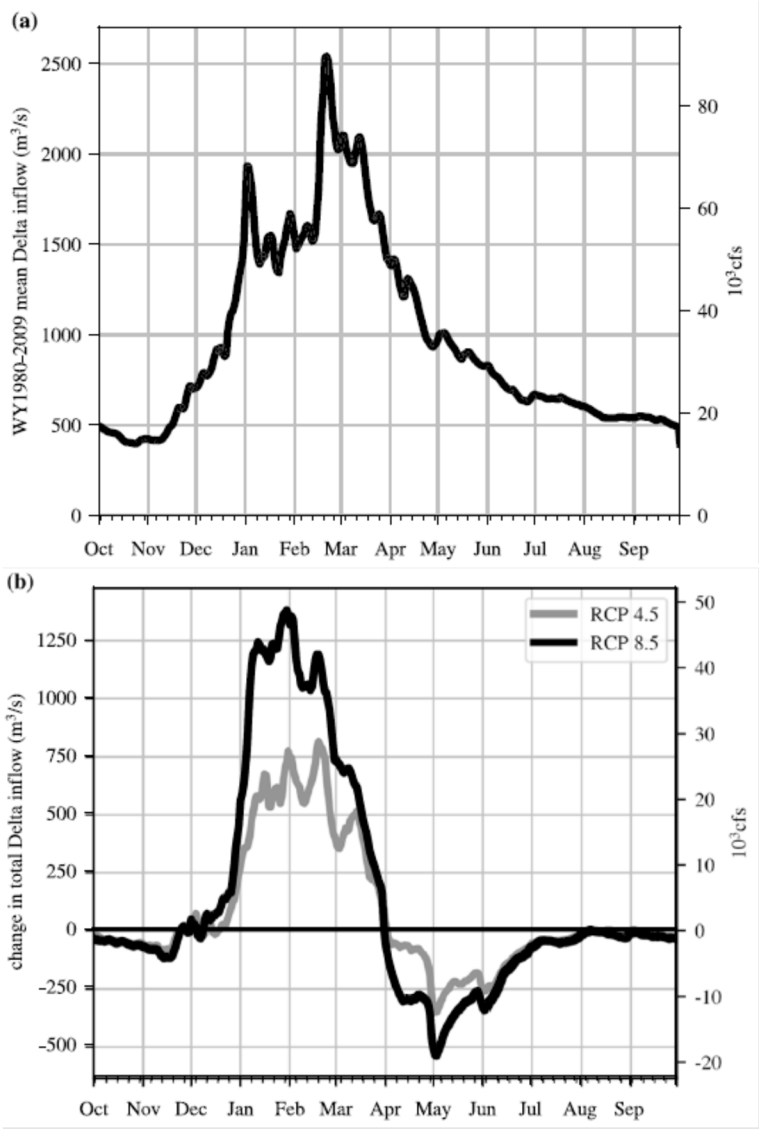


Figure 4.11: (a) The mean annual cycle of Delta inflow for WY1980-2009. The typical WY consists of rising inflow beginning in mid-December, peaking in February-March, before leveling off in July. (b)

Difference between WY2070-2099 and the 1980-2009 mean annual cycles of managed total Delta inflow averaged over scenarios of each RCP. Source: Knowles *et al.* 2018, p. 7646.

Historically, spring-summer Delta inflow is influenced by snowmelt and reservoir operations, with a gradual decline lasting into the fall until seasonal wet weather returns. Snowmelt runoff has accounted for an average of 40% of annual inflow, as indicated by discharge occurring after April 1st (Knowles and Cayan 2004, p. 320). But with rising temperatures under both scenarios, more snowfall would become rainfall, and the snowpack that does accumulate would melt earlier, potentially resulting in higher frequency extreme flow events during the winter and early spring. By 2099, the total annual snowpack on April 1 is projected to decline by 89% for the Sacramento Basin and 50% in the San Joaquin Basin compared to 1995 levels (see Figure 4.12). Due mainly to the overall reduced snowpack, projected inflow could face a steep drop during the month of April continuing through the summer (Fig. 4.11). Even in very wet years, this could result in very dry conditions in the late spring and early summer (Knowles *et al.* 2018, p. 7647).

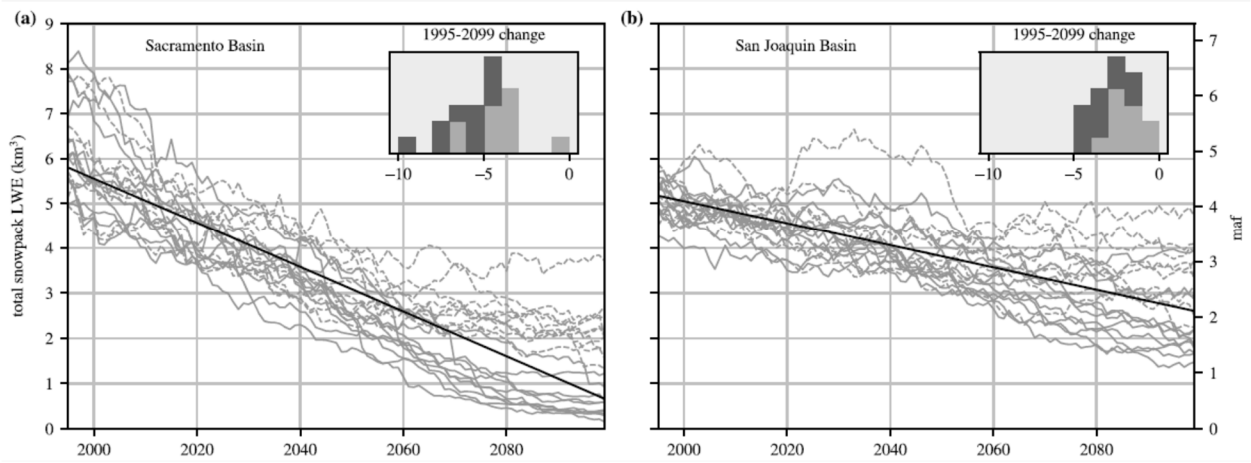


Figure 4.12: Smoothed scenario time series (gray), ensemble regression fit (black), and change histograms (inset) for total April 1st snowpack LWE volume for (a) Sacramento basin and (b) San Joaquin basin. Both panels have a common *y* axis, with units of cubic kilometer. Representative concentration pathway 4.5 scenarios are dashed lines in the plot and light gray in the histograms; representative concentration pathway 8.5 are solid and dark gray. LWE = liquid water equivalent. Source: Knowles *et al.* 2018, p. 7641.

As mentioned in the Species Needs section, freshwater flow is a primary driver of longfin smelt productivity, potentially throughout winter and spring (see Fig. 2.8). It is unknown if certain months of freshwater flow are more important than others. In the projected scenarios, the frequency of higher favorable flows during the January-March time periods is anticipated to increase (Fig. 4.11). However, the frequency of favorable flows during the April-June time period is anticipated to decrease. Thus, we anticipate that longfin smelt recruitment would likely be hindered as a result of the lower outflow later in the reproductive and early life rearing season.

The projected annual increases in flow during winter may be beneficial for longfin smelt spawning and egg incubation/larval rearing for as long as water temperatures remain suitable, as it may create more suitable habitat while extending the species range to occupy and utilize during this critical phase in the DPS life history. However, if the water temperatures increase as projected under Wulff *et al.* 2021, then even the annual winter inflow increases may not result in more spawning and larval rearing habitat. On the other hand, the projected lower spring flows could create less suitable conditions for larval rearing in many areas that had been conducive earlier in the season. Longfin smelt larvae have been detected at salinity up to 12 psu (Grimaldo *et al.* 2017, p. 8), though peak catch levels occur between 2-4 psu. If spring outflows decline as projected, larval productivity will likely continue to decline as well due to less accumulation of recruitment mechanisms as described in section 3.1.1. Thus, on the whole, it is uncertain whether anticipated improvement in winter freshwater flow conditions can compensate for anticipated decline of freshwater flow during spring.

In addition to a changing average seasonal hydrograph, the projected scenarios displayed changes in the frequency of Water Year (WY) types (Knowles *et al.* 2018, data release). Knowles *et al.* 2018 defined the most intensely dry years as falling between the 3rd and 10th percentiles of historical hydrology. Under the RCP 4.5 scenarios, intensely dry years increase in frequency over moving 31-year window projections (see Figure 4.13). For the RCP 8.5 scenarios, all ranges of dry years increased in frequency over the projected century, with the most intensely dry years experiencing the largest relative increase such that the 3rd and 10th percentile projected 3 and 2-fold increases by the end of the century, respectively.

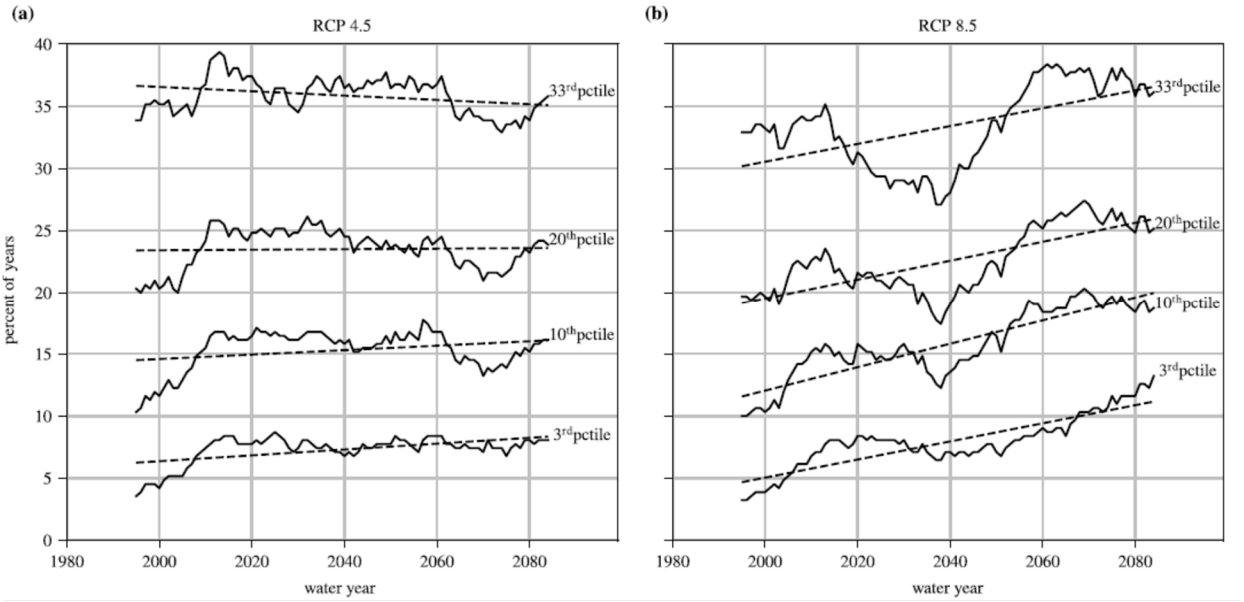


Figure 4.13: The 31-year moving-window percentages of years with projected impaired Sacramento basin WY outflow below global climate model historical-period (WY1980–2009) percentile values, averaged over (a) RCP 4.5 scenarios and (b) RCP 8.5 scenarios. Percentile thresholds are indicated (i.e. the 33rd percentile represent the historical 33% driest years on record, the 20th percentile the 20% driest years on record, etc.). Under the RCP 8.5 scenarios, all ranges of dry years increase in frequency. Under

the 4.5 scenarios, only the intensely dry years (10th and 3rd percentile) increase in frequency. Dashed lines are linear fits. RCP = representative concentration pathway; WY = October 1 – September 30 water year. Source: Knowles *et al.* 2018 p. 7646.

As mentioned under the species Current Condition, a primary cause for the longfin smelt decline over time has been the reduction in freshwater flows, which we have concluded in this SSA is the most important species need. As dry years have generally been "bad years" for longfin smelt recruitment, the frequency of dry years and droughts has already taken a substantial toll on longfin smelt such that abundance indices have frequently sunk to new record lows, particularly over the past two decades. The projected increases in dry year frequency under RCP 8.5 and intense dry year frequency under RCP 4.5 are anticipated to increase the intensity of decline in this already steeply declining DPS.

The 20th Century was a relatively wet one in the past millennium (Cook *et al.* 2009, pp. 51–52; Stine 1994, entire), with one of the lowest frequencies of droughts occurring between 1850-1950 (Hughes and Brown 1992, p. 166). Prior to the 21st century, dry and critically dry years occurred approximately 33% of the time. But since 2000, dry and critically dry year frequency has been 43% of years. Based on soil moisture reconstruction, the observed timeframe of 2000-2021 was probably the driest 22-year period on record in at least the past 1,200 years, exceeding even any 22-year period during the last megadrought in the late 1500s (Williams *et al.* 2022, p. 1). In essence, California's complex water infrastructure was likely constructed during a wetter era than what has been typical for the State (Fountain 2015; see Figure 4.14). As the demand for water further increases with population growth (Wilson *et al.* 2016, pp. 7–8), water withdrawals are estimated to be greater than 100% of precipitation by 2050 (Roy *et al.* 2012, p. 2554). And by 2062, water use is projected to increase by 1.8 billion cubic meters over 2015 use rates (Wilson *et al.* 2016, p. 6), so what was considered sufficient water supply in the 20th Century is unlikely to be enough to meet demand for the 21st Century (Stanton and Fitzgerald 2011, pp. 24–28). If the projected increases in frequency of dry years under either the RCP 4.5 and 8.5 scenarios come to pass, longfin smelt productivity is anticipated to keep declining over time as less water and superimposed water demand further exacerbates flow reductions unless alternative ways are found to mitigate for past, present, and future changes to the estuary hydrograph.

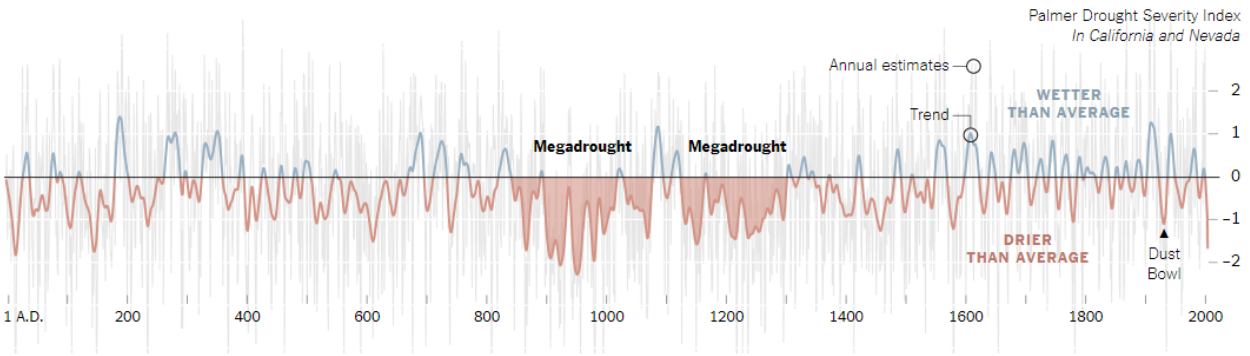


Figure 4.14: Analysis of tree rings suggests that western states have had many droughts of two decades

or longer, including two megadroughts lasting longer than 100 years. However, the 20th Century was wetter than average. Source: Fountain 2015.

Longfin smelt's resiliency against droughts in the SFE watershed appears to have decreased over time and the species has always displayed low resistance to drought, defined as the lack of a large-scale decline in occurrence from a wet period to a drought period (Mahardja *et al.* 2021, pp. 6 and 9). Longfin smelt have a predominantly 2-year life span though some experts believe that substantial numbers of individuals may spawn at age 1 and 3. We mentioned above resolving the age distribution of spawners is an important conservation science topic because more spawning age-classes translates into more resilience to brief periods of poor environmental conditions. Nonetheless, consecutive dry years, which have occurred more frequently over the past 2 decades, limit the effectiveness of this life history strategy. Using the projected scenarios, we estimated the probability that consecutive dry years will occur in the future (see Figure 4.15).

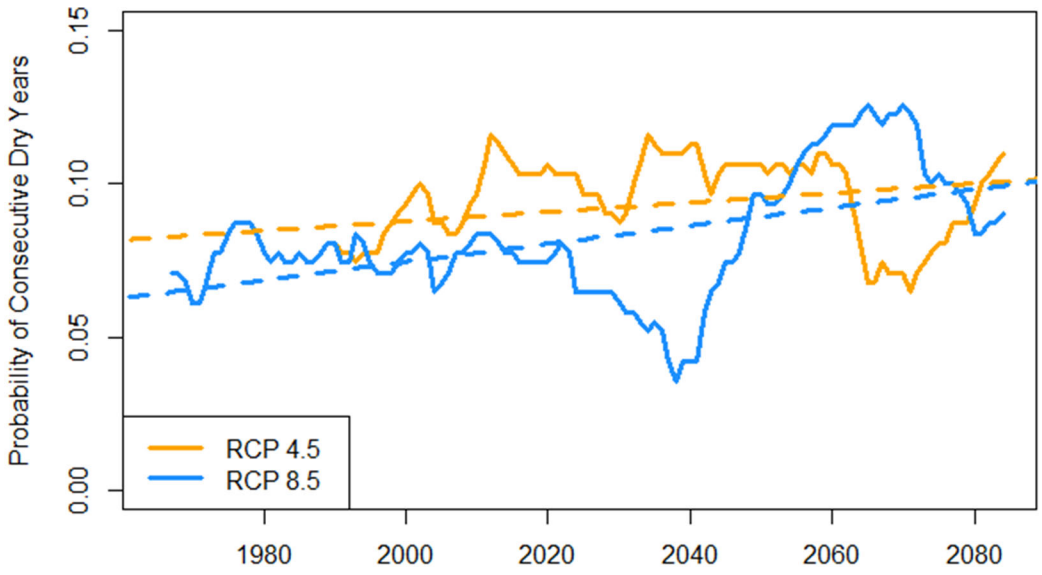


Figure 4.15: The probability of consecutive (2 or more) very dry years within a 31-year moving window (centered on the year indicated on the x-axis) occurring under the RCP 4.5 and RCP 8.5 scenarios. Very dry years are defined as dry and critically dry water year types. Dashed lines represent increasing trends over time (p-values for the slopes of both trend lines were < 0.001).

Droughts and very dry years have become more prevalent over the past two decades, and this trend is likely to increase in the future. Over time, the probability of consecutive dry years significantly increases under both the RCP 4.5 and the RCP 8.5 scenarios. This change is anticipated to steepen the rate of longfin smelt decline unless alternative ways are found to mitigate for past, present, and future changes to the estuary hydrograph.

4.2.4. Sea level rise future condition analysis

Global sea level rise (SLR) is primarily caused by thermal expansion of warming ocean water and melting of land-bound ice as air temperatures increase. Recent research has focused on potential large-scale ice sheet collapse in West Antarctica (where the Ross Ice Shelf is land bound) and Greenland (Mouginot *et al*. 2019), which would rapidly raise ocean levels worldwide (DeConto and Pollard 2016, pp. 5–6; Oppenheimer and Alley 2016, p. 1376). Regional rates of sea level rise are variable, depending on vertical land motion (i.e., uplift and subsidence), winds, and large-scale ocean circulation patterns.

Since its installation in the mid-1850s, the San Francisco tide station, located near the Golden Gate Bridge at the mouth of San Francisco Bay, has recorded an 8-inch increase in local sea levels, with other tide stations in the Bay Area showing comparable rates of sea level rise. Analysis of satellite altimetry data also indicates a recent acceleration of global sea level rise since 2011 (Nerem *et al.* 2018, pp. 2023–2024). This recent acceleration follows decades of suppressed sea level rise along the west coast of the United States (relative to rates of sea level rise elsewhere in the Pacific Ocean basin), which may be linked to variations in the Pacific Decadal Oscillation (Bromirski *et al.* 2011, pp. 11–12). As with any prediction of the future, it is unclear how long this recent accelerated trend of sea level rise will continue, but it is expected to continue for the foreseeable future (Ruckert *et al.* 2017, Fig. 1, p. 3).

In our sea level rise analysis, we utilize existing data and summaries from the Delta Stewardship Council's Delta Adapts document (DSC 2021), which consists of climate change vulnerability assessments and adaptation plans specific for the Bay-Delta developed in partnership with technical experts, state and local governments, stakeholders, and community representatives. Our analysis focuses on the upper estuary. This focus is helpful in that higher estuary salinity caused by higher sea level, more frequent drought conditions, and higher human water demand (all reviewed above) will compel a larger average fraction of longfin smelt to spawn and rear in and near the Delta. We note that by year 2100 higher end estimates of sea level rise have potential negative implications for tidal marsh restoration efforts throughout the estuary (Buffington *et al.* 2021, Fig. 4, p. 14). Sea level trends in the Delta may be complicated by land subsidence. Decomposition of drained and converted marsh and peat soils within diked Delta islands have caused much of the Delta region to lie below sea level—in some places by as much as 15 to 20 feet (Deverel *et al.* 2020, p. 839). Continued land subsidence may increase the relative rate of locally observed sea level change for the Delta area when comparing water levels to local land elevations.

The State of California Ocean Protection Council (OPC 2018, p. 18) has created sea level rise projections for the San Francisco tide station (Table 4.1). The San Francisco tide station, although not located within the Delta, provides an ocean boundary that is considered representative of regional oceanic sea level conditions that will influence local sea level rise and peak water level response in the Delta. Based on the OPC Guidance, sea levels in the San Francisco Bay-Delta Estuary are likely (66 percent probability) to rise between 0.6 to 1.1 feet by 2050, with an upper range (1-in-200 chance) projection of 1.9 feet. By 2100, sea levels are likely to rise between 1.2 to 3.4 feet, with an upper range projection of 6.9 feet.

However, the combination of extreme rates of ice-sheet loss and complex feedback mechanisms could result in accumulated sea level rise of up to 10.2 feet by the end of the century (OPC 2018, p. 25).

Table 4.1: Sea level rise projections for the San Francisco Bay-Delta. Probabilistic projections are based on the Kopp *et al.* 2014 (pp. 388–393) method and shown in feet. Projections are shown for the San Francisco tide station. Source: DSC 2021 p. 51

		Median ***50% probability sea level rise meets or exceeds***	**Likely Range** ***66% probability sea level rise is between***	**1-in-20 chance** ***5% probability sea level rise meets or exceeds***	**1-in-200 chance** ***0.5% probability sea level rise meets or exceeds***	**H++ Scenario** ***Extreme scenario not associated with a probability***
Emission Scenario	**Year**	**N/A**	**Low-risk aversion**	**N/A**	**Medium-high risk aversion**	**Extreme risk aversion**
RCP 8.5	2030	0.4	0.3-0.5	0.6	0.8	1.0
RCP 8.5	2050	0.9	0.6-1.1	1.4	1.9	2.7
RCP 4.5	2070	1.3	0.8-1.7	2.1	3.2	5.2
RCP 8.5	2070	1.4	1.0-1.9	2.3	3.5	5.2
RCP 4.5	2100	1.8	1.2-2.7	3.5	5.8	10.2
RCP 8.5	2100	2.5	1.7-3.4	4.4	6.9	10.2

As with our Delta hydrology analyses, we used RCP 4.5 as the conservative scenario with stricter emissions standards and RCP 8.5 as the "business-as-usual" scenario with fewer global efforts to limit or reduce emissions. However, the State of California has recommended using RCP 8.5 through 2050 because the greenhouse gas emissions worldwide have continued to follow the business-as-usual trajectory (OPC 2018, p. 13). Based on this projection, an increase in sea level of 0.6 to 1.1 feet is likely by 2050, with an upper range estimate of 1.9 feet. Beyond 2050, sea level rise depends partly on emissions over the coming decades and the planet's response to a warmer climate. By 2070, an increase in sea level of 0.8-1.9 feet is likely, with an upper range estimate of 3.5 feet. Under a moderate emissions scenario, an increase in sea level of 1.2 to 2.7 feet is likely by 2100, with an upper range estimate of 5.8 feet. Under a high emissions scenario, an increase in sea level of 1.7 to 3.4 feet is likely, with an upper range estimate of 6.9 feet.

Sea level rise in the San Francisco Bay is likely to affect daily tide and peak storm water levels throughout the Delta (DSC 2021, pp. 5–4). Daily tide levels will respond to sea level rise differently across the Delta and Suisun Marsh, depending on the amount of sea level rise, proximity to the Bay, and local hydrodynamic conditions. In some parts of the Delta, high tide elevations are projected to increase at a faster rate than mean or low tide elevations. This suggests that the tide range in the Delta may also increase as a result of sea level rise. For one foot of sea level rise, this effect is most pronounced in the south Delta, where the tide range is projected to increase by more than 20 percent. The tide range

amplification is progressively less in the north Delta (approximately 10 to 15 percent) and central Delta (approximately 5 percent) and negligible in strongly tidally influenced areas such as Suisun Bay, Rio Vista, and the lower Yolo Bypass (DSC 2021, pp. 5–4).

These changes in mean sea level and tidal dynamics will likely affect marsh habitats in the Delta and Suisun Marsh used by longfin smelt. For tidal freshwater and brackish wetlands in the Delta and Suisun Marsh, the Delta Stewardship Council (2021) has defined vulnerability as either the transition from high marsh to low marsh or complete drowning in response to sea level rise. For mid-century (2050) sea level rise scenarios, high marsh tidal wetlands will not be at risk of habitat transitions or drowning under 1 foot of sea level rise (Table 4.2). Under 2 feet of sea level rise, tidal freshwater wetlands in the Delta will be at risk of transitioning to low marsh, but brackish tidal wetlands in Suisun Marsh are predicted to persist and keep pace with sea level rise. If sea level rise, shorter wet seasons, and more frequent drought interact to increase estuarine salinity, tidal wetland ecosystems may be subject to changes in plant community composition that further compromise their ability to keep pace with sea level rise. For late-century (2085) scenarios, tidal wetlands will not be at risk of transitioning to low marsh under two feet of sea level rise but will be at risk of transitioning under 3.5 feet of sea level rise. Under the more extreme end-of-century scenario (6 feet sea level rise by 2100), all tidal wetlands in the Delta and Suisun Marsh are at risk of drowning (becoming subtidal habitats).

Table 4.2: Predicted Habitat Changes of Un-leveed Freshwater and Brackish High/Mid Marsh in the Delta and Suisun Marsh under Different Sea Level Rise Scenarios. Source: DSC 2021, p. 147.

Year	Sea Level Rise	Delta Freshwater Marsh	Suisun Brackish Marsh
2050 (low)	1 foot	High/Mid Marsh Persists	High/Mid Marsh Persists
2050 (high)	2 feet	Conversion to Low Marsh	High/Mid Marsh Persists
2085 (low)	2 feet	High/Mid Marsh Persists	High/Mid Marsh Persists
2085 (high)	3.5 feet	Conversion to Low Marsh	Conversion to Low Marsh
2100	6 feet	Drowned	Drowned

Both the freshwater marshes, Suisun's brackish marshes and bay tributary marshes are important part of young-of-the-year longfin smelt's low-salinity rearing habitat (Grimaldo *et al.* 2017, p. 11; Hobbs *et al.* 2010, p. 565, Lewis et al. 2019, p .63). The same can be true for similar habitats further seaward when low-salinity conditions are available (Fig. 2.4). Field surveys including SLS and the 20mm Survey have historically also detected an abundance of larval and juvenile longfin smelt adjacent to these areas, suggesting that lateral connectivity between marshes and open-water habitats may be an important aspect of nursery habitat for the species. Thus, conversions of freshwater and brackish marshes to low marshes and subtidal habitats may conspire with higher salinity, warmer water temperatures, and a shorter duration wet season to affect habitat suitability for young longfin smelt.

MacWilliams and Gross (2010) evaluated salinity intrusion over one annual hydrologic cycle under five levels of sea level rise between 15 cm and 140 cm based on Delta outflows and operations as they were

during 2002. Their model predicted that these increases in sea level would result in an increase in X2 throughout the year (see Figure 4.16), with predicted median increases ranging from 0.7 km for the 15-cm sea level rise scenario to more than 7 km for the 140-cm sea level rise scenario. By 2050, this would likely extend the position of X2 by over 1 km under a moderate scenario, and by nearly 3 km under a more extreme scenario.

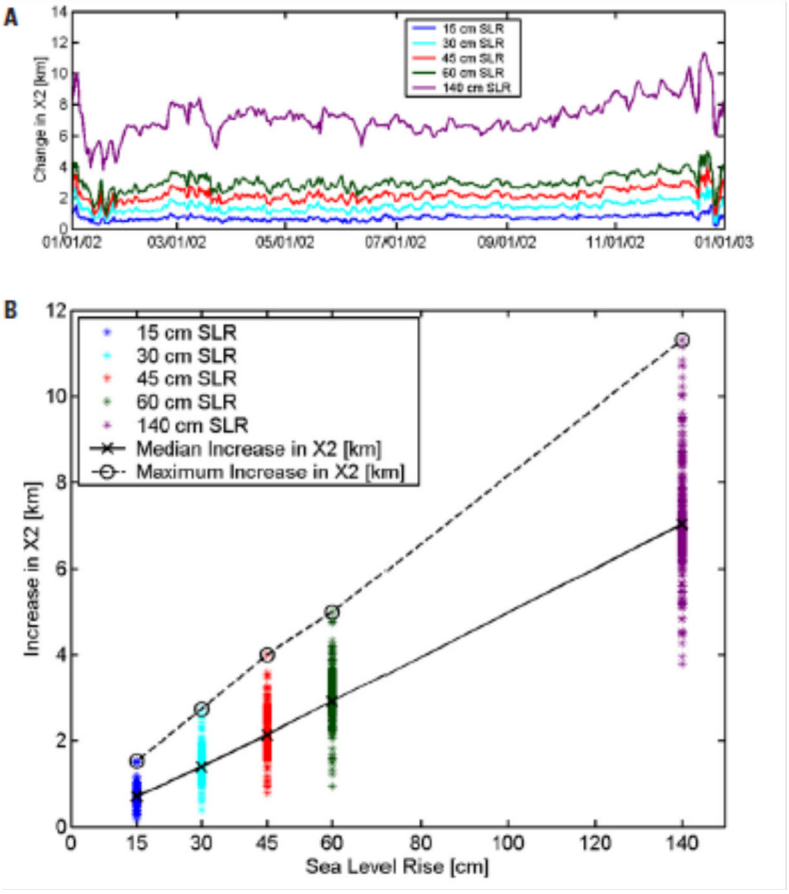


Figure 4.16: (A) Predicted change in X2 relative to Baseline scenario for 15-cm SLR scenario, 30-cm SLR scenario, 45-cm SLR scenario, 60-cm SLR scenario, and 140-cm SLR scenario; (B) Scatter plot of the predicted increase in X2 for each day during 2002 for each of the sea level rise scenarios; solid black line shows the median increase and the dashed black line shows the maximum increase in X2 for each SLR scenario. Modified from Mac Williams and Gross (2010, p. 185).

Chua and Xu (2014) estimated the effect of sea level rise on salinity intrusion, finding that at moderate steady-state flows (300 cubic meters per second), a 1.5-m sea level rise would result in a 10 km increase in X2 relative to present conditions. At higher steady-state flows (2000 cubic meters per second), the effect was less pronounced, with a similar sea level rise producing only a 4 km increase in X2. As it does presently, realized salinity intrusion resulting from sea level rise will vary depending on the natural and human-modulated availability of fresh water to repel it.

These projected changes in X2 could have a significant impact on longfin smelt spawning and rearing in the Delta and Suisun Bay (see Figure 4.17). As described in the species Life History section, longfin smelt returning adults are broadly distributed but many aggregate in the LSZ during the spawning season. If the location of X2 increases during the winter due to sea level rise and those average increases are not offset by the anticipated higher average winter flows, spawning habitat could be pushed substantially eastward, potentially resulting in constricted and lower quality spawning habitat. In addition, sea level rise may compound this effect during the spring when coupled with the projected inflow drop off. The projected eastward shift in X2 may also result in increased entrainment risk (Section 3.1.7). Further, if returning adults have longer spawning migrations, this may require additional energy expenditure or increase predation risk. However, as longfin smelt likely also use tidal currents to move upstream instead of continued directional swimming, it is unknown if extended geographic spawning movements would be a significant added stressor.

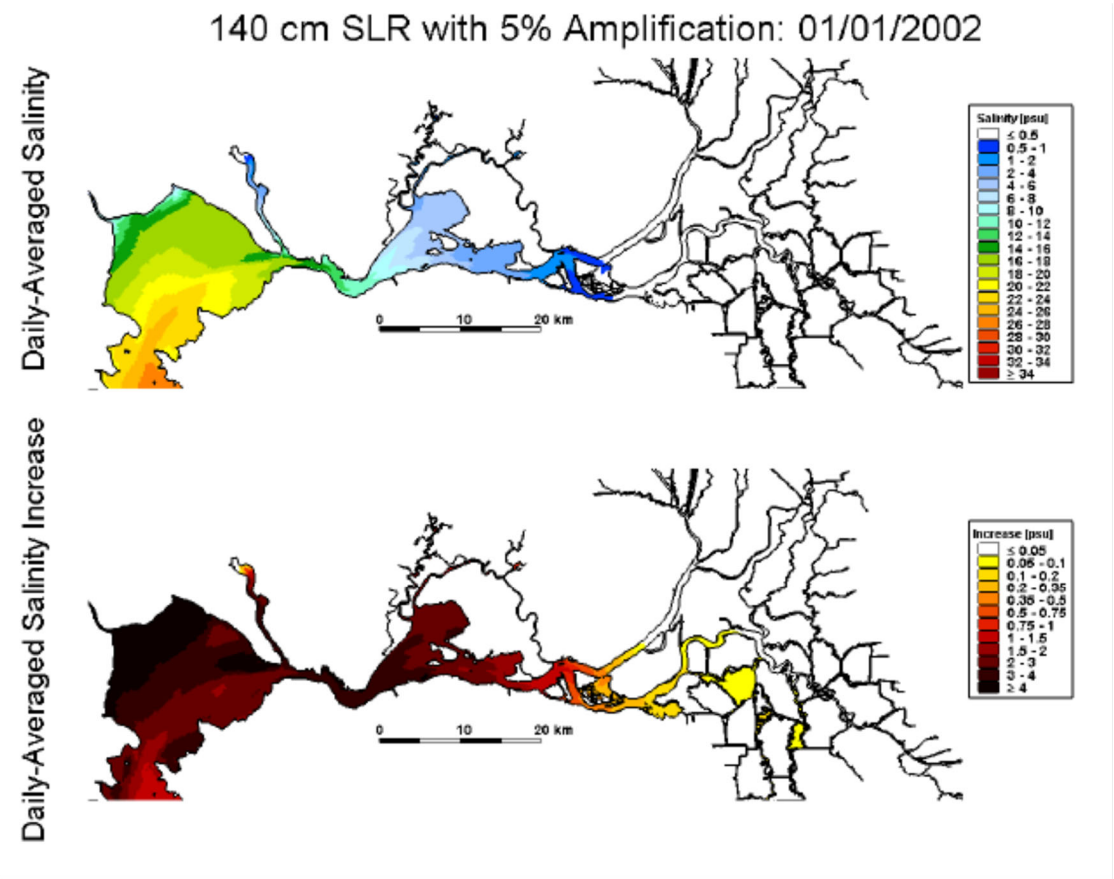


Figure 4.17: Predicted daily- and depth-averaged salinity on January 1, 2002 for the 140 cm SLR with 5% Amplification scenario (top); predicted increase in daily-averaged depth-average salinity on January 1, 2002 relative to the Baseline (0 cm SLR) scenario for the 140 cm SLR with 5% Amplification scenario. Source: MacWilliams and Gross 2010, p. 148.

In summary, the effects of sea level rise on the San Francisco Estuary could result in salt- and freshwater marsh losses and salinity intrusion. Many of the marshes currently used by longfin smelt could be

inundated and lost by the end of the century, potentially resulting in lower suitability of remaining open water habitats. The salinity intrusion, if not sufficiently abated by increasing reservoir releases and export reductions (thereby extending already stretched water supplies), would likely shift X2 eastward. The upstream shift in X2 could lengthen the spawning migration of adult longfin smelt, and the salinity increases could render San Pablo Bay and Suisun Bay more frequently inhospitable to larvae, substantially reducing suitable larval rearing habitat.

4.3. Future invasive species

The San Francisco Estuary has historically been one of the most invaded aquatic systems in the world (Cohen and Carlton 1998, p. 556). The rate of invasions increased from an average of one new species established every 55 weeks from 1851 to 1960, to an average of one species every 14 weeks from 1961 to 1995 (Cohen and Carlton 1998, p. 556). In the low-salinity zone, the accelerated invasion rate was linked to drought intensified by water diversion interacting with a ready source of colonizers in the ballast water of international shipping traffic (Winder *et al.* 2011, Fig. 2, p. 4). After the mid-1990s, the invasion rate slowed down likely due in part to new ballast water regulations (Winder *et al*. 2011, p. 7).

The overbite clam, which substantially impacted the food web (see Current Condition), was the most important ecosystem engineer introduced during the period of accelerated biological invasions of the estuary (Brown et al. 2016, pp. 8–9). Cohen and Carlton (1998, pp. 556–557) theorized that the Estuary's favorable transport vectors, depauperate and relatively young biota, and anthropogenic alterations and disturbances made the estuary especially vulnerable to invasions. Here, we consider the effect that existing invasive species will have moving forward. It is likely that additional nonnative species will be introduced to the estuary, but attempting to guess what those species will be, what order they will arrive, and what impacts they could have is far too speculative to be useful here.

There are few biomass-dominant native fish species left in the San Francisco Estuary. Some examples include marine forage fishes like northern anchovy and Pacific herring (Kimmerer 2006, p. 209; Grimaldo *et al.* 2020, p. 3). Another socially important example is Chinook salmon which only remain common due to the considerable subsidy provided by Central Valley fish hatcheries (Sturrock *et al*. 2018, entire). Currently, much of the biomass of the San Francisco Estuary is dominated by species that are not native to it (Matern *et al*. 2002, Fig. 2, p. 804; Nobriga *et al.* 2005, pp. 780–781; Cohen and Bollens 2008, p. 245; Kratina *et al*. 2014, entire). High biomass non-native species include the overbite clam, most of the relatively abundant freshwater and low-salinity epibenthic and planktonic crustaceans, striped bass, various species of gobies, Mississippi silverside, and American and threadfin shad. Nonnative fishes of the family Centrarchidae are also abundant in the legal Delta and its watershed (Mahardja *et al*. 2017, Fig. 2, p. 8).

The biomass dominant planktonic invertebrates of the estuary's low-salinity habitats include several species that invaded after the overbite clam decimated historically abundant taxa, suggesting that its grazing pressure opened niche space that could be filled by novel species that had some kind of life-history workaround to the clam's grazing pressure. The best studied example is the copepod

Pseudodiaptomus forbesi which is subject to intense grazing pressure in the low-salinity zone (Kimmerer *et al*. 2019, Fig. 6, p. 227). This copepod remains a seasonally dominant species however because its reproductive center is in the Delta upstream of immediate overbite clam grazing pressure (Kimmerer *et al*. 2019, p. 234).

Further downstream, trophic cascades in San Francisco Bay have also been reported to occur as likely effects of changes in community predation pressure on overbite clams (Cloern *et al*. 2007, Fig. 3 and elsewhere on p. 18563). These limits and oscillations of which species can persist given grazing pressure from overbite clams can be expected to continue into the future. The population-level grazing pressure exerted by overbite clams is tied to seasonally warming temperature (Kimmerer and Thompson 2014, p. 1206). Thus, a warming climate can be expected to result in an increase in the fraction of each year that overbite clam grazing will impact the food web that supports the Bay-Delta DPS of longfin smelt. It is reasonable to expect this will have negative consequences for longfin smelt if it cannot be mitigated.

We detailed elsewhere in this chapter how the future expected condition is a generally warmer and often drier estuary except for predictions of higher average winter flows and flood risks. The nonnative species that dominate the estuary's current fish fauna are for the most part species that are well-adapted to warm water. They are also species that begin their reproductive seasons later than the remnant native fish community (e.g., Meng and Matern 2001, Fig. 7, p. 760; Grimaldo *et al*. 2004, Fig. 6, p. 91). Most of these biomass-dominant nonnative fishes start spawning in the late spring and have reproductive seasons that continue into the summer or fall. This pattern is an indication that current ecological conditions better support a later reproductive season than the ancestral estuary did. Thus, it is logical to expect that generally warmer and drier conditions will further favor the nonnative species life histories and disfavor native species life histories where (e.g., longfin smelt) reproductive timing and early life are related to historical peak freshwater flow seasons. It is important to understand that increasing biomass dominance of warmwater nonnative fishes and higher frequency of poor recruitment years for longfin smelt are a likely outcome of a warmer and drier climate whether or not nonnative species have important direct interactions with longfin smelt (e.g., competition or predation).

4.4. Future DPS Viability Under Our Risk Profile

Climate change is likely to have a substantial impact on the San Francisco Estuary, and thus on longfin smelt. This impact is anticipated to accelerate into the future. As a cooler-water species, the Bay Delta DPS is already experiencing seasonal habitat conditions with water temperatures above its physiological tolerance, which may be exerting downward pressure on the DPS. Both RCP 4.5 and 8.5 global warming scenarios project significant water temperature increases in the Suisun Bay and the Delta, thus potentially rendering certain regions inhospitable for the fish not only during summer, but throughout the year. By 2090, it is projected that there will no longer be any areas below 12°C for larval rearing, which could result in extirpation. In addition, the spawning window, which likely occurs when water temperatures are below 14°C, could be shortened, thus reducing reproductive success.

Climate change is also projected to alter the San Francisco Estuary's hydrology under both the RCP 4.5 and 8.5 scenarios. Warmer winter weather, coupled with increased precipitation in the form of rain,

reduces snowpack which will likely result in more extreme flows from January-March, followed by a sharp cut off from April onward. Longfin smelt productivity could benefit from the favorable flows earlier in the season, but the steep drop-off beginning in April could induce dry year-like conditions, thus potentially rendering many of the formerly suitable habitats for larval rearing no longer hospitable. The current impaired April-June flows are already insufficient for longfin smelt productivity, and projected further declines will likely negatively impact the Bay-Delta DPS.

In addition to changes in hydrology, the proportions of WY types are projected to change in the future with a greater frequency of dry years under the RCP 8.5 scenario as well as critically dry years under both RCP 4.5 and 8.5 scenarios. In the past, longfin smelt exhibited poor survival and reproduction during droughts. If patterns in productivity that were observed in recent droughts are an indicator of patterns in future droughts, the projected increased frequency of consecutive dry years and drought under both scenarios would hamper species productivity. Furthermore, the DPS is not expected to exhibit resiliency to bounce back from droughts as they did historically (i.e., prior to the POD and the overbite clam invasion) because of their depleted abundance and food supply. Thus, the projected increased frequency of dry and critically dry years will likely cause further declines in the species productivity and increase the probability of extirpation.

Climate change is also expected to cause considerable sea level rise in the San Francisco Estuary under both the RCP 4.5 and 8.5 scenarios. The resulting sea level rise could lead to more open water and less marsh habitat for longfin smelt rearing. A further consequence of sea level rise is that if salinity intrusion moves the position of X2 eastward, it would reduce spawning and rearing habitat in the Delta. Although salinity intrusion can be mitigated through management actions such as reservoir storage releases, the system would likely continue to be vulnerable during dry years and particularly drought. Further, increased water demand during all year types would functionally further reduce flows in the system. Thus, the projected increased frequency of dry years, coupled with sea level rise under both scenarios, could be catastrophic for the Bay-Delta DPS in light of the relationship between abundance and flow as well as the need to spawn and rear in fresher water environments.

The San Francisco Estuary is one the world's most invaded estuaries, and invasions have increased over time. Invasive species have rarely had a positive impact on the Bay-Delta DPS. The first step decline in abundance was likely caused by the exotic overbite clam, which decimated the food web. Other exotic species have induced additional predation pressure and competition for the population. The San Francisco Estuary will likely experience further invasions in the future, some of which may negatively affect longfin smelt. An invasion into the San Francisco Estuary could also have a similar effect on the already impaired food web.

In Table 4.4, following, we summarize how we anticipate future environmental conditions will impact the risk profile to the Bay-Delta DPS under our scenario planning.

Table 4.4: Future environmental factors and their potential to change the risk profile for the Bay-Delta DPS. Color coding signifies: Red = A significant increase in the magnitude and/or frequency this factor will reduce survival, growth, or reproduction; Yellow = An appreciable increase in risk from this driver, and/or a less certain magnitude of influence/mechanism of action; Green = A significant decrease in risk (benefit to the DPS); Gray = An uncertain effect.

Factor	Effect	Potential mechanisms of response(s)	Changes to Magnitude of Risk
Climate Change: SFE warming	Fewer days <12 °C	Loss of rearing habitat	Substantial increase
	Fewer days <14 °C	Constricted spawning window; spawning earlier may cause a mismatch in timing with larval food sources; loss of spawning and rearing habitat	Moderate increase
	Fewer days <22 °C	Reduced adult habitability; earlier exit from the Delta may increase physiological stress/predation risk; potential habitat loss	Moderate increase
Climate Change: Altered Hydrology	Extremely high freshwater flow from Jan-March	Extensive favorable conditions for larvae; increase in spawning and rearing habitat	Substantial decrease
	Steep drop off in flow from April-July	Dry-like conditions reduce estuarine productivity, causing reduced growth, and survival	Substantial increase
Climate Change: Projected Water Year types or Realized Outflow	Increased frequency of dry years	Reduced productivity reduces abundance and potential to rebound	Substantial increase
Climate Change: Sea level rise	Loss of tidal marshes	Spawning habitat loss	Moderate increase
	Salinity intrusion	Constricted habitat to upstream areas, increased risk of entrainment, reduced fitness	Substantial increase
Invasive species	Competition	Reduced food supply	Potentially catastrophic
	Predation	Increased predation stress	Uncertain

Chapter 5 – Conclusion

In our Current Condition evaluation from Chapter 3, we evaluated the status of the Bay-Delta DPS under the framework of the 3 Rs, and indicated we believe that of these, population resilience is the driving consideration for risk evaluation (because redundancy is extremely low, and representation is also believed to be low). We presented abundance index data through time, using multiple available long-term monitoring surveys, and we discussed the extant stressors and how we understand they are limiting the population growth rates. We then evaluated those rates, as indicated through the varied and extensive field-monitoring data, in a simplified, and then a more sophisticated age-structured PVA and were able to quantify the risk profile under a "baseline" scenario which effectively is assuming forward through time, that historical abundance trends and stressors will continue at the same magnitude and frequency.

This Current Condition analysis documented that field survey data indicate that Bay-Delta DPS numbers have declined substantially over time, with abundance indices over the past decade being the lowest on record. We discussed the Bay-Delta DPS resilience in light of historically, and currently extant stressors and their postulated mechanisms of impact. Our meta-analysis based on mean population growth rates and using our baseline scenario (no changes from historically observed conditions or indices moving forward) indicated that the probability of quasi-extinction for the Bay-Delta DPS exceeds 20% by 2035 and reaches 50% in thirty years. Applying the same assumptions over a longer time horizon (i.e., 2050-2065), the suite of surveys predicts that the probability of extinction for the Bay-Delta DPS under current conditions is roughly 55-75%. Our age-structured vital rates PVA tested four age-specific abundance thresholds for quasi-extinction risk and showed a substantial risk of extinction even at low threshold values. For example, using the 10 percent quantile of abundance estimates as the threshold, approximately a 60% of simulations reached extinction by 2050. The 10 percent quantile line also meets Lindley *et al*.'s (2007, Table 1, p. 4) criteria for high risk of extinction (20% probability over two decades). All thresholds used in our Age-Structured PVA meet Lindley et al.'s criteria for moderate risk of extinction (>5% over 100 years).

Our Future Condition scenario planning and analyses for the Bay-Delta DPS from Chapter 4, predicted that declines in population-wide abundance will continue into the foreseeable future under both the RCP 4.5 and 8.5 scenarios. We identified substantial increased population stress imposed by climate change through several mechanisms of impact. These projected changes in the San Francisco Estuary resulting from climate change (and the potential for future invaders) are likely to significantly exacerbate the negative influence of numerous stressors on the Bay-Delta DPS' abundance.

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Technical Appendix A: Summary of Survey Data Used in this SSA

With the exception of eggs, all longfin smelt life stages are collected by one or more of the current fish-monitoring surveys in the San Francisco Estuary. The following 6 long-term fish monitoring surveys currently capture longfin smelt larvae, juveniles, and adults. We use these surveys to determine trends, relative abundance, and to imply geographic distribution for longfin smelt for all age classes. Each of the surveys targets a specific component of the LFS population, while collectively, the information provides excellent data coverage for habitats utilized by juveniles and adult LFS. These surveys also provide information helpful to determine the changes in abundance and distribution over time.

Smelt Larval Survey (SLS)

The SLS was initiated in 2009 and samples for early-stage longfin smelt larvae biweekly January- March by using a single oblique tow per station in the Delta down to eastern San Pablo Bay. The original purpose of the SLS was to provide density and proximity information for larval longfin smelt in relation to the South Delta export facilities and density information within the species range under conditions of low Delta outflow. The primary limitation of the SLS is that it misses larvae hatching in December and April-May and excludes a significant portion of the larval habitat in San Pablo and the South San Francisco Bay and their tributaries (Grimaldo et al. 2020, Fig. 6, p. 10). However, as required by DFW's 2020 Incidental Take Permit, SLS will now cover the Napa River, and two surveys will now be conducted in December.

20-mm Survey

The 20-mm Survey was initiated in 1995 and samples biweekly from late March to early July using three oblique tows per station from the Delta to eastern San Pablo Bay and the Napa River. The original purpose of this survey was to provide density and proximity information for larval and small juvenile Delta Smelt in relation to the South Delta Export facilities as they neared sizes at which they would be counted in salvage. The 20-mm Survey's extended sampling period overlaps with the spawning and early life rearing of many species of fish, including longfin smelt, which has been one of the most commonly collected species (Mahardja et al. 2017, Table 1, p. 491). Like the SLS, a key limitation of the 20-mm Survey for longfin smelt is that it excludes most waters west of the Napa River (Mahardja et al. 2017, Fig. 1, p. 490). The survey's late-March initiation also means it misses sampling opportunities beginning in February, where some fish may be large enough to be regularly retained by the 20-mm fishing gear. Although, it is not clear if this data would further inform conservation or scientific interpretation in any meaningful way.

Fall Midwater Trawl (FMWT)

The FMWT was initiated in 1967 and has consistently sampled pelagic waters from the Delta into San Pablo Bay monthly from September-December by using 1 oblique tow per station in the water column

up to a depth of 40 feet. In some years prior to 2002, the survey extended through March (CDFG 2009, p. 8 and CDFG 2009, Appendix A). Note that the FMWT has more than twice as many stations as other Interagency Ecological Program (IEP) surveys. The survey mostly collects juvenile to small-sized adult fishes and decapod shrimps (50-150 mm). The original purpose of the FMWT was to determine if age-0 striped bass mortality varied enough from summer to fall to affect adult abundance (Stevens 1977, p. 96). The FMWT does not sample the full range of longfin smelt in the fall, but the time series of the longfin smelt FMWT index tracks those from the more spatially comprehensive Bay Study because longfin smelt return to the upper estuary in the fall and early winter placing enough of them in the FMWT sampling grid to generate a reasonable index (Rosenfield and Baxter 2007, p. 1590). One other potential survey limitation from the perspective of indexing the relative abundance of longfin smelt is they do not fully recruit to the gear until 60-70 mm, a size the average age-0 individual often does not reach until it is transitioning to age-1 (December-February). To the extent that longfin smelt may be using deeper water during the day, this phenomenon would also reduce the capture efficiency of the FMWT.

Bay Study Survey

The Bay Study was initiated in 1980 and samples monthly year-round with 2 tows per station: one with an otter trawl (OT), which samples demersal fishes, crab, and shrimp, and one with a midwater trawl (MWT), which samples pelagic fishes in the water column to approximately 40 feet deep. The Bay Study samples most of the San Francisco Estuary from the western Delta to South San Francisco Bay, and targets juvenile to small-sized adult fishes (20-250 mm). The original purpose of the Bay Study was to provide data to monitor the influence of freshwater outflow on abundance trends and distributional patterns of a suite of fish and invertebrate species (Armor and Herrgesell 1985, p. 211). The limitation of the Bay Study from the perspective of sampling longfin smelt is that juvenile longfin smelt do not fully recruit to OT gear until they are about 40-50 mm and fully to MWT gear until 60-70 mm. Like the FMWT, the Bay Study produces annual indices of relative abundance for longfin smelt. Bay Study data are used to calculate separate indices for OT and MWT for three age classes (age-0, age-1, and age 2+) of longfin smelt for a total of 6 annual abundance indices. The survey's distinction between longfin smelt age classes allows for extensive analysis for species vital rates, life stage transition, and geographical range analyses.

Chipps Island Trawl

Chipps Island Trawl was initiated in 1976 and samples 3-7 days a week using a MWT near the surface to conduct ten 20-minute tows per sampling day and across three trawl lanes (north, central, south) to produce varied catches. The survey used to sample only during spring, but has been sampling year round since 1995. The survey targets juvenile to small adult fishes (40-150 mm), and was originally designed to estimate the percent passage or survival of emigrating juvenile Chinook Salmon. For longfin smelt, Chipps Island Trawl's primary limitation is its sampling location which is restricted to the waters around Chipps Island, and the use of only a surface-towed MWT net meaning it does not sample fish residing lower in the water column. Unlike the surveys above, Chipps Island Trawl makes 10 tows each

day and sampling during the winter and spring provides frequent catches of longfin smelt leading up to and during their spawning season.

Suisun Marsh Survey

The Suisun Marsh Survey was initiated in 1980 and samples monthly year-round using a single OT tow at each sampling location within the sloughs of Suisun Marsh. The survey targets juvenile and small adult fish (25-250 mm), and was originally intended to track trends in distribution and abundance of shrimp and fish communities within the marsh complex. From the perspective of sampling longfin smelt, the primary limitation of Suisun Marsh Survey is its location that is restricted to the marsh.

None of the above listed field surveys sample the local coastal waters believed to provide habitat for some age-1 longfin smelt during July-September (Rosenfield and Baxter 2007). The CDFW Longfin Smelt Science Program is currently discussing the expansion and addition of field surveys to help improve monitoring of the Bay-Delta DPS. This program is intended to both promote science which analyzes assumptions and hypotheses used in the incidental take permit, while also addressing important, longstanding questions related to the ecology of the species. Since FMWT and Bay Study are thought to sample sufficient extent of the range for the DPS for the covered time periods, they produce representative annual abundance indices for tracking abundance trends over time. In addition, we use data from the SLS and 20-mm Survey to estimate the abundance, trends, and distribution of larval and small juvenile Longfin Smelt.

Technical Appendix B: Count-Based Population Viability Analysis Using Indices of Abundance for the Bay-Delta DPS Longfin Smelt

Tobias, V. D., Chen, E., Hobbs, J., Eakin, M., & Detwiler, S. (2023). Informing extinction risk: Summarizing population viability through a meta-analysis of multiple long-term monitoring programs for a declining estuarine fish species. *Biological Conservation*, *288*, 110348.

Introduction

Designating species as endangered or threatened may serve to highlight the need for conservation in a scientific, popular, or political context, as is the goal of the IUCN Red List (Mace et al. 2008), or to confer regulatory protections, as is the case for the U.S. Endangered Species Act of 1973 (ESA) and the California Endangered Species Act of 1970 (CESA). For the ESA, the decision to include a species under its protection is made by the Secretaries of the Interior and Commerce, acting through the U.S. Fish and Wildlife Service (USFWS) and National Oceanic and Atmospheric Administration, based on an assessment of the best available scientific and commercial data for the species (United States 1983; Smith et al. 2018). The USFWS recently implemented a new framework for decision making related to the ESA. Under the new approach, experts gather the best available information and data on a species into a Species Status Assessment (SSA) report (Smith et al. 2018). The SSA framework supports decision-making by describing the species' needs and life history as well as current and potential future threats to its continued existence in the wild. SSAs also include assessments of the future condition of the species in light of anticipated changes in the magnitude of extant or future stressors. In some cases, this may be a qualitative description of the trajectory of the species, or it could be a quantitative assessment of population viability depending on the quantity and quality of data available (Smith et al. 2018).

Population viability analyses (PVAs) can take many forms, depending on the questions of interest and the data that are available (Morris and Doak 2002), and they have been applied to a wide range of species including insects, fish, reptiles, mammals, and birds (e.g., Shaffer 1983 on grizzly bears, Schultz and Hammond 2003 on butterflies, Tucker et al. 2021 on boa constrictors). Many monitoring programs collect data that is used to assess population status and trends in abundance for species of management concern. From these, a time series of abundance indices can be used to calculate population growth rates and to forecast the risk of extinction into the future through the application of a population viability analysis (PVA). Different methods for conducting PVAs have been developed for different types of available data. For example, data on specific ages or stages of an organism's life cycle can be used to fit projection matrix models (e.g., Leslie 1945; Lefkovitch 1965). A count based PVA is classically applied to census data (counts of an entire population), but it is not necessary to count the entire population (Dennis et al. 1991a; Morris and Doak 2002). A count based PVA can also be applied to index values, where a population index represents some portion of the total population as long as the proportion of the population that is observed remains relatively constant over time (Morris and Doak 2002). When an SSA includes a quantitative assessment, the experts may need to decide how to incorporate all of the relevant data where multiple long-term datasets exist. In some cases, the textbook PVA methods need to be modified to incorporate data from multiple sources and of differing data types. For example, a

combination of available datasets on black rail (*Laterallus jamaicensis*) observations from across their range were combined into a dynamic model of demographic occupancy (McGowan et al. 2020). Population estimates of gopher tortoise (*Gopherus polyphemus*) from multiple sites have been combined into a spatially explicit PVA that is responsive to multiple population threats (Folt et al. 2022). A PVA was also developed that combined habitat area and egg counts in a Bayesian analysis for a population of monarch butterflies (*Danaus plexippus*) that migrates from Mexico to the Midwestern United States (Semmens et al. 2016).

In the fall of 2022, the USFWS completed an SSA for the San Francisco Bay-Delta distinct population segment (DPS) of the Longfin Smelt (*Spirinchus thaleichthys*) and proposed listing it as endangered under the ESA. As is the case for many other species of concern, ecological monitoring provides data on the abundance of Longfin Smelt in the San Francisco Estuary (also known as the San Francisco Bay-Delta). The San Francisco Estuary has a long history of monitoring fish abundance and distribution. Within the Estuary, a consortium of state and federal agencies called The Interagency Ecological Program (IEP) conducts a suite of fish monitoring surveys, some of which have been collecting data for over 50 years. Several of the IEP long-term monitoring surveys provide indices of Longfin Smelt abundance in the San Francisco Estuary. Some of these indices are calculated and published annually (e.g., the Fall Midwater Trawl [FMWT] abundance index), but another was calculated for this analysis (the 20-mm Survey). An index of abundance need not be a formal index at all; a summary of catch per unit effort (CPUE) can also be thought of as a population index. These long-term monitoring datasets create an opportunity to investigate trends in abundance and to estimate the probability of long-term viability of many species of management concern. Longfin Smelt abundance has declined substantially since monitoring began (Sommer et al. 2007; Rosenfield and Baxter 2007; Nobriga and Rosenfield 2016). Despite consensus that abundance has trended downward, the fluctuations around that trend can make it difficult to quantitatively assess the risk of extinction. For example, it may be difficult to see the effects of one or two years with favorable conditions on the overall trajectory of abundance.

In this paper we present a novel analysis of the long-term monitoring data for the San Francisco Bay-Delta DPS of Longfin Smelt which was developed for the SSA. Specifically, we addressed the following questions that are relevant to the SSA: (1) Does the suite of available monitoring data agree on a trajectory for the abundance of Longfin Smelt? (2) What information is gained by having multiple surveys that track Longfin Smelt abundance? (3) Is the population at risk of extinction within the next two decades? To answer these questions, we summarize the information contained in the monitoring data by applying a count based PVA to the IEP's monitoring data for Longfin Smelt. Applying the PVA method to several datasets that index the abundance of Longfin Smelt captures the landscape of available information and may be used to make decisions about the status of the species or about management actions. It also presents a way to synthesize the evidence of the direction and magnitude of change in Longfin Smelt abundance from indices calculated using various methods and that exist on varying scales. We further synthesize population growth rates by conducting a meta-analysis on these metrics.

Methods

2.1 Study Species

Longfin Smelt (*Spirinchus thaleichthys*) is a small (≤150 mm FL), facultatively anadromous, pelagic fish with an up to 3 year lifespan (Baxter 1999). It is native to coastal waters of the eastern Pacific Ocean and bays and estuaries, ranging from Monterey Bay, California, to Alaska (Moyle 2002). Both anadromous and resident populations exist at various locations, and the DPS in the San Francisco Estuary is anadromous (Dryfoos 1965; Moulton 1974; Rosenfield and Baxter 2007). The population of Longfin Smelt in the San Francisco Estuary is genetically distinct from other populations, but there is evidence of outward gene flow from the San Francisco Estuary population to populations in Humboldt Bay and the Columbia River (Garwood 2017; Sağlam et al. 2021). Based in part on the substantial decline in abundance, Longfin Smelt was listed as threatened state-wide under the CESA in 2009 by the California Department of Fish and Game [now the California Department of Fish and Wildlife] ([OAL] Office of Administrative Law 2010), and the USFWS later determined that protection of the Bay-Delta DPS of the Longfin Smelt under the ESA was warranted but precluded by higher priority actions (USFWS 2012). The novel analysis reported herein was utilized to help inform the extant and future extinction risk for the DPS and a final determination by USFWS as either threatened or endangered under the ESA.

2.2 Data

Several long-term monitoring surveys produce abundance indices for Longfin Smelt in San Francisco Bay and the Sacramento San Joaquin Delta (Tempel et al. 2021). We used these indices to estimate apparent annual population growth rates for various life stages of Longfin Smelt (Figure 1). The surveys have different periods of record, ranging from 24 to 54 years. A combination of sampling gear issues and interruption caused by the COVID-19 pandemic prevented calculation of many abundance indices from 2016-2021. Below are detailed descriptions of the methods for each survey, including gear types, index calculations, and periods of record.

The Fall Midwater Trawl Survey (FMWT) consists of four monthly surveys from September through December conducting a single oblique tow of a midwater trawl net amongst 100 fixed stations, spatially stratified into 14 regions encompassing the Sacramento-San Joaquin Delta downstream to San Pablo Bay. The midwater trawl net has mouth dimensions of 3.66 m x 3.66 m. Net mesh sizes graduate in nine sections from 20.3 cm stretch-mesh at the mouth to 1.27 cm stretch-mesh at the cod-end which is effective at retaining pelagic forage fish (>60mm FL) including the Delta Smelt which has a similar body shape to Longfin Smelt (Mitchell et al. 2017; Mitchell and Baxter 2021). Monthly indices of abundance are calculated by averaging catch per tow in each region and multiplying means by regional weighting factors scaled to water volume and summing these products for all 14 regions with annual indices being the sum of the four monthly indices (Stevens and Miller 1983). The annual abundance indices for Longfin Smelt from the FMWT represents all age classes, however; the majority of fish caught by the FMWT are age-0. The FMWT began sampling in 1967.

The San Francisco Bay Study (SFBS) conducts monthly year-round sampling with an obliquely towed midwater trawl and benthic otter trawl amongst 35 stations spatially stratified into 5 regions from the

confluence of the Sacramento-San Joaquin River to South San Francisco Bay. The SFBS midwater trawl has the same specifications as the FMWT, while the otter trawl has a headrope length of 4.9-m with 2.5-cm mesh body and 1.3-cm stretch mesh codend which is effective at retaining fishes >40mm FL (Baxter et al. 1999). Monthly abundance indices are calculated by averaging the catch per unit effort, where effort is reported as volume for the midwater trawl and area swept for the otter trawl by region and multiplying regional means by regional weighting factors scaled to water volume (MWT) or area (OT) and summing these products across the 5 regions. Annual indices are calculated as the average monthly index over the period for which the life-stage was most abundant, for age-0 (40-mm-82-mm FL) from May through October, age-1 (43-mm to 92-mm FL) and age-2 (>93-mm FL) from February through May, where ages were assigned using an age-length key derived from length-frequency analysis (Baxter et al. 1999). The SFBS began sampling in 1980.

The 20-mm Survey conducts bi-weekly surveys conducted 3 tows at 47 stations from eastern San Pablo Bay to the Sacramento-San Joaquin Delt from March through July using a 1,600 μm mesh net 5.5-m in length with a mouth opening of 1.51m^2 affixed to a steel D-frame with skids that retains Delta Smelt from 6-50-mm and is most effective at retaining fish approximately 20mm FL (Mitchell et al. 2019). The 20-mm Survey does not produce an index of Longfin Smelt abundance, so we adapted the existing methods used for the Delta Smelt (*Hypomesus transpacificus*) index to create a 20-mm survey abundance index for Longfin Smelt. It is reasonable to adapt methods developed for Delta Smelt in the 20-mm survey to Longfin Smelt because detection probabilities are similar for these species in this survey (Mahardja et al. 2017). We calculated CPUE for sampling stations that are included in the Delta Smelt index calculation ("index stations") as Longfin Smelt catch per 10,000 m^3 of water, and log transformed the CPUE values, using ($\log_{10}$(CPUE + 1); Tempel et al. 2021). We calculated the mean of the log-CPUE values for each month within a year and then back-transformed the mean values to put them back on the untransformed density scale. The 20-mm survey index for Delta Smelt bases the selection of surveys on the size of Delta Smelt. For this survey, we used the maximum density value as the index of abundance rather than a mean of specific months. We did this partially for simplicity and partially because using the maximum value reduces the impact of any issues with changing timing of Longfin Smelt presence in the 20-mm survey sampling area. The 20-mm Survey began sampling in 1995.

2.3 Trends in Abundance Indices

We tested for monotonic trends in the individual time series (Tables

Table 1) using a Mann-Kendall test (Pohlert 2020). Missing values at the beginning and end of each time series were trimmed; missing values inside of the time series were filled with the mean of the two closest non-missing values (i.e., usually the prior and subsequent year unless there were consecutive missing values).

2.4 Population Growth Rates

Annual population growth rate (λ) estimates were based on a simple unstructured population time series, where a single estimate or index of population size (or the size of a segment of the population) is produced annually. We used an exponential growth state space (EGSS) model to estimate λ, the details

of which are below. Our method was similar to a restricted maximum likelihood (REML) model that Staples et al. (2004) proposed as a solution to accounting for observation error in estimating population growth rates. Staples et al. (2004) found that their REML method of estimating population growth rates produced unbiased estimates of λ. When compared to the diffusion approximation method proposed by Dennis et al. (1991b), Staples et al. (2004) found that the REML method gave more accurate estimations of process error, which is necessary for PVA applications.

We used program R (R Core Team 2019) for all calculations and estimates. We modified code for an EGSS model that was published by Humbert et al. (2009) to estimate $\mu = \log(\lambda)$ and its associated variance. This method does not allow for zeros in the timeseries so the few instances where an abundance index was zero were replaced with a one. Years that were missing values for an abundance index were omitted from the analysis. Our approach to calculating population growth rates for multiple surveys was similar to that of Schultz and Hammond (2003), but we took an additional step to summarize the growth rates. We first applied the EGSS model to the time series from each survey. We then conducted a meta-analysis using the mean and variance of population growth rates from the individual surveys as independent measurements of population trajectory and variability. A meta-analysis calculated a pooled estimate of the mean growth rate from results of the population growth rate calculations from each of the monitoring programs described above. We used the metamean function from the meta package in R (Balduzzi et al. 2019).

2.5 Evaluation of Assumptions

Count PVAs assume that the mean and variance of the population growth rate do not change over time. We assessed this assumption by plotting the λ values over time. We also checked that the mean population growth rate is density-independent. To do this, we graphed the population growth rates against the abundance index for each year in the datasets and looked for a trend. Graphs for most of the surveys did not show trends, but a few plots were inconclusive, so we compared models for density dependence (Beverton-Holt and Ricker) against a density-independent (linear) model for the population growth rate. In order to guard against any undetected density dependence, we followed the advice of Morris and Doak (2002). First, we limited our projections of extinction risk to a relatively short time frame (50 years) to limit the influence of negative density dependence (i.e., predicting extremely high populations that exceed carrying capacity). Second, we set quasi-extinction thresholds to limit the influence of positive density dependence (i.e., Allee effects). To test for environmental autocorrelation, i.e., a tendency for adjacent population growth rates to be more similar to one another, in the timeseries we performed a Durbin-Watson test. When the Durbin-Watson test indicated significant autocorrelation, we used the autocorrelation function (ACF) to assess the degree.

2.6 Selection of Quasi-Extinction Thresholds

A quasi-extinction threshold is a population size below which a population cannot recover or when the abundance is so low that plausible management strategies would need to drastically change for the species to recover; for example changing the focus of restoration efforts from habitat management to captive breeding (Tucker et al. 2021). Quasi-extinction is used as the threshold for PVAs rather than full

extirpation because this strategy minimizes the uncertainty around population dynamics for small populations (Busch et al. 2013) and because it is consistent with the endangered designation used for the ESA (Boyd et al. 2017). Our analysis uses distinct values for the quasi-extinction thresholds for each of the surveys because the scale of the abundance indices differs across surveys; note that even surveys that address the same age class or use the same gear do not have the same abundance index values in the same year (Table 1). Like the abundance indices on which they are based, quasi-extinction thresholds are different for each survey because variations among surveys, such as differences in sampling methods, sites, and index calculations. The following procedure was developed to account for the different magnitudes of index values across the surveys. First, we calculated the mean value of all available index values for each survey over 2009-2018. These years were chosen to represent recent abundance index values for each monitoring survey, specifically the years since Longfin Smelt were listed as threatened under the California Endangered Species Act. We then multiplied that mean value by 0.01 to represent a value of a major decline in abundance from the recent average. The major decline value was then rounded to zero decimal places to create an integer value. We also set the minimum value to 1 *a priori*, but none of the calculated threshold values was less than 1. Final quasi-extinction threshold values are reported in Table 3.

2.7 Probability of Extinction over Time

We modified the function extCDF (package popbio; Stubben and Milligan 2007) to estimate the cumulative density function (CDF) for the probability of extinction. The modified functions use the results of the regression analysis and can incorporate autocorrelation into the simulated abundance at each timestep. The modified function requires the following input values: mean and variance of the population growth rate, the current abundance index, a quasi-extinction threshold, and a level of autocorrelation. Mean and variance of the population growth rate as well as the quasi-extinction thresholds were set as described above. The value for the current abundance index was set to the most recent index value that was available (Table 1). A second function produced bootstrapped means and confidence intervals for the CDFs.

To calculate the cumulative probability of extinction over time for the population growth rate derived from the meta-analysis, additional values had to be chosen for the quasi-extinction threshold and the starting abundance index. There is no single method for setting the quasi-extinction level for viability analyses, but previous studies have employed techniques such as expert opinion (Schultz et al. 2017) and calculating viability metrics for a range of quasi-extinction thresholds (Holmes et al. 2007). We set the initial populations size to the mean of the initial abundance index for the surveys. We set the quasi-extinction threshold to 2.6% of the initial abundance index to be consistent with the mean ratio of quasi-extinction values and initial abundance indices of the surveys, resulting in a quasi-extinction threshold of 50. We considered setting the quasi-extinction threshold to the mean of the quasi-extinction thresholds for all of the surveys, but this value (16) was so low that it risked introducing unaccounted-for positive density-dependent effects (e.g., Allee effects) into the simulation. Realizing that these choices could affect the results of the PVA based on the meta-analysis, we also ran the simulation with several additional values and graphed the results to illustrate the potential effect of these choices.

Results

3.1 Monotonic Trends

Values for the indices of Longfin Smelt abundance have decreased substantially over the time series (Tables

Table 1). None of the abundance indices show evidence of increasing abundance indices: Mann-Kendall tests for all of the abundance indices have negative z-values, indicating negative monotonic trends, and all except for the SFBS age-0 otter trawl index have p-values less than 0.05 (Table 2).

3.2 Population Growth Rates

For all surveys, the most likely direction for annual population growth is negative (Figure 2). Mean population growth rates were less than one for all of the abundance indices, which indicates that abundance is declining over time (Table 3); however, variability was high for all surveys and confidence limits on all estimates included the possibility of a stable abundance (Figure 2). The mean from the meta-analysis was 0.92 (95% CI: 0.86, 0.99), which indicates a population growth rate below that of a stable population. Population growth rates for all surveys suggest that there have been substantial reductions in abundance. Based on the mean lambda values, and the assumption that lambda is constant over time, declines over 20 years amount to between roughly 42% and 89% of the population.

3.3 Probability of Quasi-Extinction over Time

The predicted probability of quasi-extinction for most surveys, except for the SFBS age-2 OT, exceeds 20% by 2040 (Figure 3). Applying the same assumptions over a longer time horizon (i.e., 2050-2065), the suite of surveys predicts that the probability of extinction for the DPS under current conditions is roughly 45-90% across all surveys. Based on the meta-analysis, the mean quasi-extinction value for the population is 33% (25%, 41%) over 20 years and rises to 50% (42%, 58%) in 30 years (Figure 4).

Although the confidence interval on mean population growth rate from the meta-analysis does not include zero, we also calculated the probability of quasi-extinction for a stable population. If the population were not declining (i.e., if the population growth rate were zero), but the initial abundance and variance of the estimated population growth rate were the same as calculated for the meta-analysis, the quasi-extinction risk would reach 40% (31%, 47%) in 20 years and 52% (43%, 59%) in 30 years. Varying the initial abundance could make the extinction probabilities higher, and the values that were tested here reached as high as 80% (Figure 5). The values in Figure 3 were near the lower part of the range of extinction probabilities (high abundance). Quasi-extinction predictions were sensitive to the choice of starting abundance and quasi-extinction threshold (Figure 5). As the starting abundance increases, the probability of quasi-extinction rapidly approaches a lower threshold. Increasing the quasi-extinction threshold creates the opposite effect, causing the quasi-extinction probability to reach an upper threshold.

3.4 Evaluation of Assumptions

Our datasets did not show strong evidence of changes in the mean and variance of population growth rates over time. Plots of population growth rates over time did not show obvious trends in either direction. Similarly, plots of population growth rates as a function of the previous year's abundance did not show a strong slope, but visual evidence was inconclusive for surveys with outliers. To be sure the linear models were sufficient, we compared them to density-dependent formulations. Model comparisons indicated no evidence to support either of the density-dependent formulations being better than the density-independent formulation (Table S1). Durbin-Watson tests showed evidence for negative autocorrelation in all indices except the Bay Study age-2 midwater trawl and 20-mm. ACF plots indicated that the strengths of the first-order autocorrelations were about -0.5 when they were significant.

Discussion

4.1 Population Growth Rates

The suite of long-term monitoring programs in the SFE agrees and provides evidence that the abundance on Longfin Smelt in the SFE has declined dramatically over the monitoring record. The abundance index values for recent years are lower than they have ever been and estimates of annual population growth indicate a declining population on average. Longfin Smelt abundance has undergone several step declines since monitoring began (Thomson et al. 2010) as a result of changes in environmental conditions such as the introduction of invasive clams in the mid-1980s (Alpine and Cloern 1992). This decline is consistent with patterns in several other organisms, a phenomenon that is locally known as the pelagic organism decline (Sommer et al. 2007). The variability in the estimated population growth rates reflects the fact that in some years longfin smelt abundance indices increase substantially, but in most years they decrease. Generally, populations with highly variable growth rates tend to have lower levels of population viability over time because the inherent variability tends to make a population grow more slowly over the long term and the abundance index is more likely to fall below the quasi-extinction threshold than it is for populations with less variable growth rates (Morris and Doak 2002). Without intervention, populations with negative growth rates are expected to go extinct, regardless of the initial abundance or the variability in their growth rates. The main question to investigate becomes when extinction is likely to happen, because this determines the timeframe for implementing management actions to increase the growth rate.

Information from multiple surveys can help the conservation community be more confident in summaries of population trajectories and thus in decisions based on those summaries. The meta-analysis indicates that on average, the surveys are tracking an actual decrease in Longfin Smelt abundance over time confirming the apparent visible data trend. This finding is consistent with previous studies that used indices of abundance to evaluate patterns in Longfin Smelt populations. For example, a pattern of decreasing catch per trawl has been fairly consistently over time, with a few short increases (Sommer et al. 2007; Thomson et al. 2010). By leveraging data from multiple monitoring surveys, we derived better information about the population growth rate. When looking at each survey individually,

the mean growth rate indicates a declining population, but the error bars indicate a possibility of stability or even increase. When using information from all of the surveys together, the variability around the mean is dampened, compared to the individual surveys, and the error bars do not include zero. Variation around the population growth rate contributes to high estimated risk of quasi-extinction, whether predictions are made using the mean population growth rate or using an assumption that population growth is stable. In either scenario, the message for managers is that increased population growth rates and reducing variation around those growth rates are necessary to reduce the risk of quasi-extinction.

4.2 Evaluation of assumptions

PVAs are best used to predict extinction probabilities when extensive and reliable data exist, and if the distribution of vital rates is stationary or predictable in the future (Coulson et al. 2001). For Longfin Smelt, several long-running datasets exist that can inform a PVA. Although the structure of some PVAs can be very complex to account for various issues with data quality or for a lack of data, here we took a simple approach, in part because we have an abundance of consistent data. Our approach leverages the same assumptions as are used for interpreting monitoring data in general, specifically that indices of abundance are proportional to true abundance. For this analysis, the relationship between the index values and true abundance need not be known because as long as it is constant over time, the population growth rate would be unaffected. If detection probabilities vary in space or time, or if the proportion of the population that is sampled varies in a systematic way, both the indices and any analysis based on them would be suspect (Thompson 2004). On the other hand, when capture probabilities are the same between two time periods, the ratio of count-based statistics between the two time periods is sufficient (Thompson 2004), and such methods give a more precise estimate of changes in the population than a more complex method such as single mark-recapture estimators (Skalski et al. 1983). Another assumption for PVAs is that there is no observation error in the time series. The EGSS model accounts for observation error that is present in the timeseries for the surveys. Removing observation error guards against introducing unrealistic variability in the estimates of the population growth rate, which would lead to underestimating viability and thus pessimistic predictions (Staples et al. 2004; Humbert et al. 2009). The meta-analysis approach further dampens this effect because incorporating information from all of the surveys to reduce variation in the estimate of the population growth rate overall.

Different assumptions about the form of population regulation (i.e., density dependence) have been shown to affect the expected population viability significantly (LaMontagne et al. 2002). Our analysis relies on a simplified version of the population dynamics for Longfin Smelt. In particular, it does not account for age structure in the population or potential density dependence in the time series of recruits per spawner, which has been suggested by previous studies (Nobriga and Rosenfield 2016). Our evaluation of the data supports a density-independent modeling structure when examining year to year variation of the same age class. Recent abundance levels are very low so it is unlikely that negative density dependence would affect our forecasted simulations, especially those using the meta-analysis mean and SD values. Even if high population values were predicted, the effect of negative density dependence would not change our conclusions that Longfin Smelt are at risk for extinction. Left

unaccounted for, negative density dependence could cause unrealistically high estimates of abundance, which would make the calculated quasi extinction rates lower than they should be (i.e., the population would appear less likely to go extinct than it should). Longfin Smelt might already be experiencing Allee effects (i.e. positive density dependence); thus, the calculated quasi extinction rates might be optimistic. The selected quasi-extinction thresholds were chosen to represent values that are much lower than recent averages and to be tailored to the magnitudes of individual indices, but they were not informed by considerations of when management actions could still affect abundance.

4.3 Management Implications

The ESA does not set quantitative definitions or thresholds for applying threatened or endangered designations, but other benchmarks can provide context for interpreting quasi-extinction rates. For example, a 20% risk of extinction over 20 years has been proposed by multiple sources as a criterion for a species being endangered (Lindley et al. 2007; Mace et al. 2008). Another definition holds that a population with a 95% probability of persisting for 100 years can be considered secure and not at risk of extinction (Hamilton and Moller 1995). We found that the Longfin Smelt population in the San Francisco Estuary meets these three sets of criteria to be considered at risk of extinction. For Longfin Smelt, most of the increase in cumulative probability of extinction occurs in the next two decades, but there may be time to make changes before Longfin Smelt become undetectable by monitoring surveys. In cases like that of Longfin Smelt where abundance is low, but the risk of extinction is relatively small for the immediate future, conservation and management efforts should focus on achieving long-term viability by taking steps to increase the population growth rate for the species (Morris and Doak 2002). The structure in the population growth rates can also suggest some management strategies. Negative autocorrelation in the time series of population growth rates indicates that Longfin Smelt abundance tends to decrease after years when abundance increased. Knowing that this tendency exists, management strategies can be formulated to boost recovery even more in years with favorable conditions and limit the expected decrease in the following year, thus encouraging an overall positive trend in abundance.

This study demonstrates a practical way that having multiple sources of information creates better information about the trajectory of the population. Individually, the monitoring surveys contribute information about specific life stages or ages to the understanding of the population. Here we demonstrated that, as a data synthesis, combining information from multiple surveys improves confidence in estimates of the trajectory of the population. In effect, we show how a set of simple PVAs used as preliminary models can formalize uncertainty in the current understanding of the trajectory of the species (Hamilton and Moller 1995). Even when multiple data sources identify likely species declines, synthesizing the landscape of information into one metric and an associated graphical summary may be useful for ecosystem managers and decision-makers. In addition to succinctly communicating risk information, this summary presents a benchmark for evaluating future decisions. This PVA represents a baseline scenario that considers a future with no changes in management or environmental conditions, which can be used for comparison with potential future management actions and recovery planning. One limitation of the current study for management implications is the wide confidence intervals, even on the meta-analysis mean. We addressed this issue by bootstrapping

estimates, but it could also be addressed through more sophisticated models that account for variability in a more structured way. A Bayesian state-space model approach to combining data from multiple surveys has been shown to produce precise parameter estimates, even when data were sparse (e.g., Johnson et al. 2010) and we plan to pursue this approach in an upcoming manuscript.

R code for the analyses and graphs can be found at https://github.com/USFWS/Longfin-Smelt-Meta-Analysis-PVA

**Note: The published version of this article (https://doi.org/10.1016/j.biocon.2023.110348) was published with incorrect 20-mm indices in Table 1. Table 1 has been corrected in this Appendix.*

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Tables

Table 1: Abundance index data that were used in the population viability analysis. Years when indices were not calculated are denoted by NA.

Year	BSMT Age-0	BSMT Age-1	BSMT Age-2	BSOT Age-0	BSOT Age-1	BSOT Age-2	FMWT	20-mm
1967	NA	NA	NA	NA	NA	NA	81737.0	NA
1968	NA	NA	NA	NA	NA	NA	3279.0	NA
1969	NA	NA	NA	NA	NA	NA	59350.0	NA
1970	NA	NA	NA	NA	NA	NA	6515.0	NA
1971	NA	NA	NA	NA	NA	NA	15903.0	NA
1972	NA	NA	NA	NA	NA	NA	760.0	NA
1973	NA	NA	NA	NA	NA	NA	5896.0	NA
1974	NA	NA	NA	NA	NA	NA	NA	NA
1975	NA	NA	NA	NA	NA	NA	2819.0	NA
1976	NA	NA	NA	NA	NA	NA	658.0	NA
1977	NA	NA	NA	NA	NA	NA	210.0	NA
1978	NA	NA	NA	NA	NA	NA	6619.0	NA
1979	NA	NA	NA	NA	NA	NA	NA	NA
1980	190790.1	1385.9	2572.0	128320.8	141.6	106.0	31184.0	NA
1981	1958.6	51371.8	644.4	4139.0	11836.6	121.6	2202.0	NA
1982	299068.9	9785.5	2306.9	257965.5	3069.6	1004.8	62905.0	NA
1983	33650.8	296253.0	2865.2	23859.5	142861.2	917.2	11864.0	NA
1984	29218.2	25462.8	4081.8	44329.0	29399.2	5766.6	7408.0	NA
1985	2894.6	58525.3	2693.5	11786.9	12626.3	1185.0	992.0	NA
1986	24908.3	12523.6	2479.2	12069.6	2784.1	287.9	6160.0	NA
1987	2872.3	33470.8	2286.5	1983.6	7173.7	1284.1	1520.0	NA
1988	1724.0	18360.0	4920.9	1093.9	4322.6	2220.2	791.0	NA
1989	1136.7	6594.5	1514.1	970.6	2178.8	368.7	456.0	NA
1990	744.5	2776.2	1058.4	680.6	385.7	316.6	243.0	NA
1991	131.1	3851.7	540.7	244.9	473.8	160.5	134.0	NA
1992	369.9	1133.9	86.0	620.2	447.5	218.3	76.0	NA
1993	5085.7	810.8	22.4	7006.0	53.3	0.0	798.0	NA
1994	NA	16515.5	349.0	2847.2	4206.9	479.1	545.0	NA
1995	555398.1	NA	NA	152973.0	791.0	503.9	8205.0	1.7
1996	666.3	NA	NA	11045.5	4152.9	247.9	1346.0	9.0
1997	4585.3	6862.8	1788.8	10691.6	3705.8	1075.3	690.0	43.7
1998	62852.8	6240.4	1120.2	20605.1	196.7	89.0	6654.0	3.2
1999	59040.3	17545.6	895.0	57979.8	6827.4	600.4	5243.0	18.8
2000	12325.8	12132.1	1168.2	16079.2	1841.5	240.4	3437.0	85.6
2001	2107.4	10706.9	1154.1	812.0	15507.1	1162.9	247.0	60.1
2002	1172.6	2471.5	1899.7	18132.4	1069.8	1604.3	707.0	212.2

2003	230.4	2557.6	185.6	4006.9	6338.8	1291.7	467.0	39.0
2004	1307.1	2030.2	485.5	3529.1	4322.9	885.9	191.0	20.3
2005	617.3	3487.4	544.7	8459.1	4074.4	593.3	129.0	7.1
2006	2779.8	1341.9	91.1	21516.8	1378.6	284.7	1949.0	1.4
2007	441.3	1350.9	78.9	3636.5	5155.9	814.9	13.0	6.1
2008	1207.3	217.6	229.6	6154.8	343.4	177.5	139.0	24.2
2009	322.6	3485.1	75.2	970.8	10317.0	469.5	65.0	17.7
2010	867.1	598.9	114.4	628.3	1373.2	279.8	191.0	16.4
2011	1404.5	950.4	106.2	14261.5	7134.5	503.3	477.0	3.0
2012	397.5	5353.0	1105.6	2170.2	839.9	359.9	61.0	6.9
2013	1445.0	647.0	324.0	15545.0	574.0	277.0	164.0	39.3
2014	1193.7	615.5	0.0	1228.4	518.7	63.1	16.0	6.8
2015	230.8	402.9	43.6	536.2	692.6	195.7	4.0	1.4
2016	NA	NA	NA	NA	NA	NA	7.0	2.8
2017	NA	NA	NA	NA	NA	NA	141.0	1.3
2018	NA	NA	NA	NA	NA	NA	52.0	1.8
2019	NA	NA	NA	16196	1042	146	44.0	2.6
2020	837	NA	NA	12109	NA	NA	28.0	2.8
2021	NA	NA	NA	NA	NA	NA	323.0	5.0

Table 2: Results of a Mann-Kendall test for monotonic trends in the available time series of abundance indices for Longfin Smelt. Missing values were trimmed from the ends of the time series and missing values within the time series were filled with local average values.

Survey	**z**	**p**	**n**	**S**	**varS**	**tau**
SFBS MWT Age-0	-3.1	< 0.001	41	-277	7921	-0.34
SFBS MWT Age-1	-5.1	< 0.001	38	-405	6322	-0.58
SFBS MWT Age-2	-4.8	< 0.001	38	-383	6322	-0.55
SFBS OT Age-0	-1.5	0.13	41	-136	7924	-0.17
SFBS OT Age-1	-2.3	0.02	40	-197	7363	-0.25
SFBS OT Age-2	-2.3	0.02	40	-202	7364	-0.26
FMWT	-6.1	< 0.001	55	-836	18972	-0.56
20-mm	-2.7	0.01	27	-129	2301	-0.37

Table 3: Values corresponding to the mean and 95% confidence limits presented in Figure 1 and the values used to produce Figure 2. These values include all years for which index values were calculated.

Survey	Lambda Mean	Lambda Lower CL	Lambda Upper CL	Quasi-Extinction Threshold	Current Abundance Value
SFBS MWT Age-0	0.90	0.71	1.15	10	837
SFBS MWT Age-1	0.90	0.76	1.08	15	161
SFBS MWT Age-2	0.89	0.72	1.10	2	49
SFBS OT Age-0	0.96	0.74	1.23	66	12109
SFBS OT Age-1	0.93	0.73	1.18	25	1042
SFBS OT Age-2	0.97	0.81	1.17	3	146
FMWT	0.90	0.74	1.09	1	323
20-mm	1.02	0.68	1.53	1	5
Meta-Analysis	0.92	0.86	0.99	50	1834

Figures

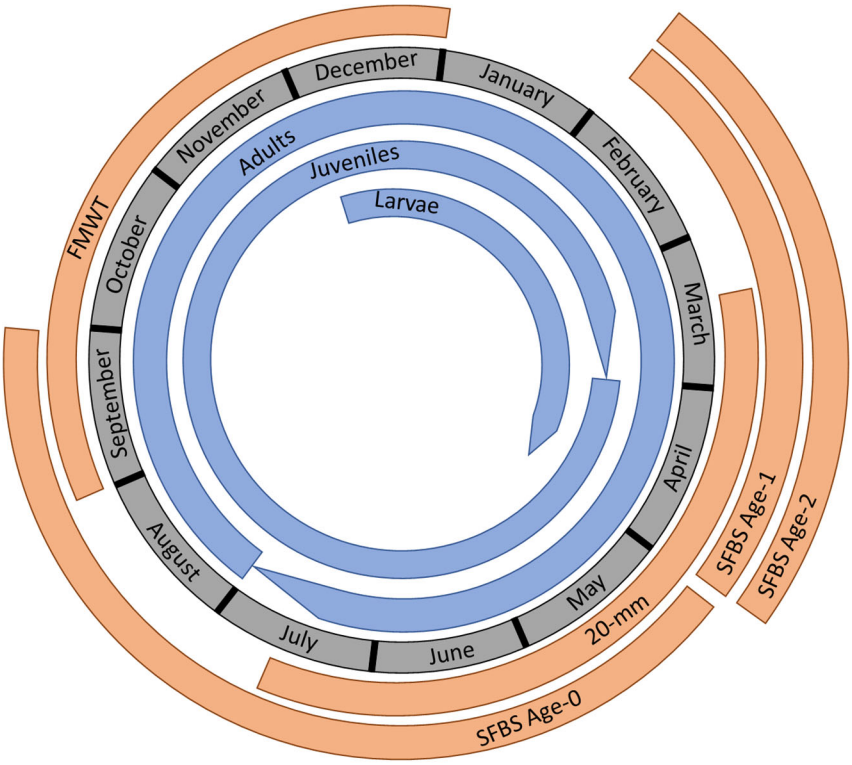


Figure 1: Presence of Longfin Smelt life stages (blue; inside of the ring of months) in the estuary and the timing of monitoring surveys (orange; outside) throughout a calendar year. Bands for life stages taper at the end of the time frame to differentiate the beginning from the end of the bands.

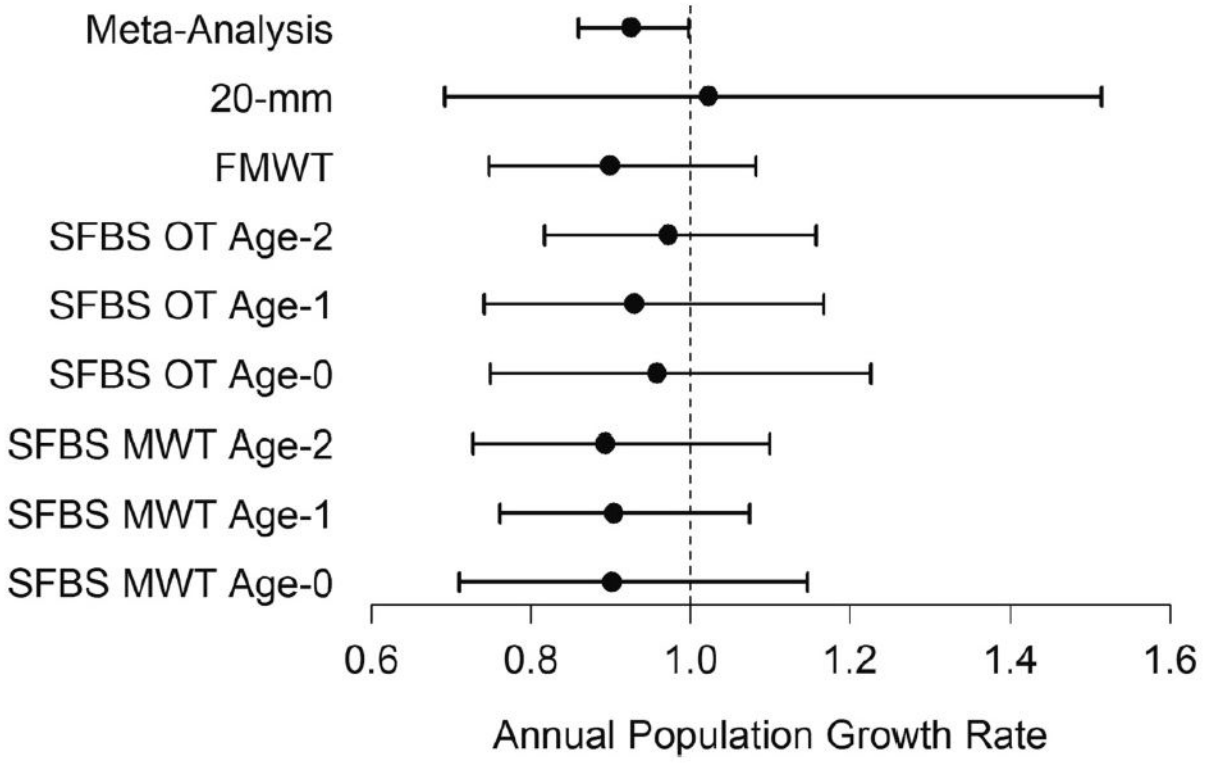


Figure 2: Mean population growth rates from several monitoring programs and a meta-analysis of the mean. Calculations were made using all available years for each survey, based on a count PVA framework. Error bars are 95 % confidence limits.

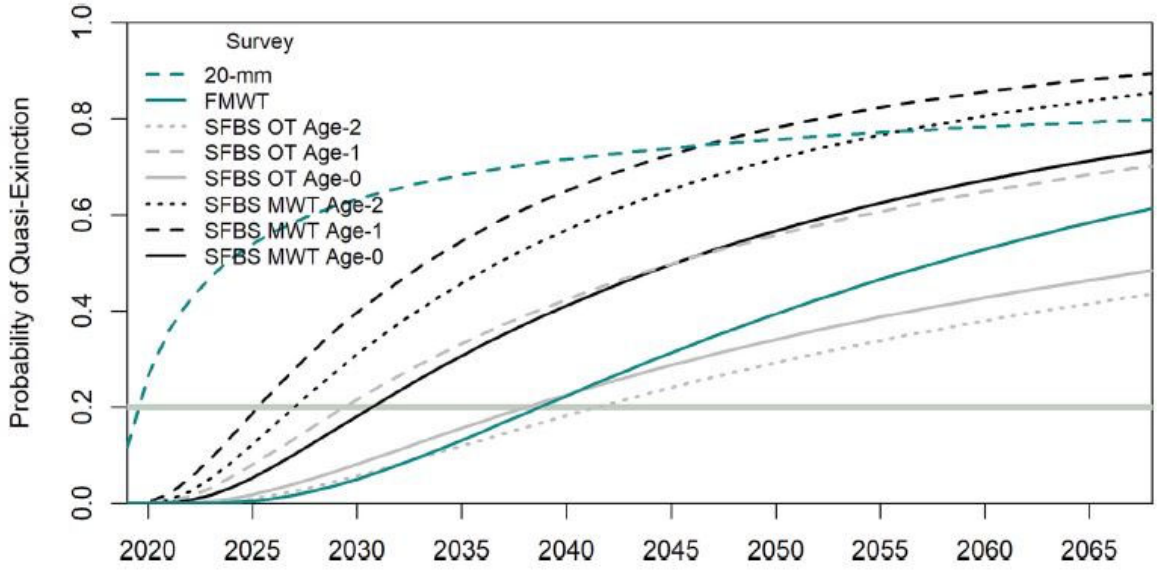


Figure 3: Probability of quasi-extinction for several surveys that report population indices for Longfin Smelt. Values for quasi-extinction thresholds are defined in Table 3. Estimates of population growth rates and variability were derived from all available years for each survey. The wide, horizontal line highlights a 20% probability of quasi-extinction.

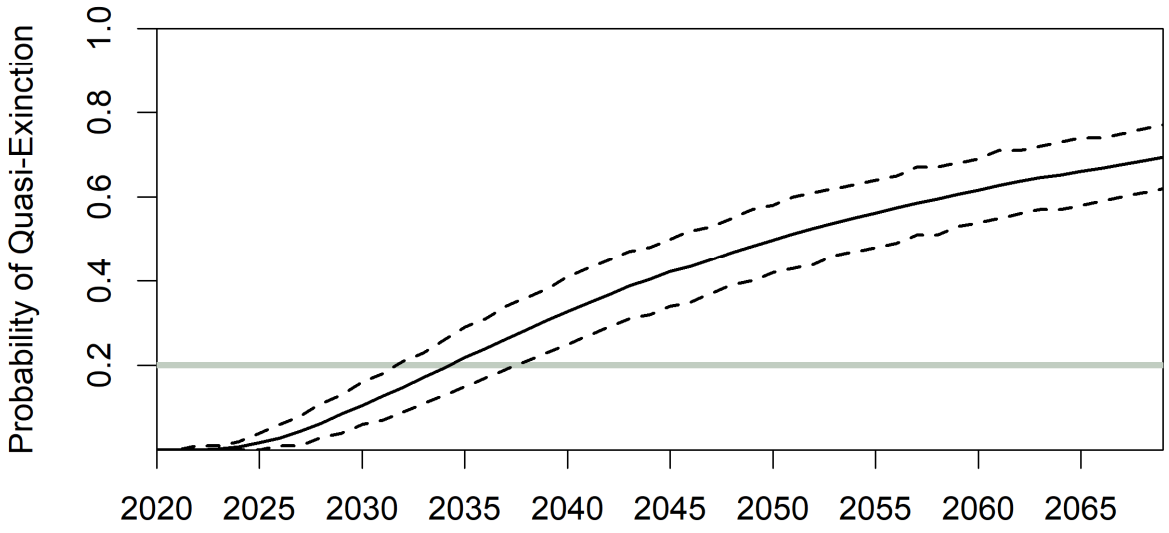


Figure 4: Mean probability of quasi-extinction (solid line), as calculated from the values in the meta-analysis, with bootstrapped 95% confidence bands (dashed lines). The wide, horizontal line highlights a 20% probability of quasi-extinction.

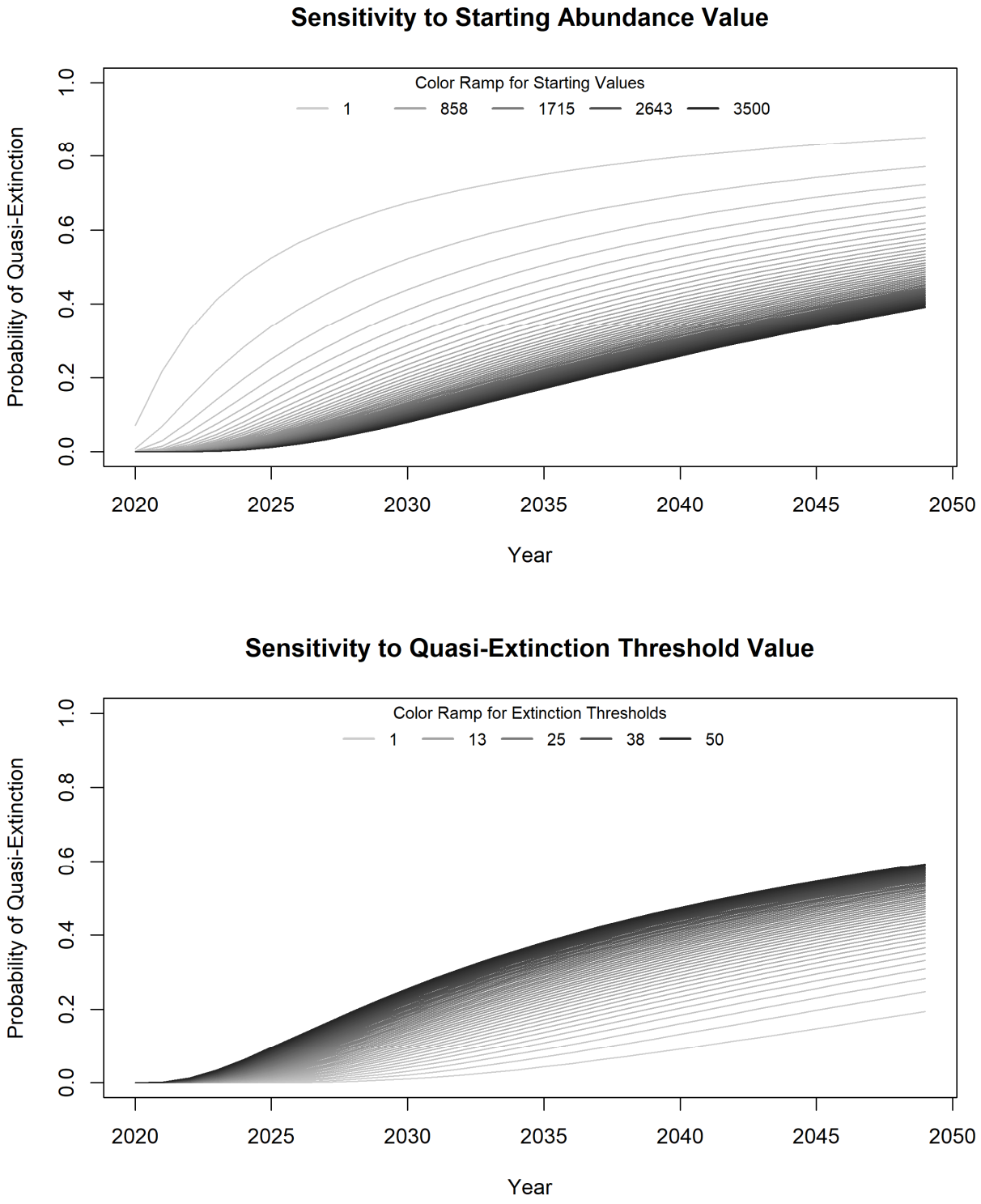


Figure 5: Sensitivity of the probability of quasi-extinction to the choice of starting abundance value (top) and quasi-extinction threshold (bottom). Grayscale lines represent a range of choices.

Supplemental Information

Table S1: Model comparisons for different density-dependence formulations.

Survey	Model	df	AICc	Converged	r	R_p	K	K_p
FMWT	Beverton-Holt	3	212.12	TRUE	0.14	0.42	-680.10	0.40
FMWT	Ricker	3	213.03	TRUE	-0.21	0.47	8597.13	0.44
FMWT	Linear	3	213.03	TRUE	0.00	0.14	NA	NA
mwt.0	Beverton-Holt	3	160.15	FALSE	1003.09	1.00	86.04	0.85
mwt.0	Ricker	3	159.17	TRUE	-0.19	0.64	42811.97	0.62
mwt.0	Linear	3	159.17	TRUE	0.00	0.22	NA	NA
mwt.1	Beverton-Holt	3	122.52	FALSE	-286790.80	1.00	-74.62	0.89
mwt.1	Ricker	3	121.43	TRUE	-0.27	0.31	52207.57	0.36
mwt.1	Linear	3	121.43	TRUE	0.00	0.29	NA	NA
mwt.2	Beverton-Holt	3	101.12	TRUE	0.83	0.43	53.10	0.64
mwt.2	Ricker	3	102.37	TRUE	0.08	0.78	1010.97	0.70
mwt.2	Linear	3	102.37	TRUE	0.00	0.63	NA	NA
ot.0	Beverton-Holt	3	159.76	FALSE	385.39	1.00	66.88	0.93
ot.0	Ricker	3	159.14	TRUE	-0.09	0.78	17670.29	0.76
ot.0	Linear	3	159.14	TRUE	0.00	0.39	NA	NA
ot.1	Beverton-Holt	3	147.24	TRUE	0.19	0.47	-4780.19	0.42
ot.1	Ricker	3	148.16	FALSE	NA	NA	NA	NA
ot.1	Linear	3	148.10	TRUE	0.00	0.91	NA	NA
ot.2	Beverton-Holt	3	115.15	TRUE	0.02	0.90	-138.66	0.89
ot.2	Ricker	3	116.97	FALSE	NA	NA	NA	NA
ot.2	Linear	3	115.75	TRUE	0.00	0.39	NA	NA
20-mm	Beverton-Holt	3	65.63	FALSE	1354.41	1.00	25.26	0.81
20-mm	Ricker	3	67.87	FALSE	NA	NA	NA	NA
20-mm	Linear	3	65.32	TRUE	0.00	0.31	NA	NA

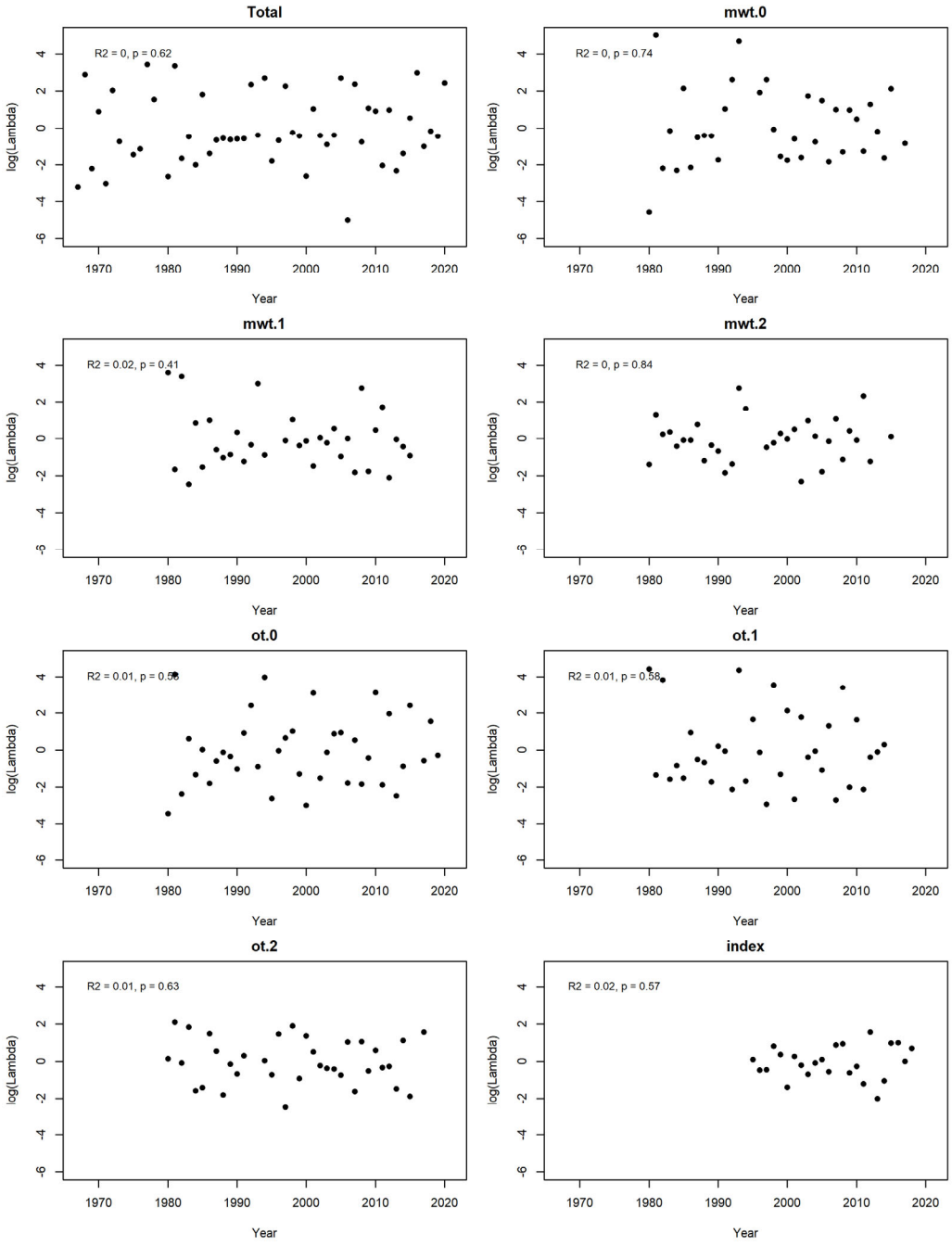


Figure S1: Graphs and regression statistics for checking the assumption that population growth rates to not change over time.

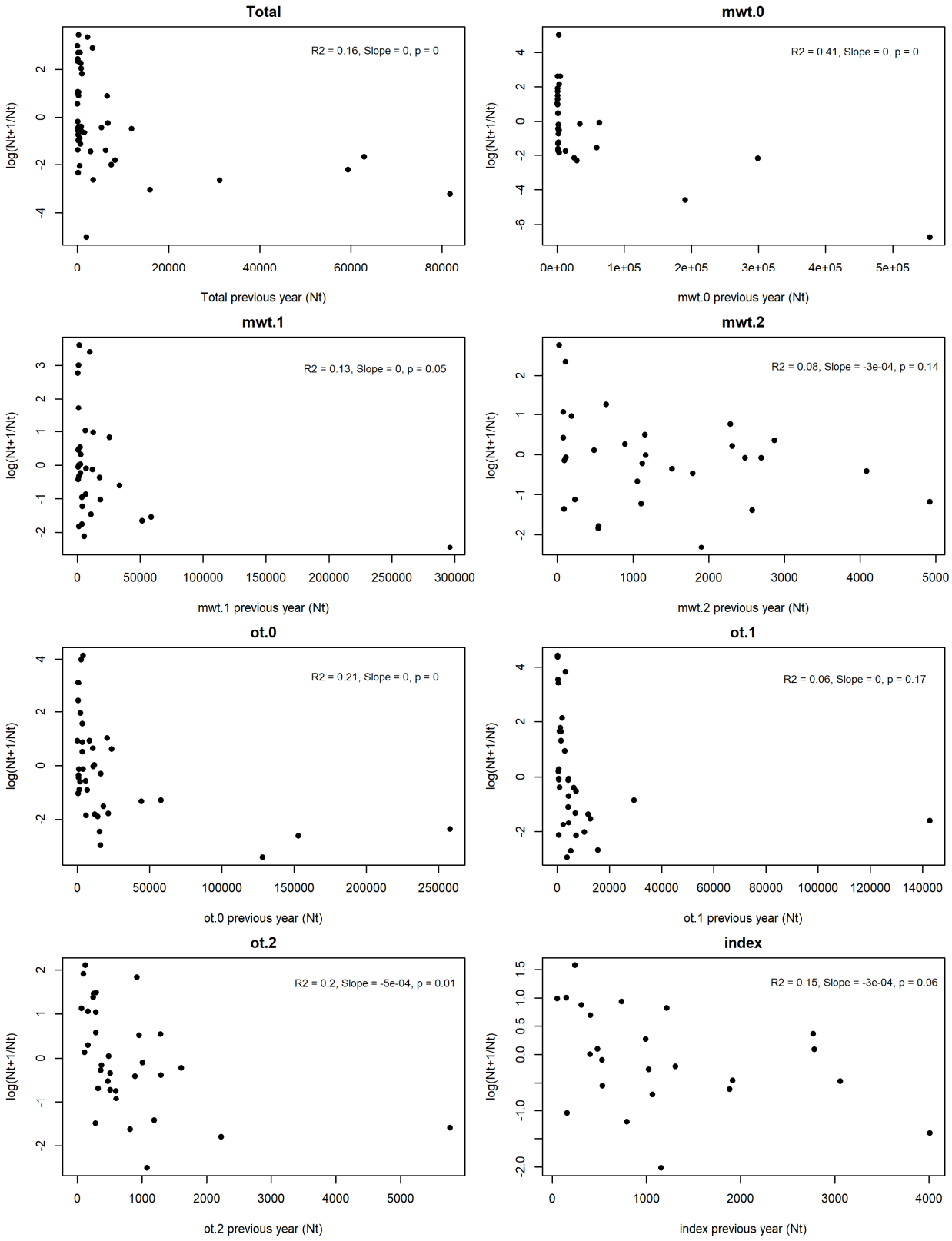


Figure S2: Graphs and regression statistics for checking whether the population growth rates are density dependent.

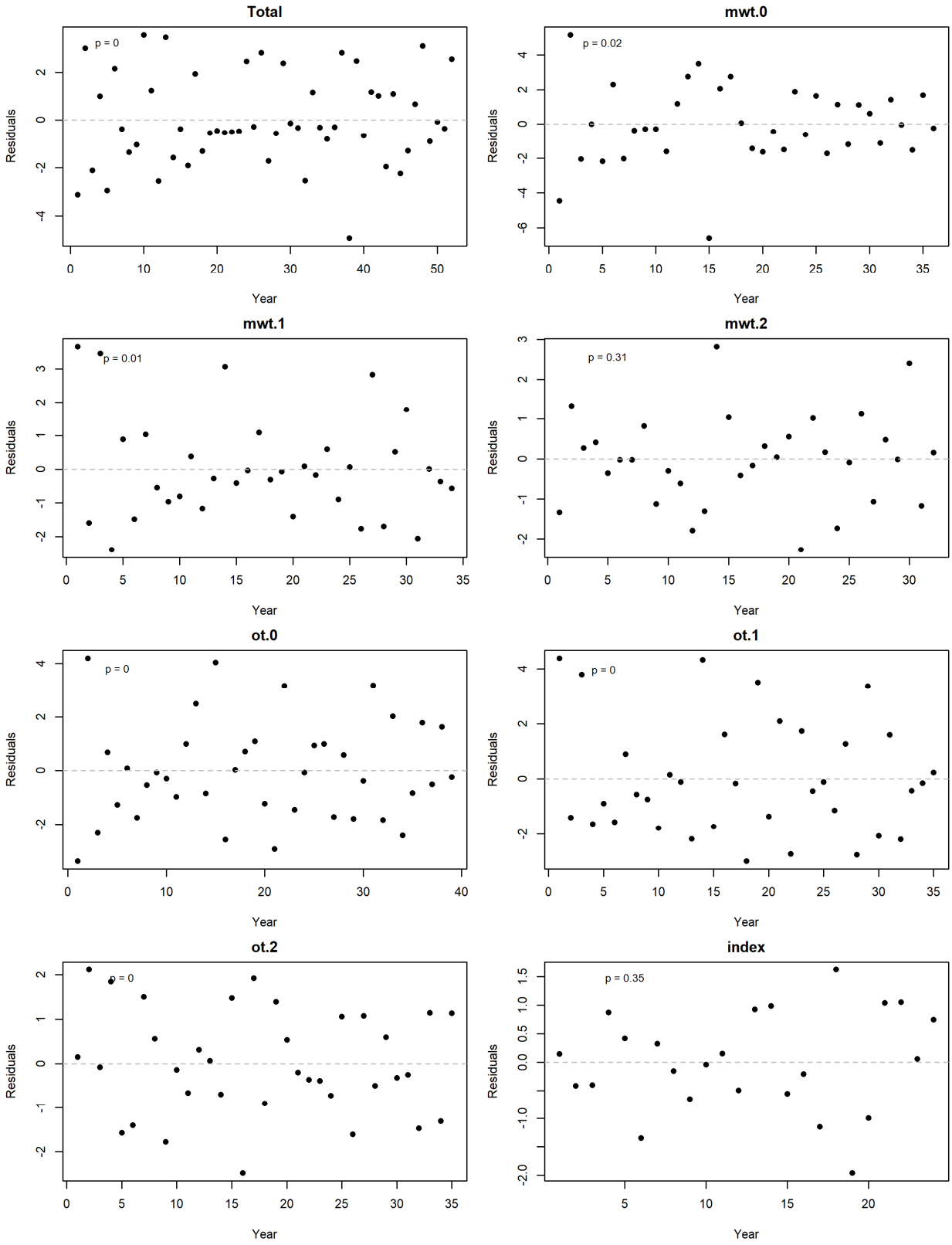


Figure S3: Graphs and statistics for checking for environmental autocorrelation. Results of Durbin-Watson tests for first-order autocorrelation are represented as p-values printed on each panel.

Technical Appendix C: Separating Year Classes of Longfin Smelt by Patterns in Length at Date Data

Version 1.2.0 – May 17, 2022

Vanessa D. Tobias, USFWS

Introduction

Longfin Smelt (*Spirinchus thaleichthys*) can live for several years; most are thought to spawn and die at age-2, but some fish live to age-3 (Dryfoos 1965, Moulton 1974, Moyle 1976). Early estimates of maximum size were about 150 mm TL (Miller and Lea 1972), but larger Longfin Smelt have been observed in the San Francisco Estuary. The smallest Longfin Smelt recorded in the San Francisco Estuary are about 5 mm. It is necessary to estimate the ages of fish that are caught in monitoring surveys so that it is possible to calculate age-specific abundance indices and track population dynamics.

Scientists commonly use size to estimate ages for fish. Existing criteria for separating Longfin Smelt into age classes using information about the length of the fish and the month in which it was caught were developed using catches from 1980–1989 (Baxter 1999). These length criteria are used here for calculating the abundance index values for the San Francisco Bay Study. However, the use of static length cutoff values for determining age at specific dates would be problematic if growth rates vary annually or by cohort.

Here I use patterns in the length frequency distributions to reclassify each cohort. The methods are conceptually similar to those described by Baxter (1999), but using data from additional years and surveys allowed me to estimate cutoff values for each month and year combination rather than a single, static set of cutoff values.

Methods

Length Data

Several of the Interagency Ecological Program long-term monitoring surveys collect Longfin Smelt and record their forklengths. I compiled these data and used patterns in observed forklength frequencies in each calendar month to identify age classes. The surveys used in this technical note were: San Francisco Bay Study's (SFBS) long-running otter trawl and midwater trawl, the SFBS larval surveys which occurred in the 1980s, the 20-mm survey, Fall Midwater Trawl (FMWT), Enhanced Delta Smelt Monitoring program's 20-mm and Kodiak surveys, and the Smelt Larval Survey (SLS). Data used in this technical note were limited to the years 1980 through 2020 because these were the years for which the most monitoring studies were active. Details on survey methods can be found in Honey et al. 2004, but some relevant details are described below.

The long-running SFBS survey collects data throughout the year at locations ranging from South San Francisco Bay to the Sacramento-San Joaquin Delta (the Delta). It has the broadest geographic range of the surveys. The SFBS collects fish using two gears: an otter trawl that samples the bottom of the water

column and a midwater trawl that takes an oblique sample throughout the water column. The SFBS truncates length data; it does not record fish less than 40mm. The 20-mm study provides data on smaller fish in the spring. The SFBS larval survey sampled for a shorter run of years (1980–1989) and it used a 505 µ mesh plankton net to sample larval fish. The data used in this technical note are only those fish that were identified as Longfin Smelt. It does not use any unidentified Osmerids, although these do exist in the dataset.

The FMWT uses a net that is similar to the SFBS midwater trawl gear. Geographically, the FWMT covers eastern San Pablo Bay to the Delta and as the name suggests, it generally samples in September through December. EDSM was designed to follow the life cycle of Delta Smelt, but it regularly catches Longfin Smelt as well. It began in 2016. EDSM uses a Kodiak net for most of the year, except during the spring when it uses a 20 mm net. It uses a geographically stratified sampling design to collect data from eastern San Pablo Bay to the Delta, although the sampling frame changes seasonally to accomplish the goals of estimating abundance and distribution of Delta Smelt. The Smelt Larval Survey began in 2005 and its geographic coverage includes Carquinez Strait through the Delta, and sometimes the Napa River. The SLS uses 505 µ mesh plankton nets, similar to the SFBS larval survey.

Existing Length at Date Cutoffs

The SFBS program uses length at date charts to help distinguish between age classes of Longfin Smelt for calculating age-specific abundance indices (**Table 4**). These are the cutoff values that were developed by Baxter (1999) using data from 1980-1989. Baxter (1999) observed that these cutoff values provided good separation of age-0 and age-1 fish but could only approximately separate age-1 fish from age-2 fish because the lengths of these age classes overlapped considerably. He also noted the apparent existence of older age classes in the data. These length at date criteria were used as rough guides for the age classification process described by this technical note to aid in automated labeling of cutoff values for each month.

Analysis

Changes in the fork length distribution of two year-classes of Longfin Smelt can be seen in monthly frequency distributions of fork lengths observed during one calendar year (e.g., 1983; **Figure 6**). In this example, from January to May, only the year class that hatched in 1982 is visible and the cluster of length measurements shows growth over time. From June through December, the 1983 year class appears as a cluster of shorter fork length measurements. Visually it is fairly easy to differentiate these year classes and follow the patterns in apparent growth over time; however, it is not obvious how to accomplish this task numerically with the data for a long-term dataset. This technical note focuses on the development of a method to automate the separation of year classes using only length and date information. The peaks of the density plots for each month in a year class identify the modal length for that month (**Figure 7**). This is a visual representation of the expected length for a fish belonging to the identified year class that is caught in that month.

All calculations and analysis were conducted using pr R (version 4.0.4 “Lost Library Book”; R Core Team 2021). I estimated the univariate density of fork lengths for each month in each year using a Gaussian kernel (stats::density). The bandwidth that I selected (6.5 mm) was based on visual comparisons of plots. This bandwidth provided adequate smoothing to the density plots, while maintaining the ability to visually distinguish groups of fork lengths within a month. I wrote an algorithm that identifies minimums and maximums in the kernel density plots by taking the first derivative of the density function and identifying places where the sign of the slope changes. Minimum values represent cutoffs between size classes, whereas maximums are the centers of the lengths for an age class. Once clusters of lengths were identified, I classify the kernel density minimums by which age classes they separate, based roughly on the length at age criteria developed for the Bay Study. In this way, the SFBS cutoffs inform the identification of the age classes without influencing the selection of the cutoff values themselves.

Not all age classes are apparent in every month and calendar year combination. To fill these in and adjust any very unusual cutoff values, I used a flexible regression technique. This was a hierarchical generalized additive model (HGAM) based on the GI model formulation from Pedersen et al. (2019). This model formulation has a shared global term (G) that creates a similar shape across all the year classes. The individual term (I) allows individual years to deviate from the shared global term, but applies a penalty to years that are very different. This regression method fills in gaps where no information exists for a cohort using information from all years and the pattern for that particular year. The model was fit using the mcgv package (Wood 2011).

After creating the cutoff values, I classify lengths into age-classes using the specific cutoff values that were identified for that combination of month and year. I then rearranged the resulting data to create cohorts. For example, an age-1 fish that was caught in 2010 was part of the cohort of fish that hatched in 2009.

Validation

I used k-fold validation to test the consistency of the classification method (Hastie et al. 2009, Ch. 7, pp. 241–247). I split the dataset into five folds (or partitions). The first fold was held out to be used as a testing dataset and the other four folds were used as training datasets. I repeated the methods described above on each of the four testing datasets to produce four unique sets of length at age cutoff values. I used these cutoff values to classify the lengths in the testing dataset, which produced four sets of age classifications for each length value in the testing dataset. I compared agreement across these age classifications to assess consistency.

Results

Length at Date Cutoffs

Maximum lengths were calculated for each combination of month and year in the dataset. Appendix B shows a subset of the cutoffs. **Figure 8** shows an example of a single year class or cohort (fish that hatched in 1982), as stacked kernel density plots for individual months. The year class panel shows the kernel densities for fork lengths of fish that were classified as belonging to the 1982 year-class. This plot

highlights the pattern in apparent growth of the year class over potentially three years (months 1–36). Months 13 – 24 correspond to calendar year 1983, i.e., the example calendar year plot. Lines are missing in the graph for months when no fish that were caught during that month were identified as belonging to the 1982 year-class.

The maximum lengths for age-0 and age-1 fish used by the SFBS are within the range of maximum length at age that was calculated for individual years when the KD method is applied to the length at date data from multiple surveys (**Figure 9**). There is some variation in the cutoff values for each month using the KD method that isn't captured by using a single value across all years. There is less separation between the three age classes late in the calendar year. Lines for individual years corroborate the SFBS maximum length at age pattern that shows a steeper slope in the first six months of the age-0 year, followed by a flattening of the cutoff curve. The SFBS cutoff values that separate age-0 from age-1 fish early in the year fall in the lower part of the range of the cohort-specific cutoff values for that time of year. For many cohorts, some fish that the SFBS values identify as age-1 would be identified as age-0.

Using the adjusted cutoff values to assign ages to each length measurement allows the cutoff values to borrow information from adjacent months to make more realistic assignments than would be possible using the nadir values identified by the densities of individual months. For example, in the length data form 1997 (**Figure 10**), using the density from April alone, the mass of lengths greater than about 35 mm might be classified as all age-1 fish. By incorporating information from before and after that month, it becomes apparent that some of those fish should be classified as age-2+ because there are centers of mass in the age-2+ range for the previous three months. This shows how the regression technique borrows information. Without it, we might think that there were no age-2 fish present and that the age-1 fish were unreasonably large during that month. This use of adjacent information is more important for the cutoff between age-1 and age-2+ than for the separation between the age-0 and age-1 classes because of the differences in the degree of separation between the age groups. There is generally more complete separation between the 0 and 1 than between 1 and 2. This also means that the precision of the 0–1 cutoff values is less important for accurate age discrimination than precision in the 1–2+ cutoffs. There is less data for the older ages, though, as abundance is higher for younger fish.

In the length data collected in the SFE, there was generally good visual separation between the lengths of age-0 and age-1 fish (**Figure 10**). Distinguishing age-1 and older age classes was not always possible without the help of density plots and code to separate the masses of lengths. In Lake Washington studies, distinguishing year classes was not problematic because there was no overlap in sizes of fish belonging to age-0 and age-1 classes (Dryfoos 1965; Chigbu and Sibley 1994).

Validation

The kernel density method and the SFBS age classes placed lengths into the same age classes in 99.4% of cases (**Table 5**). Most of the discrepancies between the two methods were in discriminating between age-1 and age-2+ fish. In the k-fold validation tests, the age classifications for lengths in the testing data fold were highly consistent across the classifiers that were trained by the data in the four training folds (**Table 6**).

Discussion

1. This paper updates the existing static length at date cutoff values for Longfin Smelt. Results here are similar to the existing cutoffs, but this method will allow cutoffs to be more responsive to cohort- or year-specific differences in observed growth patterns.
2. This method is a mathematical way to approximate the visually intuitive process of separating age classes using length frequency data.
3. Classification of length data into consistent age classes by algorithms trained on different subsets of the data indicate that the method is robust.
4. Separating age-0 from age-1 fish was easier than separating age-1 from age-2. This is similar to what other studies found.
    a. Chigbu & Sibley 1994: Bimodal distribution of lengths within a year with no overlap used to separate 1989 and 1990 cohorts.
    b. *Dryfoos 1965: "Determination of the age of the longfin smelt was simplified because no overlap in length frequency between year classes occurred." In the Lake Washington freshwater population, and applied to the separation between age-0 and age-1 fish.*
5. From MAST Biology section, to support age class cutoffs: *Little information has been published on the growth of Longfin Smelt in the San Francisco Estuary. Using 20-mm Survey length data for 1995 through 2004 and analyzing for apparent growth, Souza et al. (2006) found that larval through young juvenile Longfin Smelt grew at a mean apparent rate between 0.12 and 0.23 mm/day, and that growth was not related to outflow. These same authors reported that age-0 Longfin Smelt caught by the Fall Midwater Trawl achieved December (age-10 months) mean length that varied between 50 and 70 mm FL. In Lake Washington, Longfin Smelt at age-10 months averaged from 56 to 82 mm FL in the 1960s (converted from SL and Figure 3 in Moulton 1974). Using the Fall Midwater Trawl range of mean lengths at 10 months and assuming a February 7 mean hatch date (i.e., mid-way through the monthly sampling period) and a December 7 mean date of capture results in annual apparent growth rates varying between 0.20 to 0.23 mm/day. This represents relatively slow growth.*
6. Caveats
    a. This analysis does not include any information about fish with known ages. Data from otoliths would be necessary to confirm ages of fish. This analysis implies ages from patterns in length data using expert opinion as a guide.

Potential next steps

1. Length data that have been classified using the method developed in this technical note will be used to investigate patterns in apparent growth of Longfin Smelt in the San Francisco Estuary. (See LFS TN 3 by Bryan Matthias.)
2. Investigate whether cutoff values are stable over time and of how much interannual variation there is. (This question was identified in the comments on the LFS MAST Biology section.)
3. Length distributions might help us investigate hypotheses about different life history strategies. Length distributions have multiple modes in some years, which might lead us to looking at spatial patterns as well.
4. Additional validation may be possible in the future if data become available for fish that were aged using other techniques such as reading otoliths.

5. Try other kernel density functions. Triangular might be good for emphasizing the cutoffs. (This is currently not a priority because the existing method is largely consistent.)
6. Consider investigating whether there are patterns in the non-matching age classifications from the validation procedure. (There are very few of them, though, so that isn't a priority yet.)

R code for the analyses and graphs can be found at _____

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Tables

Table 4: Length at age cutoffs used for SFSB data.

Calendar Month	Minimum Length	Age-0 Maximum Length	Age-1 Maximum Length
1	40	40	90
2	40	42	93
3	40	46	96
4	40	52	100
5	40	59	105
6	40	67	108
7	40	71	111
8	40	75	114
9	40	80	117
10	40	83	120
11	40	85	122
12	40	87	124

Table 5: Confusion matrix comparing age class determinations made by the traditional SFBS length at date criteria and the kernel density (KD) method.

		KD Age Classes		
		0	**1**	**2+**
SFBS Age Classes	**0**	215085	139	0
	1	347	32218	482
	2+	0	465	2922

Table 6: Proportion of age classifications on the length values in the testing data fold (fold 1) that match among kernel density-based algorithms trained on four training data folds (folds 2–5).

	2	**3**	**4**	**5**
2	1.000	0.993	0.995	0.994
3	0.993	1.000	0.993	0.993
4	0.995	0.993	1.000	0.995
5	0.994	0.993	0.995	1.000

Figures

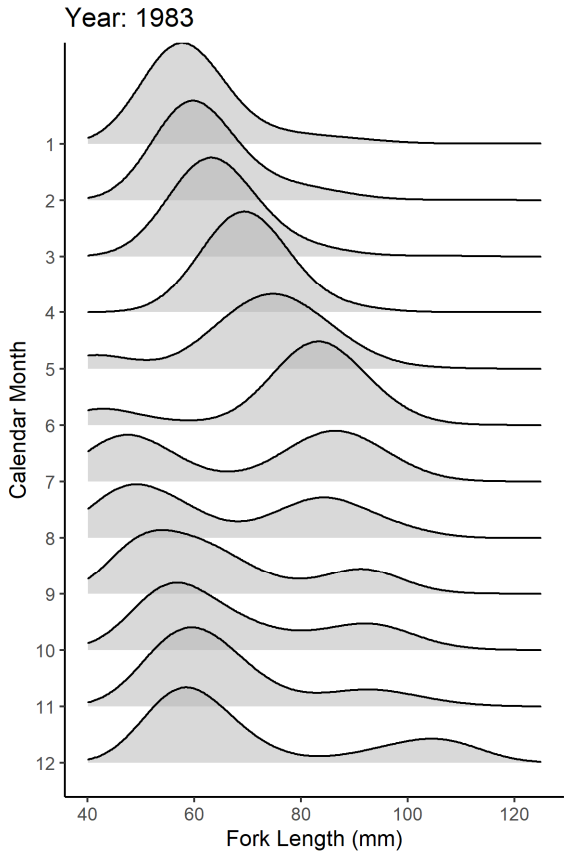


Figure 6: An example of the monthly length distribution of two year classes of Longfin Smelt over one calendar year (1983), as stacked kernel density plots for individual months. The bandwidth for kernel density estimation was 6.5 mm.

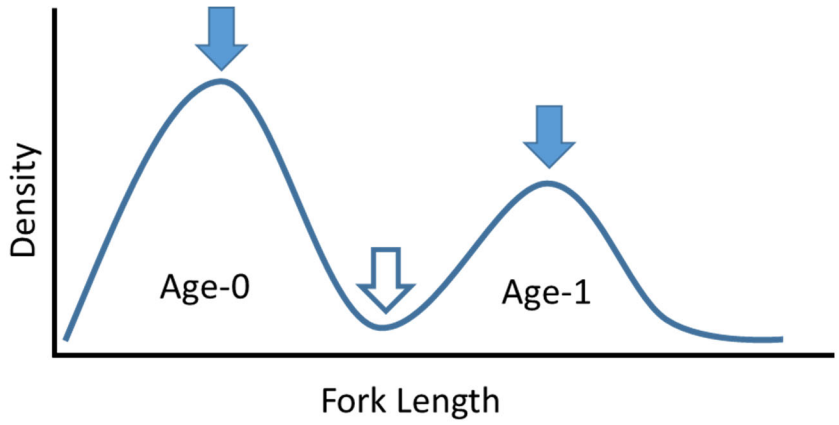


Figure 7: Conceptual diagram of using features of the kernel density plot of fork length to separate year classes of Longfin Smelt that were collected during a calendar month with two age classes present. Filled arrows identify the center of the distributions and the hollow arrow identifies the separation between the two age classes.

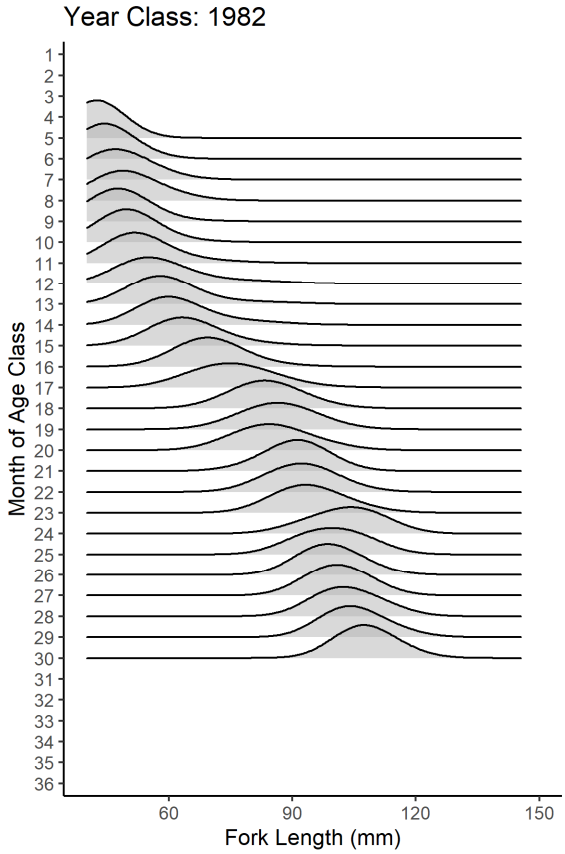


Figure 8: An example of the monthly length distributions of one year class of Longfin Smelt (fish that hatched in 1982) over 36 months, as stacked kernel density plots for individual months. The bandwidth for kernel density estimation was 6.5 mm.

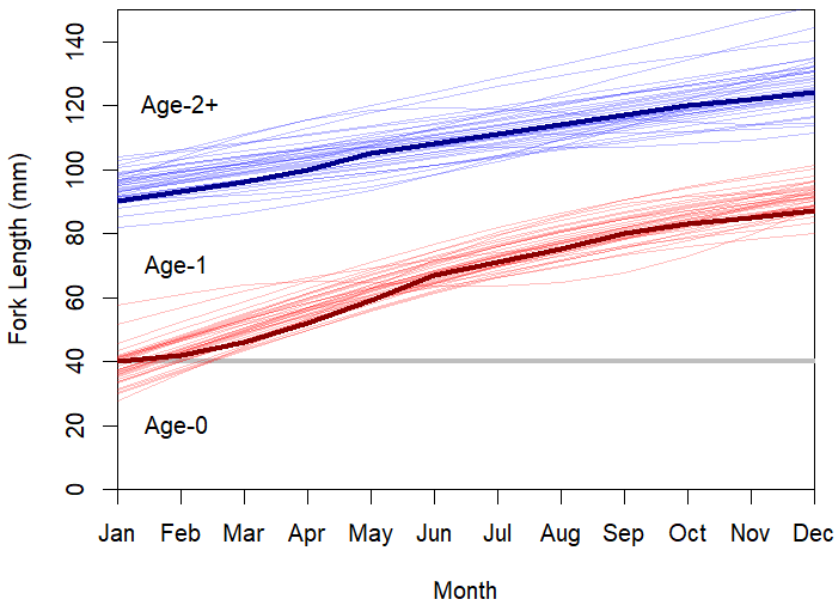


Figure 9: Length at date cutoffs for SFBS (thick lines) and as calculated in this paper (thin lines).

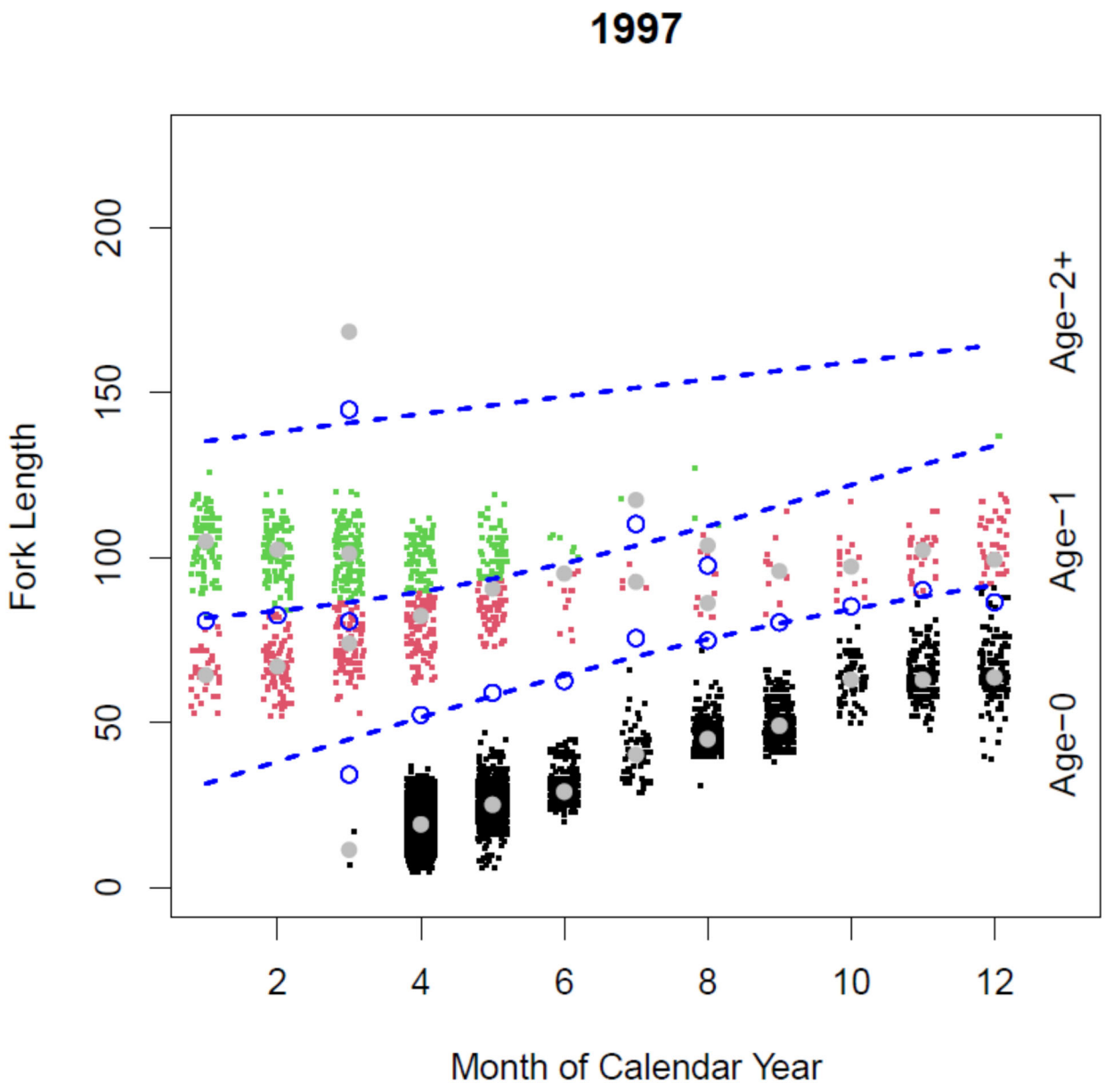


Figure 10: Separation of fork lengths into age classes in a calendar year (1997 as an example).

Technical Appendix D: Assessment of Environmental Factors on Apparent Growth in Bay –Delta DPS Longfin Smelt

Version 2.1.0 – May 16, 2022, 2022

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Introduction

In fishes, understanding how somatic growth changes over time is important because body size and condition influence population growth rates as determinants of survival, fecundity, and reproductive schedules (Sterns 1992; Charnov 1993). Fish size is an important factor influencing survival via susceptibility to predation (see Mittelbach and Persson 1998 for review on selectivity of predators) and natural mortality generally decreases as individuals attain larger body sizes (Pauly 1980; Lorenzen 1996, 2000). Thus, conditions supporting faster growth can increase survival, especially during larval and juvenile stages when mortality rates are high. Further, fecundity for many fish species increases proportional to size (i.e., by weight or length, approximately cubed; Beverton and Holt 1957; Ricker 1975; Walters and Martell 2004). For semelparous spawners, larger sizes-at-maturation can result in greater egg production and/or larger eggs (see Johnston 1997 and references within). Either of these effects can influence total reproductive output of a population by increasing the total number of eggs produced and/or offspring survival--numerous studies have identified positive relationships between eggs size and size-at-hatching, better feeding success, faster growth, etc. (see Johnston 1997 and references within). Thus, quantifying changes in somatic growth can serve as an indicator for changes in population growth rates and may be used to better inform management towards subsequent recovery efforts.

Fish populations within many estuaries globally have been declining due to a variety of causes, with overfishing and environmental degradation identified as two major drivers in population declines (reviewed in Whitfield 1999; Lotze et al. 2006; Lotze 2010). Similar to estuaries around the globe, the San Francisco Estuary (SFE) has undergone many changes over the past century and a half (for a more detailed overview of changes see Sommer et al. 2007; Cloern and Jassby 2012) and it has been transformed into a major water transportation hub that provides drinking water for over 25 million people and irrigation water for 750,000 acres of farmland (CDWR 2013). The SFE is the largest estuary on the west coast of North America with a watershed that drains roughly 40% of California's surface (Jassby and Cloern 2000). Due to flow management in the SFE watershed, roughly 39% of the unimpaired runoff is consumed or diverted annually (reaching as high as 65%; Cloern and Jassby 2012). These changes have had major impacts on species that rely on the SFE and its watershed for all or part of their life cycle (e.g., winter and spring-run Chinook *Oncorhynchus tshawytscha*, Steelhead *O. mykiss*, Delta smelt *Hypomesus transpacificus*, and green sturgeon *Acipenser medirostris* are currently federally listed as endangered or threatened in the watershed).

Several long-term monitoring studies have documented declines in SFE productivity in the late 1980s (Kimmerer and Orsi 1996; Kimmerer 2002) and several fish populations crashed in the early 2000s (referred to as the Pelagic Organism Decline [POD]; Sommer et al. 2007). The SFE Distinct Population Segment of Longfin Smelt (*Spirinchus thaleichthys*; hereafter referred to as LFS) was among the POD populations to collapse. In the SFE, LFS was once among the most abundant pelagic fishes (Baxter 1999)

and the system historically supported a commercial fishery. There have been several hypotheses put forward for the LFS population declines and assessing long-term LFS growth patterns may shed light on these hypotheses and identify if changes in growth patterns have contributed to population declines.

Given the importance of somatic growth, the 2020 Longfin Smelt Science Plan (https://water.ca.gov/-/media/DWR-Website/Web-Pages/Programs/State-Water-Project/Files/ITP/ITP-Longfin-Science-Plan_SWP_12232020_-FINAL.pdf) identified factors affecting growth as a science priority to help understand factors affecting population dynamics, improve management tools, and developing informative predictive modeling tools. Here we assess a suite of abiotic (i.e., temperature, outflow, conductivity) and biotic (i.e., abundance, prey availability) factors that may affect LFS apparent growth rates. We acknowledge that other factors may also be important, and these may be addressed in future versions of or extensions of this work. To assess LFS apparent growth, we modified the Lester et al. (2004) biphasic growth model to estimate the impacts of factors on somatic growth patterns based on a similar model developed by the author (Matthias et al. 2016). Our primary objective was to quantify the strength, direction, and potential interactions of abiotic and biotic factors influencing LFS growth. Our secondary objective was to assess systematic or long-term changes or patterns in LFS growth. Quantifying factors that influence growth allows us to investigate growth and its correlates as a potential mechanism for the abundance-outflow relationship (Stevens and Miller 1983; Jassby et al. 1995; Kimmerer 2002). For example, studies have detected changes in the abundance-outflow relationship associated with the establishment of the invasive clam *Potamocorbula amurensis* (Kimmerer 2002) and another attributed to the POD after 2002 (Thomson et al. 2010), but it is unknown if growth patterns changed during these periods.

Methods

Longfin Smelt Length Data

Longfin Smelt length data were obtained and combined from multiple long-term monitoring surveys (Table 1). These surveys spanned from 1981–2020 and contained year-round sampling programs. The San Francisco Bay Study (SFBS), focused on large juveniles and adult fish, had year-round sampling from 1981–2015, covered almost the whole SFE from South San Francisco Bay through the western Delta, and sampled 102,776 fish (mean length of 59 mm and range of 40–197 mm). The 20 mm Survey, designed to capture larval and juvenile fish, sampled from March-August from 1995–2019, covers from eastern San Pablo Bay through the Delta, and has sampled 182,103 fish (mean length of 19 mm and range of 3–183 mm). The Fall Midwater Trawl (FMWT) surveys, designed to capture juvenile and adult fish, mainly surveys from September-January (but has sampled year-round on occasion) from 1981–2017, is similar is spatial coverage to the 20 mm Survey, and has sampled 18,643 fish (mean length of 71 mm and range of 15–185 mm). The Enhanced Delta Smelt Monitoring (EDSM) surveys started recently (2017–2020) using 20 mm nets for targeting the larval stage and Kodiak trawls for juvenile/adults. The EDSM survey is year-round and spatial coverage varies by season (usually Suisun Bay through the north Delta), but switches between the two sampling methods, and has caught 1,297 (mean length of 21 mm and range of 6–43 mm) and 649 (mean length of 73 and range of 30–132 mm) in the 20 mm nets and Kodiak trawls respectively. Finally, the Smelt Larval Survey (SLS) survey captured 5,713 larval and small juveniles (mean length of 8 mm and range of 3–32 mm) from 2005–2007 from January through July and had a

distribution similar to the 20 mm Survey. A total of 311,166 LFS lengths were used in this analysis. A single obvious outlier (length = 282 mm) was removed from the analysis.

Age measurements were not obtained for fish caught during these surveys. Instead, age (in years) was assigned based on length-frequency distributions (see Tobias 2022 for details). Ages were converted to fractional monthly age using the date-at-capture and assuming a birthdate of January 1 each year. Individuals were assigned to cohorts (i.e., all individuals born during the spawning season in a given year), based on their assigned age and capture date.

Abiotic Environmental Data

A suite of variables was used to test for the effects of environmental conditions on LFS growth (Figure 1). These parameters were chosen based on a combination of data availability and expert opinion. Environmental variables related to outflow were obtained from the DayFlow model and averaged over a given month (https://data.ca.gov/dataset/dayflow; daily predictions from 1956–2020). Variables representing temperature (in degrees C) and conductivity (in uS/cm) were obtained from the California Data Exchange Center (CDEC; https://cdec.water.ca.gov/riv_flows.html) at the Martinez (MRZ) long-term automated monitoring station operated by the CA Department of Water Resources Division of Environmental Services. The station is located at 38.03^{o}N, 122.14^{o}W in the Carquinez Straight between San Pablo Bay and Suisun Bay and was chosen for its centralized location that seems representative of general conditions throughout the San Francisco Estuary and availability of a long-term dataset (1983–present). While a single location might not be representative of the exact conditions each fish experience at any given time, the conditions recorded at a centralized location should be correlated with conditions across the SFE and changes in abiotic conditions here (temperature, salinity, conductivity, etc.) should indicate changes across LFS habitat within the SFE (although not necessarily correlated with marine habitats).

Prior to use in the model, environmental data underwent several processing steps. The first step, when possible, was to obtain monthly mean estimates from 1979–2020. Preliminary model fitting using monthly minimum and maximum observed environmental data did not converge properly and were omitted from the analysis. This date range encompasses all age classes for each cohort that were present in the LFS dataset. We then searched for obvious outliers and removed those values (i.e., set to NA; note that this occurred two times with the DayFlow dataset when estimating the monthly minimum values). Finally, negative values for outflow (six instances) were set to 0.01 to allow for logging of minimum outflow in the model (these values were used to calculate the monthly mean discharge, but were changed when processing the minimum observed values and did not impact the analysis because minimum monthly covariates were not analyzed). Due to the nature of these environmental datasets, there were also occurrences of missing values (65–66 instances with temperature and conductivity, with unobserved values prior to 4/1983 and 11/2007–7/2008 accounting for most missing values). We used the monthly geometric mean across all years to fill in missing data points (e.g., a missing temperature in February 1980 was filled in with the geometric mean of February temperatures from all available years). While this does not capture the unobserved variability within a given year, this method allowed us to capture the seasonal trends in temperature and conductivity. Additionally, we calculated mean,

minimum, and maximum temperature and conductivity over the spawning season (Dec 1 – May 31). Finally, we calculated the number of days during the spawning season with 1) Delta outflow over 20,000 cfs and 2) temperatures below 9^{o}C. Because there were years when no days met the above criteria, we added one to these variables to allow for log-transformation (15 of 38 years with temperatures recorded during the spawning season had zero days below 9^{o}C). All parameters were logged (base e) and centered prior to use in the model, unless noted below.

Biotic Environmental Data

We also investigated the impacts of age-0 and age-1+ density on growth (Figure 1). For density, we relied on the SFBS abundance indices using otter and midwater trawls (available from 1980–2018; CDFW 2021). We utilized indices for both age-0 and age-1+ for this analysis to account for differences in early-life and older life stages (Table 2). Unlike other model variables, the abundance indices were on an annual time step (not monthly). Missing age-specific indices from 1977–2020 (17 times for the otter and 23 times for the midwater trawl datasets) and those recorded as zero catch (2 times for the otter and 1 time for the midwater trawl datasets; see Table 2) were filled in using various techniques based on linear regression. Note the difference in date ranges from these surveys ensured that we had abundance indices to cover growth of LFS starting with the 1979 cohort, which required reconstructed abundances for age-2 fish in 1979 (hatched in 1977). All age-zero indices were filled in using predictions from linear regression between the SFBS indices and/or the FMWT index (averaged when possible; the FWMT index was from CDFW 2021). For example, to fill in missing age-0 indices for the SFBS midwater trawl, we utilized a linear regression between the FMWT and SFBS midwater trawl for 1977, 1978, and 2016 and used the average predictions from both the FMWT and SFBS otter trawl for 1994 and 2018. For the 1979 age-0 index, we filled in the missing values using methods outlined below for missing age-1 and age-2 indices. Finally, we used the average observed values from 2000–2018 for age-0 in 2019–2020 to account for differences in recruitment since the 1980s.

Linear regression techniques were also used to fill in missing age-1 and age-2 abundance indices. Briefly, missing values were interpolated using linear regression on log-abundance for a given cohort, based on methods used to estimate mortality in catch curves (e.g., estimating the slope [total mortality rate] for the 2000 cohort using age-0 fish caught in 2000, age-1 fish caught in 2001, and age-2 fish caught in 2002). Because each cohort only had three age-class indices (we excluded values from incomplete cohorts), we used the weighted average slope (estimated as -1.15) calculated from all cohorts using estimates from both otter and midwater trawl where each cohort was given the same weight in the global calculation. This method allowed us to minimize the uncertainty associated with small sample sizes (i.e., number of age-classes). Missing abundance indices were calculated based on the average predictions from forward and/or backward projection where a missing Age-1 index was the average of 1) preceding year log(Age-0 abundance index) minus the weighted average slope and 2) the following year log(Age-2 abundance index) plus the weighted average slope. When multiple abundance indices were missing for a given cohort, observations were filled in using the only available forward/backward projections. Finally, the sum of the Age-1 and Age-2 indices described above were used as the Age-1+ abundance indices for each year. Only abundance indices for Age-0 and Age-1+ abundance from the SFBS otter trawl survey were used in the growth model.

To account for prey availability, we utilized prey density estimated from the zooper R package created by Bashevkin (2020) and estimated the monthly geometric mean of *Eurytemora affinis* and mysid shrimp across all sampling sites within the SFE (NZ028, NZ048, NZ054, NZ060, NZ064, NZ086, NZ0325; Figure 1). Zooplankton sampling at these sites has been regularly conducted since 1972 (see Bashevkin et al. 2022 for details on the zooplankton surveys).

Overview of growth models

The model developed here differs in two important ways from traditional growth models used in fisheries by incorporating 1) multiple growth phases and 2) adding covariates to describe changes in the growth increment. The von Bertalanffy growth model (von Bertalanffy 1938) is the most common model to describe the lifetime growth patterns of fish because it has strong biological and empirical support (Beverton and Holt 1957; Chen et al. 1992; Lester et al. 2004). This model was developed to describe growth patterns of adult fish (e.g., von Bertalanffy 1968) but is often applied to describe lifetime growth patterns of fish. Further, this model is relatively inflexible as it assumes growth in length follows a constant pattern of decreasing incremental growth. Because the von Bertalanffy model is habitually applied to describe lifetime growth patterns, it has been criticized for its inability to represent complex changes occurring over the life of an organism (Day and Taylor 1997; Czarnołe'ski and Kozłowski 1998; Lester et al. 2004). Specifically, criticism has focused on the representation of growth associated with larval and early juvenile stages (Beverton and Holt 1957; Ricker 1975; Walters and Martell 2004) and changes in energy allocation associated with reproduction (Charnov 1993; Charnov et al. 2001; Lester et al. 2004), both of which likely violate the von Bertalanffy model assumptions.

To account for some of these critiques, we utilized the Lester et al. (2004) bi-phasic growth model to describe the underlying growth pattern (i.e., in the absence of inter-annual variation). The Lester et al. (2004) growth model was derived to account for changes in growth patterns associated with maturation. Growth during the first growth phase (i.e., pre-maturation growth) is assumed to be linear and transitions to the von Bertalanffy following maturation during the second growth phase. This model can also be adapted to describe additional growth phases, such as those associated with sex change in sequential hermaphrodites (Matthias et al. 2016) and incorporating early-life growth transitions can be accomplished by utilizing the appropriate data. One of the strengths of this model is the ability to jointly estimate the transition points between the growth phases by utilizing maturity and sex change data (see Matthias et al. 2016; Wilson et al. 2018). If data are available to inform additional transition points, such as early life transitions, it is also possible to incorporate those growth transitions into the model. In the absence of information to inform maturity or other transition points, the model will estimate a growth transition point solely based on length-at-age data and this transition could be independent from maturation (an important critique discussed below).

The second way this model differs from traditional growth models is that we incorporated covariates into the model to estimate the effects of environmental variation on somatic growth, similar to methods outlined by Matthias et al. (2018). Analytical methods commonly used to quantify the impacts of environmental variation on fish growth utilize ANOVA or multiple regression models on growth increments obtained from mark-recapture methods (e.g., Haugen et al. 2007; Vøllestad and Olsen 2008;

Davidson et al. 2010) or back-calculation (e.g., Maceina and Shireman 1982; Maceina et al. 1991). These methods ignore biological processes underlying fish growth often captured by von Bertalanffy and Lester et al. (2004) growth models (see Matthias et al. 2018). Here we apply log-normal regression techniques to describe variation in monthly growth increments resulting from environmental variation within the Lester et al. (2004) model framework (outlined in detail below). For example, individuals growing under ideal conditions will have a larger growth increment than those growing under poor conditions.

Collectively these two differences between our approach and the traditional application of growth models for fish populations allows us to quantify impacts of environmental variation on LFS growth. It should also be noted that we are using data collected via long-term monitoring programs (detailed above). Thus, the data we have available are correlative in nature and the trends we identify will need to be verified using targeted studies. However, this type of analysis has commonly been used to assess temporal variation in growth (e.g., Dorn 1992; Lorenzen 1996; Porch et al. 2002; Shelton et al. 2013; Matthias et al. 2018).

Growth in length

The model developed here was designed to assess variation in apparent growth of LFS using a combination of observed environmental conditions (e.g., water temperature, conductivity, outflow, density, and prey availability) and random effects. We did this by estimating parameters for the Lester et al. (2004) biphasic growth model (equation 1 and Table 3). Traditional formulation of the Lester et al. (2004) growth model was derived using assumptions related to maturation where phase-1 growth was pre-maturation and phase-2 was post-maturation. Histological data were not routinely collected as part of the monitoring surveys and preliminary model runs consistently indicate the estimated age-at-growth transition was not likely associated with maturation (i.e., we estimated a growth transition during the first year of life and Longfin Smelt primarily spawn during their second year of life; CDFG 2009). Therefore, we refer to growth during the initial phase as phase-1 and phase-2 as the second phase to aid clarity. It should also be noted that we do not explicitly account for changes in growth during early life (i.e., post-hatching, post-yolk sac, early juvenile, etc.) and close examination of the predicted length-at-age will be required to determine the ability of this model structure to predict early-life growth. While it is possible to incorporate additional transition points to describe early-life growth, this dramatically increases model complexity, increases the number of parameters to estimate (which may cause convergence issues), and increases the chance of memory allocation issues. Any added complexity to the model will require the use of supercomputers. Equations used in the model can be found in Table 3. The model was run in Template Model Builder (TMB; Kristensen et al. 2016) through Program R (R Core Team 2021).

The Lester et al. (2004) model is a biphasic growth model that separates growth into two growth phases. Phase-1 growth is assumed to be linear and phase-2 growth follows the von Bertalanffy model. An incremental formulation of the Lester et al. (2004) model is

$$L_A = \begin{cases} L_0 & if\ A = 0 \\ L_{A-1} + h & if\ A \leq T_{50} \\ L_{A-1}e^{-k} + L_\infty\left(1 - e^{-k}\right) & if\ A > T_{50} \end{cases} \quad (1)$$

where L_A is the predicted length-at-age A (whole age in months), L_0 is the length-at-age-0, h is the maximum annual growth rate and represents the growth rate in the absence of reproduction, k is the Brody growth coefficient describing how quickly growth approaches L_∞, L_∞ is the average asymptotic size, and T_{50} is the age-at-which 50% of the individuals transition between phase-1 and phase-2 growth. The Lester et al. (2004) model estimates reproductive investment g, which is used to calculate $k = \ln(1 + g/3)$ and $L_\infty = 3hg^{-1}$. If the transition point between the two growth phases (i.e., T_{50}) is linked with the age-at-50% maturity, they can be jointly estimated with a model describing maturation when data are available (see Matthias et al. 2016; 2019 and Wilson et al. 2018). As described above, the exact interpretation of these parameters (specifically h, g and T_{50}) may be confounded if the T_{50} is not associated with maturation and by additional growth transitions for which we are not accounting.

We modified the Lester et al. (2004) growth model described above to account for temporal changes in parameters using a mixed effects framework to describe temporal patterns in growth (Table 3). The model estimates the growth increment from age A to age $A + t$ (where A is whole age in months and $t = 1$ if calculating monthly growth increment or $0 \leq t < 1$ if an individual was captured at age $a = A + t$ months old). Three separate equations were used to describe phase-1, transitioning, and phase-2 growth (equations T1a–T1c respectively; Table 3). The transitioning growth phase (equation T1b; Table 3) was used to account for the possibility that individuals transitioned between growth phases at any time of the year (i.e., $A < T_{50,c} < A + 1$). Equations must be differentiable in TMB, therefore conditional statements cannot be used (e.g., if $A < T_{50}$ then use equation T1a, else use equation T1c). To overcome this constraint, we used logistic equations to represent the transition between growth phases (equations T1d–T1g; Table 3). Under certain conditions, such as with a temporally invariant growth model, equations described in Table 3 will give the same results as the traditional Lester et al. (2004) growth model (equation 1). Within the Lester et al. (2004) model framework, we used both random and fixed effects to assess how unobserved and observed conditions in the San Francisco Estuary influence LFS growth.

We compared two different random effects structures assessing whether growth patterns might have changed over time using cohort- and year-specific effects. These two random effects structures represent different hypotheses on how unobserved conditions influence growth patterns. Holding all fixed effects constant, the cohort-specific growth model assumes all individuals within a cohort have the same underlying growth curve (e.g., h, g, and/or T_{50}), but these growth patterns can change between cohorts. Cohort-specific random effects parameters were phase-1 growth rate h_c and the transition point between pre growth phases $T_{50,c}$, where $\ln(h_c)$ and $\ln(T_{50,c})$ were normally distributed with hyperparameters describing the mean and standard deviation (μ_x and σ_x; equation T1; Table 3). Additional cohort-specific parameters were asymptotic size $L_{\infty,c} = 3h_c g^{-1}$. For this model, the random effects describing year-specific growth effects were set to zero (i.e., $I_y = 0$). The year-specific growth model assumes growth patterns change over time and conditions in calendar year y have similar impacts on all individuals alive at that time (again while holding all fixed effects constant over time). We

modeled a year-specific effect on the growth increment obtained between each time step during a given year I_y (equation T1g; Table 3). The parameter $T_{50,c}$ was modeled as a cohort-specific effect to avoid additional complexity of accounting for changes in maturation patterns over multiple ages and years. This model assumes that L_∞ remains constant over time, but the rate which individuals approach L_∞ changes annually via variation in the growth increment (equation T1g; Table 3). The random effects I_y and $\ln(T_{50,c})$ were normally distributed with hyperparameters describing the mean and standard deviation (μ_x and σ_x; equation T1; Table 3). In the year-specific growth model, the phase-1 growth rate h_c was temporally constant (i.e., $\ln(h_c) = \mu_h$; equation T1; Table 3). Using this framework, if we set both the year- and cohort-specific random effects to the mean (i.e., $I_y = 0$, $\ln(h_c) = \mu_h$, and $\ln(T_{50,c}) = \mu_{T_{50}}$, plus $\gamma_y = \mu_\gamma$ the random effect for length-at-age-0 [described below]), then we have no random effects in the model. To aid convergence, we combined data from multiple cohorts (or years) when data were limited. For instance, with the cohort-specific effects we estimated a single random effect for the first four cohorts and another random effect covering the last three cohorts due to small sample sizes. Similarly, with the year-specific effects, we did not estimate a random year effect on growth for the first three years (i.e., $I_y = 0$ when $1979 \leq y \leq 1981$).

A suite of fixed effects variables was used to describe environmental conditions that may influence growth between age A and $A + 1$ (equations T1–T3; Table 3). We compared the effects of environmental conditions assuming they have equal effects on both phase-1 and phase-2 growth increments (i.e., setting the regression coefficients for phase-1 growth $\beta_x^* = 0$) and unequal effects (i.e., estimating regression coefficients for phase-1 β_x^* and phase-2 β_x growth components; equations T2–T3; Table 3). If $\beta_x^* \neq 0$, this would suggest an ontogenetic shift associated with the growth transition that changes how environmental conditions influence growth patterns. Environmental effects tested were water temperature (not logged; applied as a parabolic relationship where t_{crit}^* and t_{crit} represent the vertex of the parabola for phase-1 and phase-2 growth, respectively; equations T2–T3; Table 3), log(Delta outflow), log(conductivity), and a temperature – log(Delta outflow) (or log(conductivity)) interaction. Delta outflow and conductivity are highly correlated (>80% in log-space) and therefore measuring the similar environmental conditions within the San Francisco Estuary. Thus, they were not used in the same model but were statistically compared using a model comparison framework. To estimate the effects of density dependence and prey availability, we included several interactions between abundance, time, and prey densities. Here we are representing time as numerical month since hatching (assumed to be Jan 1 of a given year) to account for declines in abundance over a given year or, depending on the direction of the interaction, increases in biomass (especially with age-0 fish). Given model complexity and difficulties with convergence, we were only able to test interactions between time * age-0 abundance and time * age-1+ abundance. Attempts to incorporate interactions between abundance and prey densities resulted in convergence and memory allocation issues due to the number of parameters being estimated.

In addition to using mixed effects to describe variation in monthly growth rates, we also used a mixed-effects framework to describe variation in length-at-age-0 $L_{0,y}$ for each year y (i.e., length-at-hatching; equation T4; Table 3). Annual differences in $L_{0,y}$ can represent variation in hatch size and/or variation in hatch date (likely a combination of both, but we are unable to distinguish using these data). If we

assume the distribution of size-at-hatching is relatively consistent across years, then variation in $L_{0,y}$ should primarily be driven by variation in hatch dates. Variation in hatch dates arise from annual differences the timing of spawning and/or lengths of the incubation periods. Therefore, larger values of $L_{0,y}$ should represent earlier mean hatch dates and smaller values (or more negative values) should represent later mean hatch dates because we assumed a hatch date Jan 1 each year. Alternatively, if the hatch dates are constant over time, which is very unlikely, then variation in $L_{0,y}$ would be driven by variation in hatching size. We used a random-intercept regression to describe both random annual variation in $L_{0,y}$ (γ_y, where $\gamma_y \sim N(\mu_\gamma, \sigma_\gamma)$) and assumed linear interactions with environmental covariates (where α_x represent regression coefficients; equation T4; Table 3). Covariates used to estimate $L_{0,y}$ were related to spawning season (Dec 1 – May 31) and included the number of days with Delta outflow over 20,000 cfs (plus one), the number of days under 9ºC (plus one) and mean temperature and conductivity. All environmental covariates predicting the $L_{0,y}$ were logged and centered. Note that we experienced model convergence issues with the random-intercept and therefore only used a single intercept for the model (i.e., $\gamma_y = \gamma_{y=1}$).

The probability of observing a given individual is a function of both the underlying growth pattern, selectivity of the gear used to capture that individual, and spatial overlap of the survey relative to the distribution of the population. If gear selectivity is constant over all sizes, we would expect to observe individuals in proportion to their numeric availability (e.g., mostly small, young fish and few big, old fish). However, due to selectivity we observe fish in proportion to their numeric availability and their availability to the sampling gear. We used a truncated normal distribution to represent minimum size selectivity in the likelihood (equation T5; Table 3) where $N(x)$ is the normal probability density function with mean L_a and standard deviation σ and $\Phi(x)$ is the cumulative distribution function with a left truncation at a specified length $S_{g_i}$for individual i caught with gear g_i. We set g_i as the minimum observed length across all surveys. This is especially important for surveys such as the SFBS, as they only measured individuals above 40 mm in length. For simplicity, we assumed no maximum size selectivity. While we were able to account for some size-selectivity in the model, we were unable to account for spatial selectivity. Due to the migration patterns of LFS, the sampling programs do not cover the entire spatial distribution of the population when LFS move to the Pacific Ocean during summer months. If migration is driven in part by size, then there may be additional size-selectivity that could introduce bias into the model results. To assess this bias and improve the predictive capacity of the model, additional sampling across the entire distribution is needed. Additionally, methods combining otolith increment analysis and back-calculation could also provide valuable insight into potential bias.

Analysis

Following recommendations by Zuur et al. (2009), we planned on using a combined approach of restricted maximum likelihood (REML) estimation to select between different random effects structures (i.e., cohort versus year-specific growth model) and maximum likelihood (MLE) estimation to compare different fixed effects (i.e., assumptions based on how environmental variables effect Phase-1 and Phase-2 growth patterns). However, we ran into convergence issues due to the number of fixed effects parameters being estimated. Therefore, we followed a slightly modified process where (step 1) we

initially fit the different random effects structures (cohort- and year-specific growth models) using REML and compared the models with Akaike Information Criterion (AIC; equations T6 and T7; Table 3; where NLL is the negative log likelihood and p is the number of parameters being estimated). Step 2 assessed if the effects of covariates differed between the growth phases (i.e., if $\beta_x^* = 0$ or $\beta_x^* \neq 0$) with AIC and verifying appropriate model structure using 95% confidence intervals of β_x^* to assess overlap with zero (i.e., assess how many parameters are significantly different from zero). Following this comparison (step 3), we used AIC to assess different representations for environmental conditions (Delta outflow versus conductivity) on both the growth increment and length-at-age-0 L_o. For the final step we compared different fixed effects structures (i.e., a comparison of interactive effects). We ran a suite of models (Table 4) to compare interaction effects, but restricted the number of interactions due to computer memory limitations.

When using the REML framework for model comparison, we are effectively treating all parameters (except those associated with estimating variance) as a random effect. This is beneficial as it allows us to generate unbiased estimates of variance (see Zuur et al. 2009 for more details on this). However, in our case this provides a challenge in comparing different numbers of fixed effects because the number of parameters p used in the AIC calculations does not change (i.e., only parameters associated with variance are counted in the AIC calculation). Therefore, the model comparison framework we used here does not effectively account for the number of parameters estimated within the model. As a result, choosing the most parsimonious model can be challenging if multiple models have close AIC values (i.e., $\Delta\mathrm{AIC} \leq 10$). This does not appear to be an issue in our case, as $\Delta\mathrm{AIC}$ values were consistently large (>250) when comparing models.

Results

Initial models run using random effects associated with length-at-age-0 random intercept γ_y and the timing of transition from between the growth phases $T_{50,c}$ were unreliable and had inconsistent model fits. Therefore, we did not pursue models with those random effects further (i.e., we set $\gamma_y = \mu_\gamma$ and $\ln(T_{50,c}) = \mu_{T_{50}}$). It is possible that having consistent larval sampling during the entire hatching period and histology data could help inform these parameters.

Step 1 of our model comparison suggests the year-specific random effects structure (M-2) is better at predicting the observed length-at-age patterns than the cohort-specific growth model (M-1; Table 4). All further model comparisons were made using the year-specific random effect (M-2). For step 2 of our analysis, we compared the assumption that the response to environmental variables was different (M-2) between phase-1 and phase-2 growth versus the same (M-3; Table 4) using both AIC and overlap of Phase-1 regression coefficients β_x^* with zero (using the 95% confidence intervals; not shown). Using both measures the best model was M-2, suggesting the magnitude and (often) direction of the growth response to environmental conditions changes over the lifetime of LFS (Table 4; 5). The third step in our selection procedure compared the effects of Delta outflow (M-2) versus conductivity (M-5) on the growth increment and length-at-age-0. Results suggest the effects of Delta outflow was a better predictor (Table 4). The final step in the model comparison included interactions on both the growth increment and length-at-age-0 (M-6) compared to the best model without interactions (M-2) and

including interactions improved model fits (Table 4). Further assessment of model predictions below focuses on the best model with interactions (M-6).

Holding all effects at their mean and ignoring the random effects (i.e., holding them constant at zero), we predict a mean monthly growth rate during phase-1 (h) of about 8.6 mm (Table 5). This corresponds to a mean asymptotic size of 137 mm ($L_\infty = 3h/g$ where g corresponds to reproductive investment under the Lester et al. (2004) interpretation of the model). The transition between phase-1 and phase-2 growth was predicted at 6.2 months (Table 5). Finally, the length-at-age-0 (holding all effects at their mean) was -0. 3 mm (Table 5).

Prey densities, temperature, Delta outflow, the temperature-outflow interaction were all significant and the effects of these covariates differed between the growth phases (i.e., coefficients did not overlap with zero; Figure 2; Table 5). We predicted a positive relationship between phase-1 growth and the *E. affinis* abundance and a negative relationship between phase-2 and *E. affinis* abundance. We also predicted negative relationships between growth and Mysis abundance for both growth phases. In general, growth increased with temperature, but this relationship had wide confidence bounds for phase-2 growth (although the 95% confidence intervals for β_1 did not overlap zero; Table 5). The critical temperature for phase-1 growth t^*_{crit} was near the upper limit of observed temperatures (23°C; Table 5). For phase-2 growth t_{crit} was extremely negative (-47°C; Table 5), suggesting a linear relationship might have been better than a parabolic relationship. The effects of Delta outflow were opposite for phase-1 (negative) and phase-2 (positive) growth. The interactions between temperature and Delta outflow were also opposite for phase-1 (negative) and phase-2 (positive). For a given temperature during phase-1 growth, we predict growth increasing with outflow until high temperatures (20-22°C), where highest growth was predicted at intermediate temperatures (Figure 2). For phase-2 growth, we predict decreasing growth with increasing Delta outflow across temperatures below 20°C and highest growth at low to intermediate levels of Delta outflow at temperatures above 20°C (Figure 2).

The density and time-related effects (plus interactions) were also significant and the effects differed between the growth phases (Figure 3; Table 5). Phase-1 predicted growth rates were positively related to abundance (both age-0 and age-1+ indices) and negatively related to time. Phase-2 predicted growth rates were negatively related to abundance (both age-0 and age-1+ indices), along with time. For detecting density-dependent effects on growth, look for negative relationships between phase-1 growth and age-0 abundance (positive effect) along with phase-2 growth and age-1+ abundance (negative effect). The effects of age-1+ abundance on phase-1 growth and/or age-0 abundance on phase-2 growth may provide insight into inter-specific effects (e.g., competition if negative relationships) or underlying conditions that may be affecting abundance and growth simultaneously (e.g., positive relationships via unaccounted for effects or interactions). For the interactive effects of abundance and time, it is useful to think about the impacts resulting from declines in abundance over time, given the temporal scale (i.e., a month effect). In general, for phase-1 growth, we predict declines in growth as time since hatching increases (i.e., declining growth over time), but the relative declines in growth are larger at lower abundance indices (Figure 3, but note the transition between the growth phases is around 6.2 months). For phase-2 growth, we see the opposite, where growth generally declines from Jan-Dec, but the relative declines in growth are larger at higher abundance indices (Figure 3).

Environmental covariates for predicting length-at-age-0 $L_{0,y}$ were all significant (i.e., 95% confidence intervals did not overlap zero; Figure 3; Table 5). In general, larger (more positive) values for $L_{0,y}$ suggest earlier mean hatch dates and smaller (more negative) values suggest later hatch dates if size-at-hatching is relatively consistent over time. The number of days during the spawning season (Dec 1 – May 31) with Delta outflow over 20,000 cfs was negatively related to $L_{0,y}$, suggesting that large rain events and/or large reservoir releases over the spawning season led to smaller values of $L_{0,y}$ or later hatch dates (Figure 4; Table 5). We predicted a negative effect of mean water temperature (from Dec 1 – May 31) on length-at-age-0 (Figure 4; Table 5). Assuming $L_{0,y}$ is related to mean spawning time, this suggests that a colder spawning season leads to earlier spawning and/or larger $L_{0,y}$ on Jan-1 than warmer spawning seasons. It is important to note that some of these variables are related and influenced by water management.

Overall, we did not see any major trends in the random year effect I_y on growth or predicted length-at-age-0, 1, or 2 (where age is in years; Figure 5). With I_y, we predicted relatively high growth through the mid-1980s, but this is likely an artifact of sampling design and high reliance on fall surveys (FMWT) and SFBS not measuring individuals <40mm until juvenile sampling increased with the 20mm program in 1995. Predicted length-at-age-0 showed high interannual variation across time and this variation persisted as the cohorts progressed over time. It is important to note that the range in mean length-at-age-0 was only around a 7 mm difference between the lowest and largest predicted values. This variation increased with predicted length-at-age-1, which a range of about 18 mm between age-1 fish in 1982 (58 mm) and 2001 (76 mm). Finally, the range in predicted length-at-age-2 was similar, about 18mm, with the smallest in 1982 (95mm) and largest in 1990 (114mm).

Looking at the general model fits, it appears that the model does a good job predicting the mean length-at-age patterns (Figure 6). There are certain cohorts where the model underestimates the length-at-age of older fish (>10 months of age), but in these instances we had large numbers of small fish caught in the 20mm, SLS, and EDSM 20mm surveys (Figure 6). This pattern could arise from low sample sizes of larger/older fish, size-based mortality patterns (we only observe fish that survived to be caught and have no information on fish that did not survive to be sampled), differences in the distribution of sampling relative to the distribution of the population (e.g., we are not sampling across the entire distribution of the population), and/or many other factors. It is also important to note the selectivity of the SFBS, where only fish >40 mm were measured (easily seen in the 1985 cohort, but present in all other cohorts) and with the SLS study where we see fish recruiting to the gear after hatching (e.g., 2006 cohort; Figure 6). Besides these instances of underestimation at older length-at-age and selectivity, there are no obvious patterns in the residuals (Figure 6).

Discussion

Utilizing a modified biphasic growth model, we identified a suite of biotic and abiotic drivers in LFS apparent growth patterns. Based on model results, we predicted similar magnitudes of abiotic (temperature, Delta outflow), biotic (prey availability), and density-related effects on the monthly growth increments (Figures 2 and 3). This suggests that LFS growth is not driven by a single environmental factor, but rather a mixture of environmental conditions. Although we found high

interannual variability in the predicted length-at-age of LFS, the growth patterns have remained remarkably stable between 1979-2020 considering the declines in SFE productivity and establishment of the invasive clam *P. amurensis* in the late 1980s (Kimmerer and Orsi 1996; Kimmerer 2002), declines in pelagic population sizes in the 2000s (i.e., POD; Sommer et al. 2007), changes to their prey base (Figure 1).

There is very little published information about the growth rates of LFS, but the general patterns in growth predicted here reflect those earlier studies on LFS growth. Using daily otolith increments, LFS display slow growth during early life (larval stage), which increases to about 6 months, then slows again (Hobbs, unpublished). This is depicted with our estimates, with 0.13 mm/day (range 0.07-0.20 mm/day) during the first month compared to 0.23 mm/day (range 0.09-0.35 mm/day) around 6 months and 0.14 mm/day (range 0.08-0.22 mm/day) around 11 months of age. These estimates also fall within the range observed by Souza et al. (2006), who predicted a mean apparent growth rate of 0.12-0.23 mm/day between 1995-2004 using data from the 20mm Survey (data also used in this study). LFS in Lake Washington from the 1960s averaged from 52-77 mm FL at 10-months (Moulton 1974), which is slightly larger than the predicted mean length-at-age-10 months from our model (mean 61 mm with a range of 51-66 mm).

A positive relationship between outflow and LFS abundance has been documented by several studies (Stevens and Miller 1983; Jassby et al. 1995; Kimmerer 2002), but hypotheses remain to be tested about the biological mechanism behind the relationship. For example, higher outflow may create a larger area of low and intermediate salinity habitat where LFS may have a competitive advantage over other estuarine species (Hieb and Baxter 1993, Baxter 1999). This hypothesis would be consistent with a positive outflow-growth relationship and may be a cause for the positive relationship between phase-1 growth and outflow (Figure 2). Another hypothesis is that the relationship between food availability and outflow may drive the apparent relationship between LFS abundance and outflow. For example, the abundance of one preferred prey item, *E. affinis*, has been correlated with the location of X2 in the spring (Kimmerer 2002), which is again consistent with observations for phase-1 growth. In contrast, there was a negative relationship between growth and the water year index using data reported by Souza et al. (2006), which is similar to the predictions between outflow and phase-2 growth (i.e., negative; Figure 2). This suggests that any relationship between outflow and growth may be age (or stage) dependent and/or influenced by other environmental conditions that the fish experience.

The SFE population of LFS occupy habitat near the upper thermal limit of its range. During the initial growth phase (up to 6.2 months) we predicted a positive relationship between temperature and growth, suggesting that LFS growth increases with temperatures, consistent with observations of larval growth from lab studies (Yanagitsuru et al. 2021; here we are classifying larval LFS as being <20mm). However, after transitioning to the secondary growth phase, we predicted a highly variable relationship between growth and temperature. This trend is likely influenced by the distribution of LFS in the SFE because summer water temperatures in many areas of the SFE regularly exceed thermal maxima for LFS and has been found to restrict the distribution of age-1 LFS (Baxter 1999).

The predicted relationships between prey availability and growth indicated that prey densities may be important drivers of early growth during phase-1 (up to ~6 months), but *E. affinis* and Mysis may not be limiting during phase-2 growth. We identified a positive relationship between *E. affinis* densities and

phase-1 growth, suggesting this prey source may be a bottom-up driver of early growth in LFS. This is supported by dietary studies on LFS that show high reliance on *E. affinis* (~90%) until about 25 mm in length, after which they switch to selecting for Mysis (Barros et al. 2022), which is about 4-5 months after hatching. The effects of Mysis on phase-1 growth were opposite of what we would expect, however caution should be used to infer the importance of Mysis during phase-1 growth using these results due to the growth transition around 6-months of age. For phase-2 growth the relationships between both *E. affinis* and Mysis were also opposite of what we would expect. This suggests that bottom-up factors play minor role in influencing growth during phase-2 or, more likely, we did not accurately capture prey availability for phase-2 when large portions of time are spent in ocean.

Density-dependent growth is a key process regulating population growth (Lorenzen and Enberg 2002) and is one of the best-established forms of density-dependence in fishes (Beverton and Holt 1957). With density-dependent growth, we expect a negative relationship between abundance and growth for a given age/phase grouping (i.e., declining phase-1 growth with age-0 abundance or declining phase-2 growth with age-1+ abundance), as predicted for the effects of age-1+ abundance index on phase-2 growth (Figure 3). For phase-1 growth we predicted the opposite, increasing growth with age-0 abundance, suggesting density dependent growth might not be a driving factor in LFS early-life growth. Additional density-dependence may arise via the abundance-time interactions, in which we would expect growth to increase as time-since hatching (Jan-1) increases. This assumes that there is a time-abundance relationship (i.e., mortality) that is relatively consistent over a given year (i.e., abundance in a given year is highest on Jan-1 and declines until Dec-31). If this were the case, we would see a positive effect of time on growth. However, density-dependent growth is also dependent on prey availability, which likely is interacting with abundance. We were unable to account for these interactions due to model complexity and size.

Using otolith increments Hobbs (unpublished), identified a growth transition around 6–7 months, in which the growth rate slowed after this transition. This timing corresponds with our estimated transition between phase-1 and phase-2 growth ($T_{50,c}$) where we are predicting a switch from linear to von Bertalanffy growth around 6.2 months of age. We should note the traditional interpretation of the model assumes this growth transition is associated with maturation and investment into reproduction (Lester et al. 2004). However, previous studies suggest that LFS tend to mature around age-22 months (Dryfoos 1965, Moulton 1974, Hieb and Baxter 1993). Therefore, it is possible that additional data on maturation could help inform the growth transition associated with maturation. The growth transition predicted in our model may be describing an ontogenetic shift occurring during the juvenile growth phase. Juvenile LFS migration from the SFE to the Pacific Ocean is expected to occur in spring or early summer (Rosenfield and Baxter 2007; Tobias and Baxter 2022*, in press*), which should be around 6 months of age. The predicted growth transition may be related changes in growth with seaward migration, could be a sampling artifact due to fish moving out of the sampling area, or could be a combination of both. Further assessment of this change in growth around 6 months of age is necessary.

The LFS dataset used in this model may introduce bias due to migration/spatial selectivity and incomplete sampling of the population during parts of the year (i.e., summer). Juvenile LFS migrate to the Pacific Ocean once water temperatures in the SFE warm from spring into early summer (roughly around 6 months of age assuming a January 1 hatch date) and return to the SFE as temperatures

decrease in the fall and early winter (Rosenfield and Baxter 2007; Jefferies et al. 2016; Tobias and Baxter 2022, *in press*). Some fish that migrate to the Pacific Ocean may remain in marine waters for extended periods (e.g., over the winter), while others return over the fall and winter (reviewed in Chen et al. 2022). Further, LFS are facultatively anadromous within the SFE and some individuals may remain in the SFE over the entire year (Moyle 2002). Given this complex life history, we expect that some of the relationships between growth and environmental factors identified here may be biased using the historical data and future efforts should explore these the magnitude and direction of potential biases. Nevertheless, our predictions of relatively stable LFS growth over time (i.e., predicted lengths-at-age 0, 1, and 2 years) should be robust to bias associated with migration because sampling occurred over the entire year throughout the entire time series and the bias associated with migration should influence all cohorts equally (i.e., we expect the bias to remain stable over all cohorts analyzed). Additionally, there should be minimal spatial sampling bias for comparing predicted length-at-age 0, 1, and 2 (i.e., over winter if we assume a January 1 hatch date) because the timing of the LFS migration patterns in the SFE follow predictable seasonal patterns (Tobias and Baxter 2022, *in press*) and there should be high spatial overlap of the population and sampling distributions over winter months. Ideally, we would be able to sample individuals from across the entire spatial extent of the population and obtaining length samples outside of the SFE would allow us to assess bias in the growth model developed here. Until an expanded sampling program is implemented, it may also be feasible to incorporate and/or compare growth trajectories informed via otolith increment analyses to assess for bias in this growth model associated with migration out of the sampling area.

It has also been suggested that using otolith increment analyses would be a better option to quantify the impacts of environmental conditions on growth and, in some ways, this is a valid argument. Otoliths should provide a better measure of individual growth assuming daily/annual growth increments accurately reflect growth in length. However, there are several major benefits for using the available LFS data. Extracting otoliths is lethal and the methods described here provide a way to measure apparent growth rates using non-lethal means. Further, otoliths from LFS were infrequently collected in the past and using the available data as we have done here may be one of the only ways to quantify changes in growth from the 1980s-present. We should note that ongoing and future efforts using otolith increment analyses are still valuable for quantifying changes in growth patterns. For instance (as mentioned above), otolith increment analyses may still provide an opportunity to assess bias in the growth models developed here. Therefore, future efforts to assess variation in LFS growth should focus on utilizing a hybrid approach that integrates multiple data types to provide robust and unbiased estimates.

Many long-term monitoring programs have been designed to collect information on abundance trends. These programs often collect demographics data (e.g., length, weight, age, maturity, sex, etc.) along with abundance trends that can also be used to monitor changes in population productivity and/or life-history characteristics. We used data from multiple long-term monitoring programs to assess for systematic changes in apparent growth of a population experiencing abundance declines. Results show that apparent growth patterns and predicted length-at-age, while variable, have remained stable from 1979–2020. Given the declining LFS population trajectories in the SFE (see Chen et al. 2022), we recommend continued monitoring of LFS growth patterns as an indicator for changes in population productivity. Additional analytical efforts should be focused on quantifying changes in other life-history characteristics that influence population productivity, such as the maturation schedule (i.e., proportion maturing at a given age), frequency of repeat spawning, and/or mortality. Additionally, changes to the

prey base and SFE productivity may drive changes in egg quality and ultimately larval survival (e.g., thiamine deficiency in Lake Trout (*Salvelinus namaycush)* and Chinook salmon; Fitzsimons et al. 2009; NOAA 2020). By identifying the drivers of the SFE LFS population decline, we can develop targeted management actions to facilitate population recovery.

R code for the analyses and graphs can be found at https://github.com/USFWS/Longfin-Smelt-Growth-Model

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Tables

Table 1: Sample sizes for each survey used in the Longfin Smelt growth model.

Year	20 mm	EDSM 20 mm	EDSM KT	FMWT	SFBS	SLS	Total
1981	-	-	-	665	3,933	-	4,598
1982	-	-	-	763	31,007	-	31,770
1983	-	-	-	92	14,413	-	15,105
1984	-	-	-	463	4,414	-	4,877
1985	-	-	-	294	2,746	-	3,040
1986	-	-	-	237	2,392	-	2,629
1987	-	-	-	552	1,820	-	2,372
1988	-	-	-	392	1,240	-	1,632
1989	-	-	-	239	429	-	668
1990	-	-	-	152	161	-	313
1991	-	-	-	214	158	-	372
1992	-	-	-	82	95	-	177
1993	-	-	-	636	552	-	1,188
1994	-	-	-	917	328	-	1,245
1995	4,054	-	-	1,940	12,426	-	18,420
1996	17,796	-	-	1,674	1,650	-	21,120
1997	13,533	-	-	603	1,322	-	15,458
1998	7,617	-	-	2,262	4,699	-	14,578
1999	11,160	-	-	1,375	6,655	-	19,190
2000	18,174	-	-	1,449	2,995	-	22,618
2001	10,251	-	-	916	1,140	-	12,307
2002	14,195	-	-	449	1,012	-	15,656
2003	6,180	-	-	316	717	-	7,213
2004	5,494	-	-	129	871	-	6,494
2005	4,197	-	-	65	638	2,104	7,004
2006	3,492	-	-	436	1,191	3,606	8,725
2007	2,603	-	-	10	476	3	3,092
2008	9,022	-	-	87	334	-	9,443
2009	13,577	-	-	51	419	-	14,047
2010	6,698	-	-	83	166	-	6,947
2011	4,923	-	-	292	1,020	-	6,235
2012	3,309	-	-	36	589	-	3,934
2013	12,648	-	-	83	528	-	13,259
2014	1,851	-	-	11	139	-	2,001
2015	451	-	-	3	101	-	555
2016	1,081	-	-	6	-	-	1,087
2017	3,199	283	66	69	-	-	3,617
2018	2,030	61	237	-	-	-	2,328
2019	4,568	34	288	-	-	-	4,890
2020	-	919	43	-	-	-	962

Year	20 mm	EDSM 20 mm	EDSM KT	FMWT	SFBS	SLS	Total
Total	182,103	1,297	634	18,643	102,776	5,713	311,166

Table 2: Indices used to calculate abundance trends in the growth model. Indices were based on Fall Midwater Trawl Catch (FMWT), San Francisco Bay Study (SFBS) midwater trawl (MWT), and SFBS otter trawl (OT). Age-specific indices were available for both SFBS surveys. Values with superscripts were estimated using linear regression techniques (see methods) and values of a indicate index was unavailable and b indicate the value was recorded as zero. The SFBS OT indices from 1979–2020 were used in the growth model.

Year	FMWT	SFBS MWT Age-0	SFBS MWT Age-1	SFBS MWT Age-2	SFBS OT Age-0	SFBS OT Age-1	SFBS OT Age-2
1977	210	1,238[a]	-	-	3,245[a]	-	-
1978	6,619	21,026[a]	393[a]	-	25,126[a]	1,029[a]	-
1979	-	5,289[a]	7,354[a]	125[a]	737[a]	1,631[a]	326[a]
1980	31,184	190,790	1,386	2,572	128,321	142	106
1981	2,202	1,959	51,372	644	4,139	11,837	122
1982	62,905	299,069	9,785	2,307	257,965	3,070	1,005
1983	11,864	33,651	296,253	2,865	23,860	142,861	917
1984	7,408	29,218	25,463	4,082	44,329	29,399	5,767
1985	992	2,895	58,525	2,693	11,787	12,626	1,185
1986	6,160	24,908	12,524	2,479	12,070	2,784	288
1987	1,520	2,872	33,471	2,287	1,984	7,174	1,284
1988	791	1,724	18,360	4,921	1,094	4,323	2,220
1989	456	1,137	6,595	1,514	971	2,179	369
1990	243	745	2,776	1,058	681	386	317
1991	134	131	3,852	541	245	474	161
1992	76	370	1,134	86	620	448	218
1993	798	5,086	811	22	7,006	53	59[b]
1994	545	1,930[a]	16,515	349	2,847	4,207	479
1995	8,205	555,398	612[a]	1,637[a]	152,973	791	504
1996	1,346	666	31,522[a]	194[a]	11,045	4,153	248
1997	690	4,585	6,863	1,789	10,692	3,706	1,075
1998	6,654	62,853	6,240	1,120	20,605	197	89
1999	5,243	59,040	17,546	895	57,980	6,827	600
2000	3,437	12,326	12,132	1,168	16,079	1,841	240
2001	247	2,107	10,707	1,154	812	15,507	954
2002	707	1,173	2,472	1,900	18,132	1,070	1,604
2003	467	230	2,558	186	4,007	6,339	1,292
2004	191	1,307	2,030	486	3,529	4,323	886
2005	129	617	3,487	545	8,459	4,074	593
2006	1,949	2,780	1,342	91	21,517	1,379	285
2007	13	441	1,351	79	3,636	5,156	815

Year	FMWT	SFBS MWT Age-0	SFBS MWT Age-1	SFBS MWT Age-2	SFBS OT Age-0	SFBS OT Age-1	SFBS OT Age-2
2008	139	1,207	218	230	6,155	343	162
2009	65	323	3,485	75	971	10,317	469
2010	191	867	599	114	628	1,373	280
2011	477	1,404	950	106	14,261	7,134	503
2012	61	398	5,353	1,106	2,170	840	360
2013	164	1,445	647	324	15,545	574	277
2014	16	1,194	615	91[b]	1,228	519	63
2015	4	231	403	44	536	693	196
2016	7	508[a]	106[a]	124[a]	432[a]	127[a]	164[a]
2017	141	2106	161	49	5,844	137[b]	30
2018	52	804[a]	668[a]	51[a]	3,387	1,854[a]	43[a]
2019	-	1,006[a]	255[a]	212[a]	3,939[a]	1,074[a]	588[a]
2020	-	1,006[a]	319[a]	81[a]	3,939[a]	1,249[a]	341[a]

Table 3: Equations used in the longfin smelt growth model (described in methods text).

Description		Equation	Components
(1a)	Growth increment – juvenile	$l_1 = h_c t e^{B^*_{y,m}}$	$L_{\infty,c} = 3h_c g^{-1}$
(1b)	Growth increment – maturing	$l_2 = h_c t e^{B^*_{y,m}}(1-A^*) + \left(L_{\infty,c} - \left(L_{A,y} + h_c t e^{B^*_{y,m}}(1-A^*)\right)\right) * \left(1-e^{-k(A^*)}\right) e^{B_{y,m}}$	$k = \ln(1+g/3)$ $A^* = (A+t) - T_{50,c}$ $\ln(T_{50,c}) \sim N(\mu_{T_{50}}, \sigma_{T_{50}})$
(1c)	Growth increment – adult	$l_3 = \left(L_{\infty,c} - L_{A,y}\right)\left(1-e^{-kt}\right) e^{B_{y,m}}$	Cohort-specific model $\ln(h_c) \sim N(\mu_h, \sigma_h)$
(1d)	Prob. of growing following juvenile growth phase	$P_1 = (1-P_2)$	$I_y = 0$
(1e)	Prob. of maturing between ages $A-1$ and A	$P_2 = \left(1+e^{-(A-T_{50,c})/0.1}\right)^{-1} - P_3$	Year-specific model $\ln(h_c) = \mu_h$
(1f)	Prob. of growing following adult growth phase	$P_3 = \left(1+e^{-\left((A+t)-T_{50,c}\right)/0.1}\right)^{-1}$	$I_y \sim N(0, \sigma_I)$
(1g)	Pred. length-at-age a for year y	$L_{a,y} = L_{A+t,y} = L_{A,y} + \sum_{j=1}^{3} l_j P_j e^{I_y}$	
(2)	Month-specific effect on adult growth	$B_{y,m} = \beta_1 (x - t_{crt})^2_{1,y,m} + \sum_{j=2}^{n_\beta} \beta_j x_{j,y,m}$	

(3)	Month-specific effect on juvenile growth	$\mathrm{B}^{*}_{y,m} = (\beta_1 + \beta^{*}_1)(x - t^{*}_{crt})^2_{1,y,m} + \sum_{j=2}^{n_\beta}(\beta_j + \beta^{*}_j)x_{j,y,m}$	
(4)	Length-at-age-0 for year y	$L_{0,y} = \gamma_y + \sum_{j=1}^{n_\alpha}\alpha_j \mathrm{x}_{j,y}$	$\gamma_y \sim N(\mu_\gamma, \sigma_\gamma)$
(5)	Likelihood for each observation i	$L_i = N(L_i \mid L_a, \sigma) * \left(1 - \Phi\left((S_{g_i} - L_a)\sigma^{-1}\right)\right)^{-1}$	
(6)	Total negative log likelihood	$\mathrm{NLL} = -\sum_{i=1}^{n} \ln(L_i)$	
(7)	AIC	$AIC = 2 * \mathrm{NLL} + 2 * p$	

Table 4: Results for model comparison. The x's indicate parameters used in the model estimated separately for phase-1 and phase-2 growth (i.e., assumed the effects on the growth increment may not equal between the growth phases). For model M-3 the **x**'s (bolded and italicized) shared parameters between phase-1 and phase-2 growth (i.e., assumed the effects on the growth increment were equal between the growth phases).

	Parameter	M-1	M-2	M-3	M-4	M-5	M-6
Random Effect Structure							
	Cohort-specific	x					
	Year-specific		x	***x***	x	x	x
Effects on Growth Increment							
1	Delta Outflow	x	x	***x***		x	x
2	Conductivity				x		
3	Temperature	x	x	***x***	x	x	x
4	Delta Outflow * Temperature						x
5	Age-0 abundance	x	x	***x***	x	x	x
6	Age-1+ abundance	x	x	***x***	x	x	x
7	Time						x
8	Age-0 abundance * Time						x
9	Age-1+ abundance * Time						x
10	*E. affinis* density	x	x	***x***	x	x	x
11	Mysid density	x	x	***x***	x	x	x
Effects on Length-at-age-0 (values calculated over spawning season)							
12	Days over 20,000 cfs	x	x	***x***		x	x
13	Mean Conductivity				x		
14	Mean Temperature	x	x	***x***	x		x
15	Days under 9C					x	
16	Days over 20,000 cfs * Mean Temp.						x
Total Number of Regression Parameters		12	12	6	12	12	20
Delta AIC Value		10,411	3,889	14,847	4,144	5,017	0
Negative Log Likelihood		985,186	981,925	987,404	982,053	982,489	979,981

Table 5: Fixed effect parameter estimates for the longfin growth model with 95% confidence intervals.

Par.	Description	Eq.	Estimate (95% CIs)
σ_I	Standard deviation for year-specific random effect on the growth increment (mean = 0)	T1	0.327 (0.263, 0.408)
σ	Standard deviation around the observed mean length-at-age	T5	6.49 (6.47, 6.51)
h_c	Mean monthly growth increment during phase-1 growth (in mm; constant across all cohorts c)	T1	8.57 (8.44, 8.69)
g	Reproductive investment	T1	0.187 (0.184, 0.190)
γ_y	Intercept for length-at-age-0 ($L_{0,y}$; constant across all years y)	T4	-0.285 (-0.431, -0.138)
$T_{50,c}$	Timing (age in months) of 50% transition between growth phases (constant across all cohorts c)	T1	6.19 (6.18, 6.21)
β_1	Coefficient for temperature on the Phase-2 growth increment	T2, T3	9.0E-5 (4.5E-5, 1.0E-4)
β_2	Coefficient for ln(outflow) on the Phase-2 growth increment	T2, T3	-0.089 (-0.101, -0.077)
β_3	Coefficient for temperature * ln(outflow) interaction on the Phase-2 growth increment	T2, T3	0.009 (0.007, 0.011)
β_4	Coefficient for Age-0 abundance index on the Phase-2 growth increment	T2, T3	-0.067 (-0.076, -0.058)
β_5	Coefficient for Age-1+ abundance index on the Phase-2 growth increment	T2, T3	-0.044 (-0.051, -0.038)
β_6	Coefficient for month on the Phase-2 growth increment	T2, T3	-0.049 (-0.053, -0.045)
β_7	Coefficient for Age-0 abundance index * month interaction on the Phase-2 growth increment	T2, T3	-0.022 (-0.023, -0.021)
β_8	Coefficient for Age-1+ abundance index * month interaction on the Phase-2 growth increment	T2, T3	0.005 (0.004, 0.006)
β_9	Coefficient for *E. affinis* abundance index on the Phase-2 growth increment	T2, T3	-0.082 (-0.087, -0.078)
β_{10}	Coefficient for Mysis abundance index on the Phase-2 growth increment	T2, T3	-0.088 (-0.098, -0.078)
β_1^*	Coefficient for temperature on the Phase-1 growth increment	T3	-0.01 (-0.01, -0.009)
β_2^*	Coefficient for ln(outflow) on the Phase-1 growth increment	T3	0.148 (0.137, 0.16)

Par.	Description	Eq.	Estimate (95% CIs)
β_3^*	Coefficient for temperature * ln(outflow) interaction on the Phase-1 growth increment	T3	-0.026 (-0.029, -0.022)
β_4^*	Coefficient for Age-0 abundance index on the Phase-1 growth increment	T3	0.178 (0.168, 0.188)
β_5^*	Coefficient for Age-1+ abundance index on the Phase-1 growth increment	T3	0.146 (0.136, 0.155)
β_6^*	Coefficient for month on the Phase-1 growth increment	T3	-0.111 (-0.127, -0.096)
β_7^*	Coefficient for Age-0 abundance index * month interaction on the Phase-1 growth increment	T3	0.057 (0.054, 0.06)
β_8^*	Coefficient for Age-1+ abundance index * month interaction on the Phase-1 growth increment	T3	0.034 (0.031, 0.037)
β_9^*	Coefficient for *E. affinis* abundance index on the Phase-1 growth increment	T3	0.119 (0.114, 0.125)
β_{10}^*	Coefficient for Mysis abundance index on the Phase-1 growth increment	T3	-0.046 (-0.057, -0.035)
t_{crt}	Vertex of the Phase-2 temperature-growth relationship (in ^{o}C)	T2	-46.7 (-62.7, -30.8)
t_{crt}^*	Vertex of the Phase-1 temperature-growth relationship (in ^{o}C)	T3	22.8 (22.5, 23.1)
α_1	Coefficient for ln(1 + days over 20,000 cfs) on $L_{0,y}$	T4	-1.69 (-1.86, -1.52)
α_2	Coefficient for ln(conductivity) on $L_{0,y}$	T4	-24.9 (-27.8, -22.1)
α_3	Coefficient for ln(temperature) on $L_{0,y}$	T4	-14.2 (-16.9, -11.6)

Figures

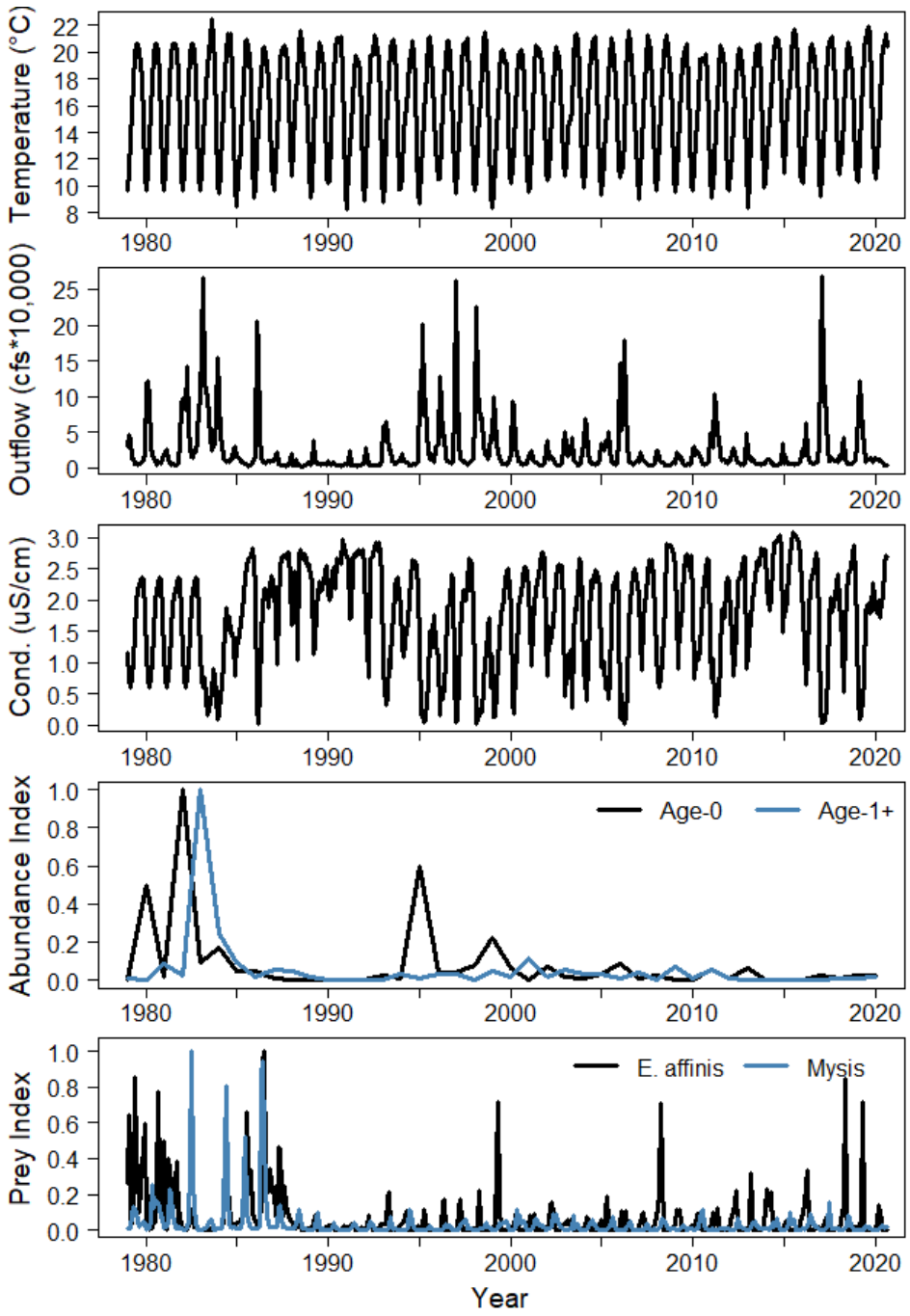


Figure 1: Time series of variables used or tested in the growth model from 1979–2020. Temperature and Conductivity were obtained from the Martinez (MRZ) long-term monitoring station. Delta outflow was obtained from the DayFlow dataset. Longfin Smelt abundance indices were the Age-0 and Age-1+ catch per unit effort (CPUE) trends from the San Francisco Bay Study otter trawl surveys. The Prey indices were the spatially averaged CPUE trends from CDFW zooplankton surveys.

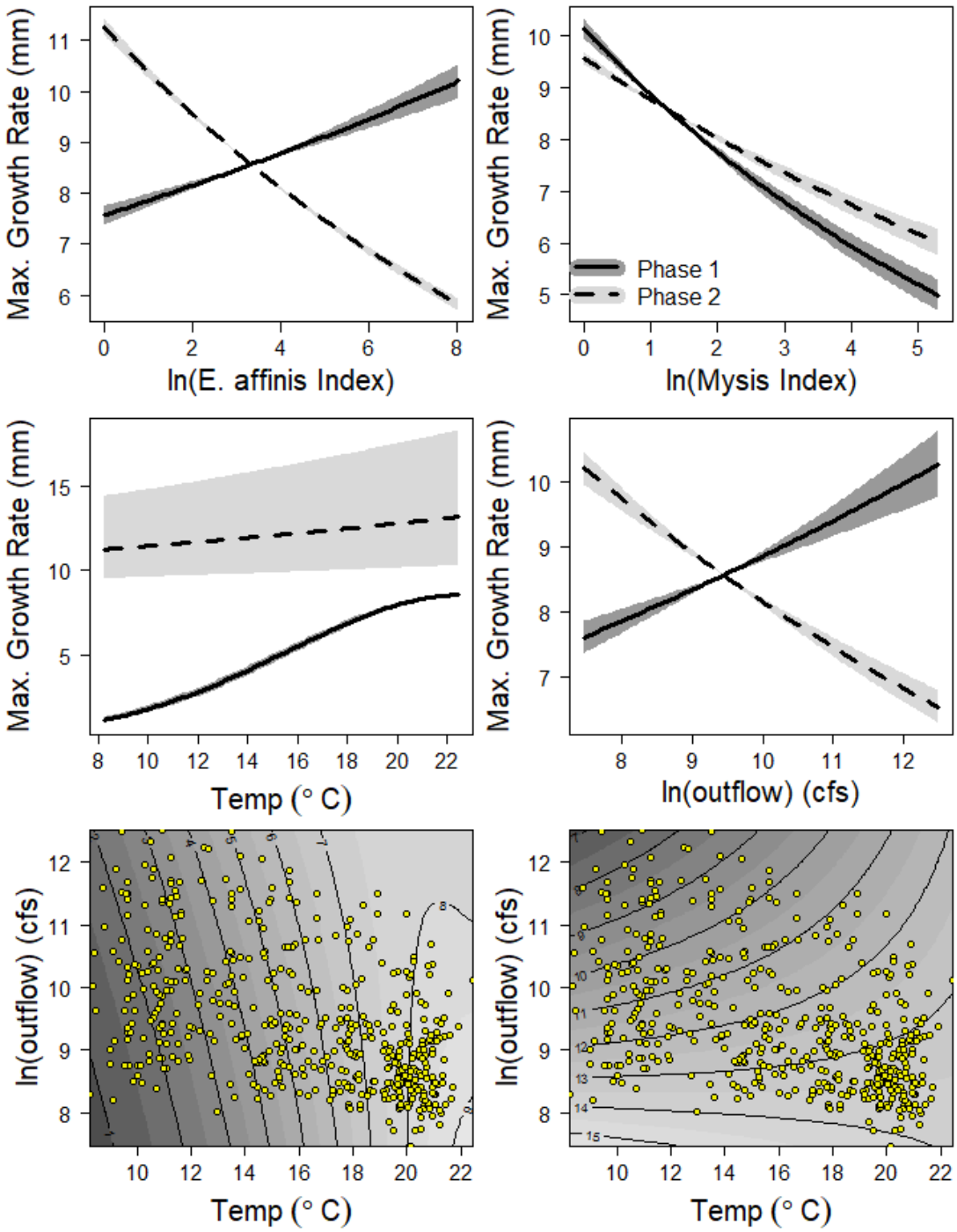


Figure 2: Predicted maximum monthly growth rate (in mm) across prey densities and environmental conditions. Each panel shows the effects across the range of observed environmental conditions while holding all other effects at the mean value. The top and middle rows depict the main effects (solid and dashed lines) and 95% confidence intervals (shaded regions) for Phase-1 and Phase-2 growth. The bottom row depicts the interaction between temperature and ln(Delta outflow) for Phase-1 (left) and Phase-2 growth (right) on maximum monthly growth rate (in mm). Yellow points represent observed temperature-outflow pairs.

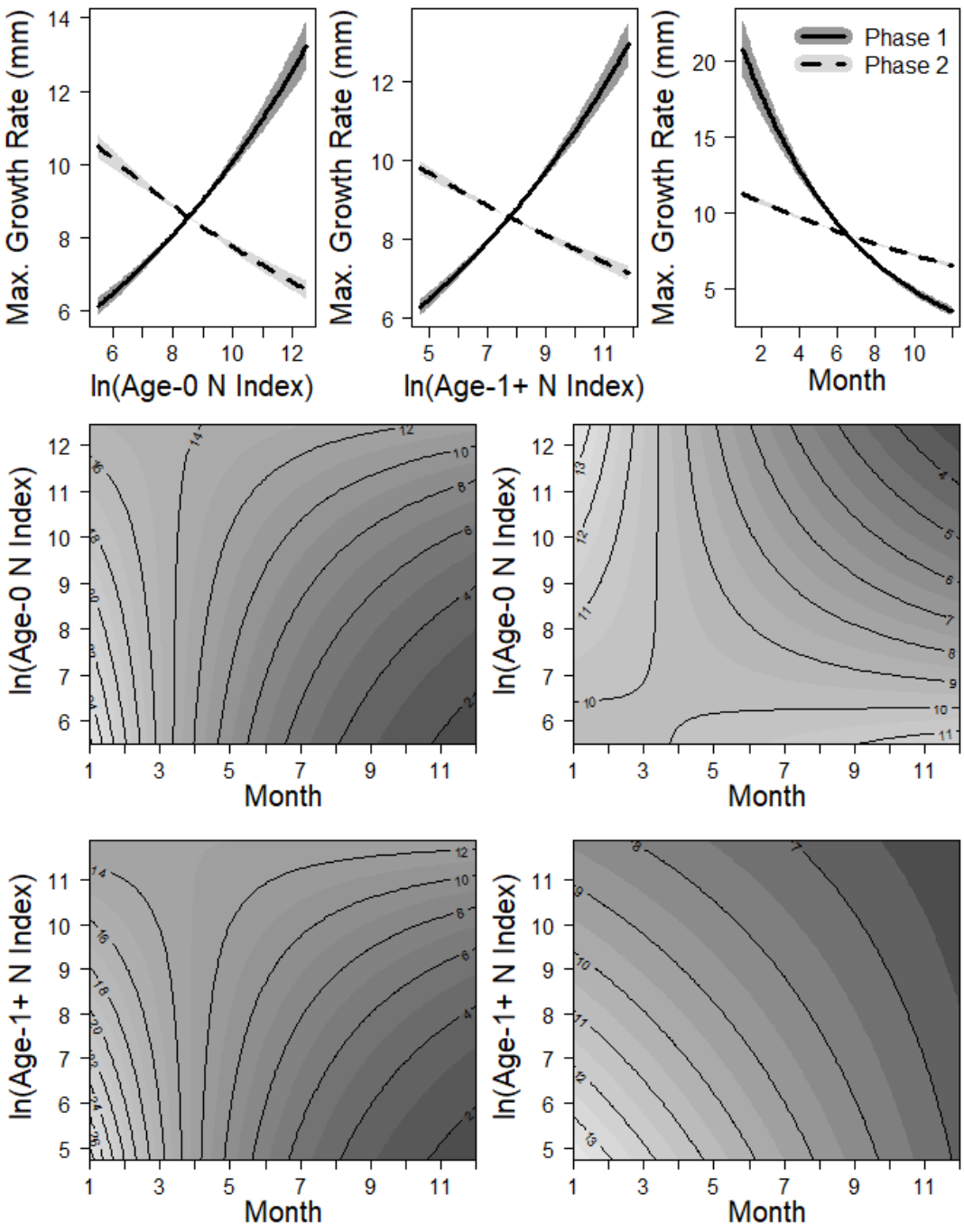


Figure 3: Predicted maximum monthly growth rate (in mm) across density and time. Each panel shows the effects across the range of observed conditions while holding all other effects at the mean value. The top row depicts the main effects (solid and dashed lines) and 95% confidence intervals (shaded regions) for Phase-1 and Phase-2 growth. The middle row depicts the interaction between Age-0 abundance indices and month of calendar year and the bottom row depicts the interaction between Age-1 abundance indices and month of calendar year for Phase-1 (left) and Phase-2 growth (right) on maximum monthly growth rate (in mm).

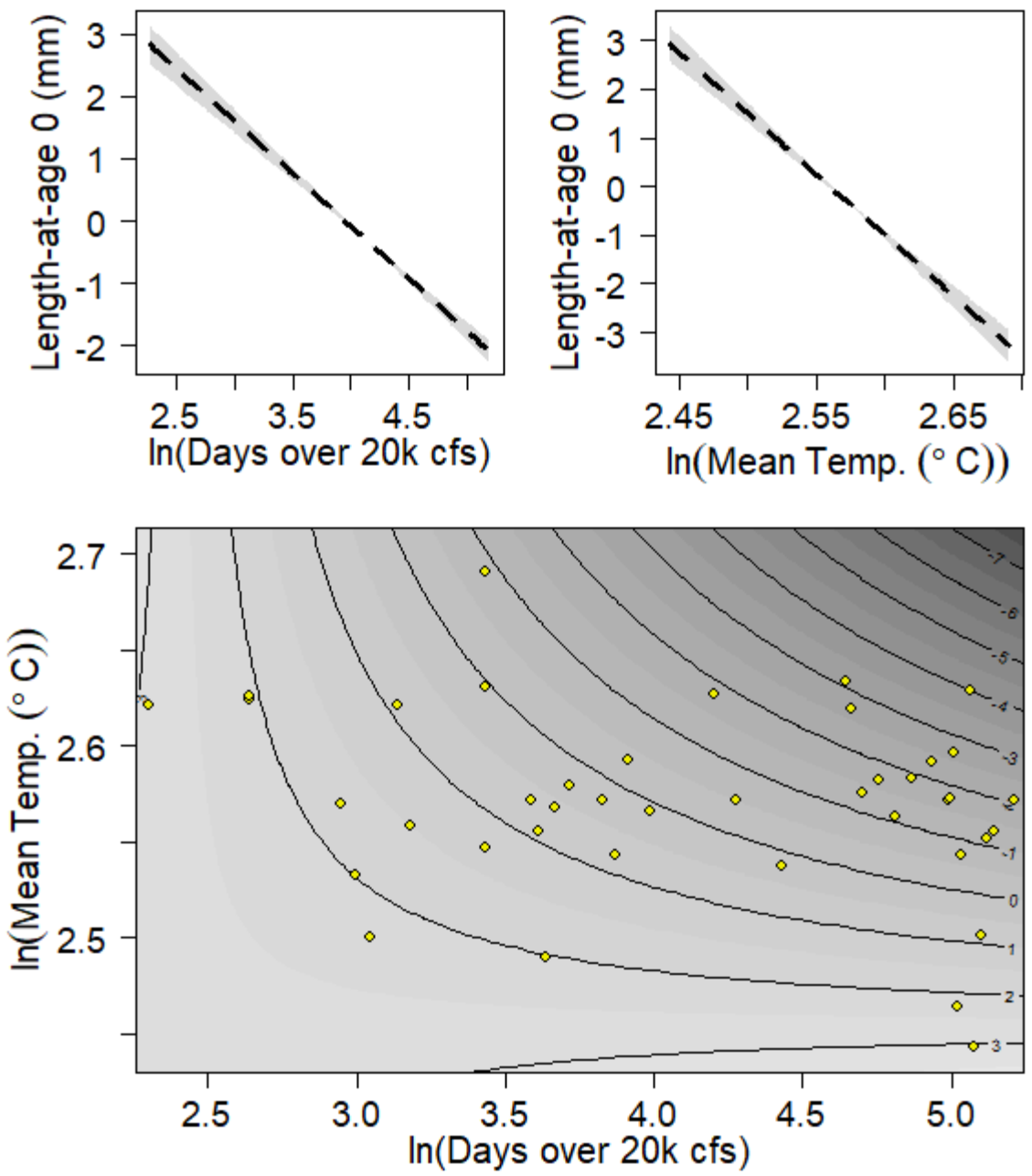


Figure 4: Predicted length-at-age-0 (in mm) across environmental conditions over the spawning season (Dec 1 – May 31). Each panel shows the effects across the range of observed environmental conditions while holding all other effects at the mean observed value. The top row depicts the main effects (dashed lines) and 95% confidence intervals (shaded regions). The bottom panel depicts the interaction between mean spawning temperature and number of days with recorded Delta outflow over 20,000 cfs on predicted length-at-age-0 (in mm).

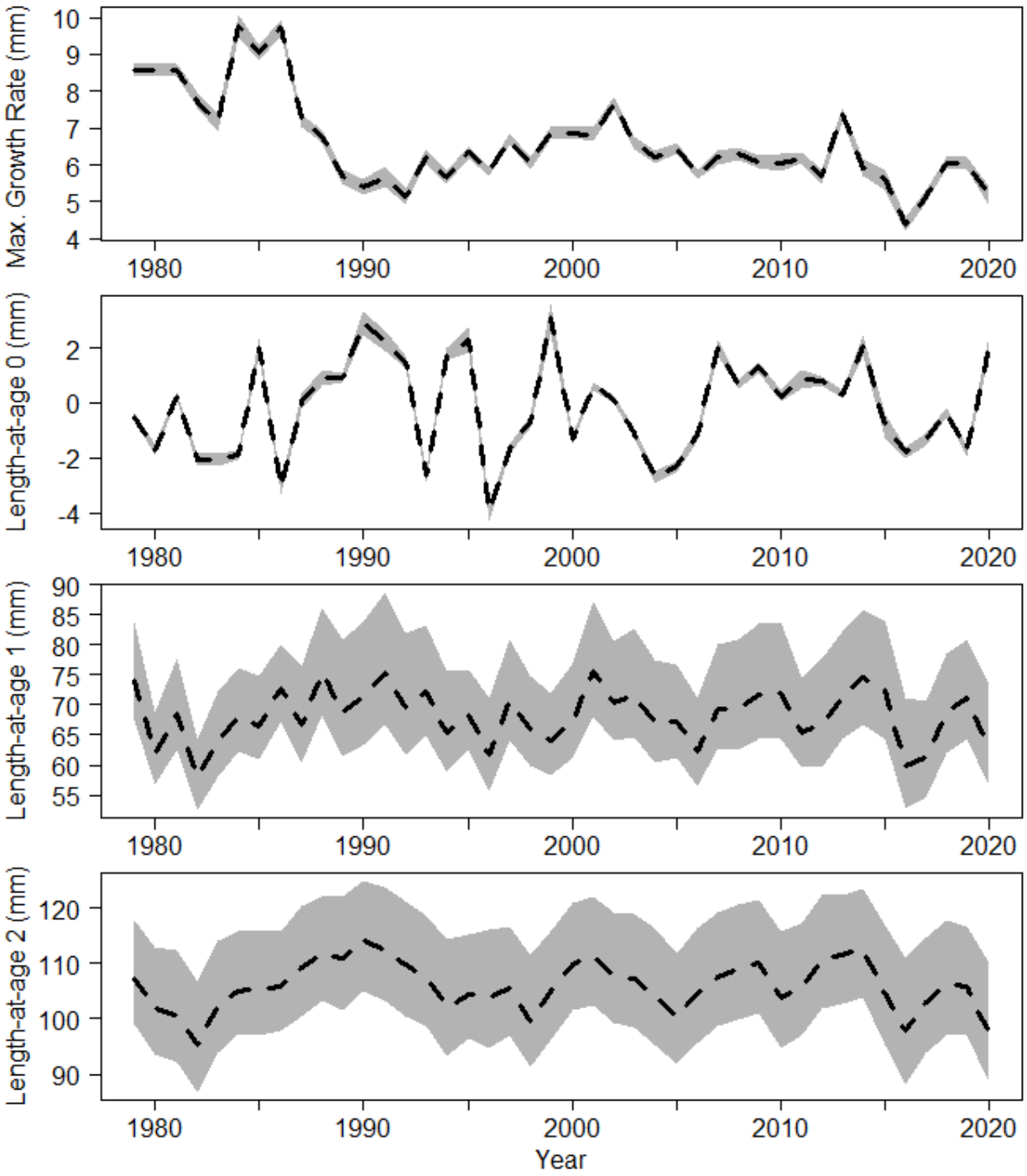


Figure 5: Predicted temporal trends in growth patterns from 1979 to 2020 (dashed lines) with 95% confidence intervals (shaded regions). The maximum growth rate per month (mm; top panel) accounts for variation in predicted non-reproductive growth (h; which is temporally constant in the best model) and a year-specific random effect predicting the unobserved annual variation in growth patterns. Panels for predicted length-at-age 0, 1, and 2 represent age in years (i.e., 0, 12, and 24 months).

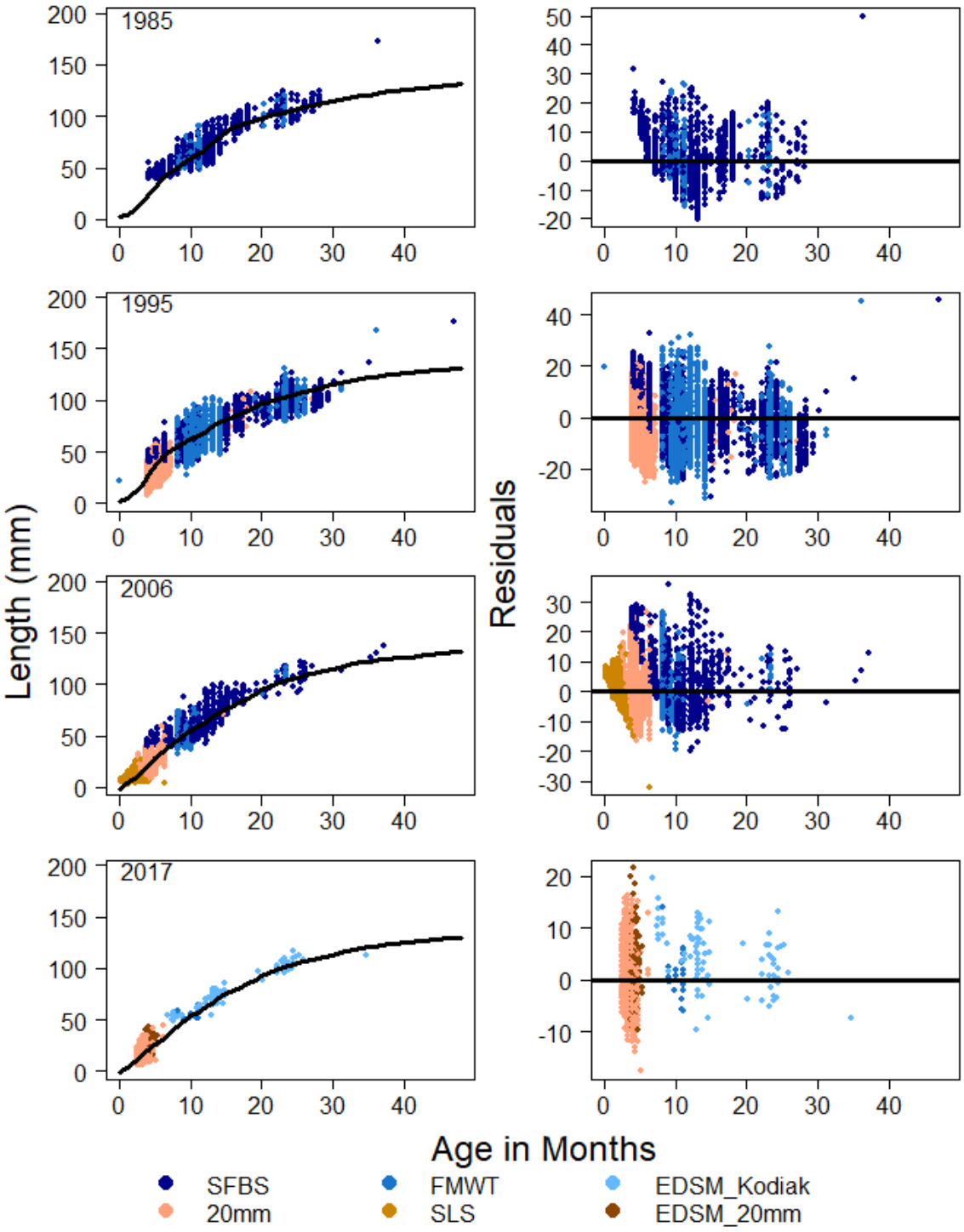


Figure 6: Predicted length-at-age (left side) and residuals (right side) from four representative cohorts depicting model fits. Solid lines represent model predictions and points represent individual length-at-age observations. Blue colors represent sampling programs focused on larger-sized individuals and brown colors represent sampling programs focused on larval individuals.

Technical Appendix E: A State-Space Model for Bay-Delta DPS Population Dynamics

Version 3.2.0 – May 20, 2022

Vanessa Tobias, USFWS

Introduction

In an earlier technical note (TN 1; Tobias 2022), I developed a count-based PVA for several long-term monitoring surveys showing that Longfin Smelt populations are likely to fall below quasi-extinction thresholds in the foreseeable future. In TN 1, each abundance index was used as an independent measurement of annual population growth, as measured in a specific LFS life stage. The count-based PVA did not account for the three-year life span of LFS, the population structure, or annual variation in vital rates.

This technical note aims to account for the structure within the LFS life cycle by proposing a matrix model for population growth with three age classes (life stages;

Figure **11**). The vital rates in the matrix are estimated using a state-space model of population dynamics. Previous research on population dynamics of Longfin Smelt developed models using two life stages or ages of Longfin Smelt. In an extensive model comparison exercise, a state-space model formulation of the Beverton-Holt stock-recruit relationship gave the best explanation for the relationship between juvenile and adult Longfin Smelt abundance values (Maunder 2015). Further research, which used the Ricker model for the stock-recruit function, estimated the effects of environmental factors on juvenile survival and recruits per spawner (Nobriga and Rosenfield 2016). The model for Longfin Smelt population dynamics that I developed for this technical note builds on this previous research to investigate how environmental factors influence the transitions between three age classes.

Methods

Data

Data on relative abundance of Longfin Smelt came from the San Francisco Bay Study's age-specific abundance indices (CDFW 2021). For the environmental covariates, I compiled a dataset of environmental variables on an annual time step from the Department of Water Resources water quality monitoring programs (including temperature and salinity), and the Environmental Monitoring Program's zooplankton data, DayFlow, and ocean upwelling index data from NOAA's monitoring programs.

Data Exploration

I graphed the Bay Study indices on a log scale to visually assess the shapes of the apparent vital rates. This assessment is meant to provide general guidance for the structure of the state-space model.

State Space Model

I fit models for Longfin Smelt population dynamics as state-space models in JAGS (Plummer 2003) via R (package R2jags; Su and Yajima 2015). State-space models are a class of models where there are two processes: a state process and an observation process. The state process is the underlying pattern or time series that is hidden in the data (i.e., the latent state). The collected data enter the model through the observation process, which assumes that the data points are independent, but noisy observations of the latent state. I used R (version 4.0.4 "Lost Library Book"; R Core Team 2021) to fit all models and make calculations.

There are several advantages of using a state space model for this application. First, the state space model framework allows estimation of observation error and process error, which can improve parameter estimation and variable selection (see Maunder et al. 2015). The ability to describe the relationships in a hierarchical structure allows simultaneous estimation of vital rates and the influence of covariates on those rates. The state-space model framework also offers the ability to include information from multiple gear types into estimating the underlying abundance pattern by including an observation process for each gear type and age class.

The conceptual model for the population dynamics represented in the model is illustrated by

Figure 11. I fit an initial model with three states (one for each age class) with two observation processes for each state (one for each of the gear types for each age class) that did not include any environmental covariates. This model served as a baseline for building the relationships and it provided initial estimates of the variability and structure of the vital rates.

The notation for age classes will be as follows: N represents the estimated abundance (latent state) and the subscripts index the age followed by t for time (in years).

Age-0 survival (S_0; age-0 to age-1) and age-1 survival (S_1; age-1 to age-2) were structured as simple ratios using a logit-normal link:

$$\text{logit}(S_{0,t}) = \frac{log(N_{1,t+1})}{log(N_{0,t})} + \varepsilon_{S0,t}\ ;\ \varepsilon_{S0,t} \sim N(\mu_{\text{S0t}},\ \sigma_{\text{S0}t})$$

$$\text{logit}(S_{1,t}) = \frac{log(N_{2,t+1})}{log(N_{1,t})} + \varepsilon_{S1,t}\ ;\ \varepsilon_{S1,t} \sim N(\mu_{\text{S1t}},\ \sigma_{\text{S1}t})$$

Reproduction was structured as a Beverton-Holt equation to represent the relationship between age-2 abundance and age-0 abundance:

$$\log(N_{0,t}) = \log\left(\frac{a_t \cdot N_{2,t}}{1 + b_t \cdot N_{2,t}}\right) + \varepsilon_{N0};\ \varepsilon_{N0,t} \sim N(\mu_{N0},\ \sigma_{N0,t})$$

where *a* describes density-independent recruitment which is proportional to fecundity, i.e. recruitment when densities are low, and *b* describes the degree of density-dependence. In the code, the model was fit using an alternate parameterization of the Beverton-Holt equation

$$\log(N_{0,t}) = \log\left(\frac{N_{2,t}}{a'_t + b'_t \cdot N_{2,t}}\right) + \varepsilon_{N0}\ ;\ \varepsilon_{N0,t} \sim N(\mu_{N0},\ \sigma_{N0,t})$$

where *a'*= 1/*a* and *b'*=*b*/*a*. Note that using the Bay Study abundance indices, the relationship between recruits and spawners comes from the abundance indices from the same year. This is different than the survival relationship, which has a one-year time lag in the relationship. The current version of the model only allows reproduction by age-2 fish (

Figure 11). Future model development may include reproduction by a portion of age-1 fish. Each state equation has a separate estimate for the process error, which allows each age class to have its own estimate of variability.

Without environmental covariates, the estimates of certain vital rates were diffuse (especially survival rates), and in the case of the *a* parameter from the Beverton-Holt equation for reproduction, the model was unable to distinguish between three likely estimates.

Subsequent models included linear models for the effects of environmental covariates on vital rates as a higher level in the hierarchical model structure. These linear models can be thought of as random effects, as the annual estimates of vital rates are treated as random variables in these models. Variables were standardized (centered on the mean and scaled by the standard deviation) prior to inclusion.

There are more environmental variables available for consideration than is prudent to include in a model, so I employed the following variable selection method. The dataset of environmental variables contains several groups of variables that measure the same or similar aspects of the environment (for example, sea surface temperature at two buoys that are near each other and various measurements of outflow volume). I selected variables for consideration such that repeated information was limited--attempting to select one variable to represent each group. Next, I used a stochastic search variable selection method (SSVS) within the model to select informative variables. To do this, I included the chosen variables in the state process with spike and slab priors (O'Hara and Sillanpää 2009) which allow the model to estimate the probability that they should be included (i.e., that their inclusion explains variability). The priors included a random effect for the number of variables that should be included in the model. This random effect had a prior that peaked around 20% with long tails to allow the number of parameters to vary as needed (dbeta(2,8)), although the selection of 20% as the peak was somewhat arbitrary.

The observation portion of the model incorporates data from both of the Bay Study's gear types, with separate estimates of observation error for each combination of gear and age class (six total observation errors). Separate observation errors account for variability in the relationships between the two gear types. I also included a scaling factor that accounts for large-scale differences in observations between the two gear types for each age class. Currently, each age class has a single scaling factor, but future

versions of the model may include a time-varying factor to account for the apparent changes in the expected catch ratios between the two gear types over time (

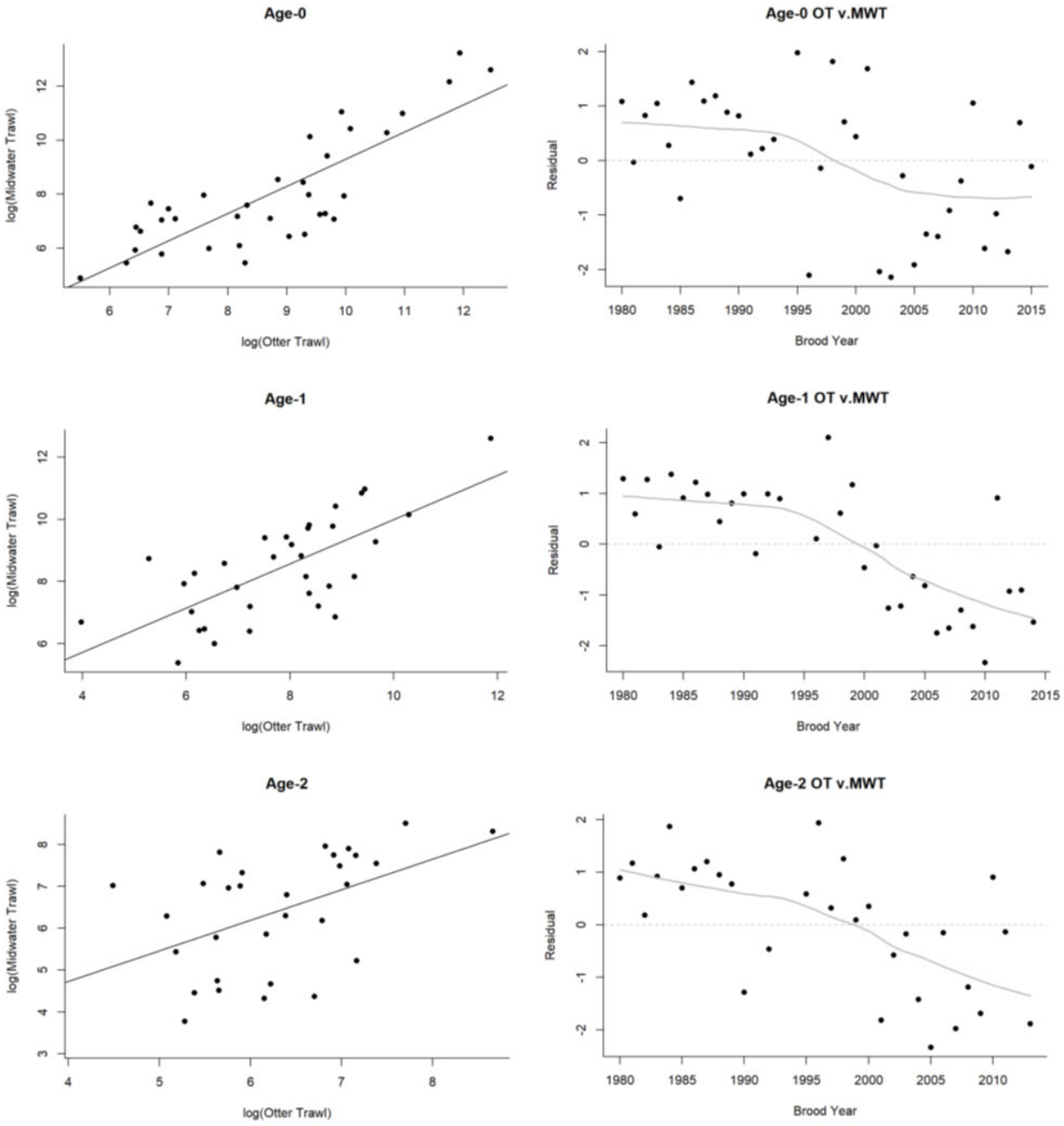


Figure 14). Variables can also be included in the observation process to identify factors that influence the observations. I chose not to do this at this stage of model development, but future development may include a structure to account more explicitly for the patterns in the relationships between the gears.

Forecasting Extinction Risk

In addition to understanding the drivers of the observed patterns in abundance for each age class, it is useful to forecast abundance into the future. To do this, I used the state equations from the state-space model as the framework and a random draw of the posterior estimates for the vital rate parameters and the starting abundance. I drew the starting abundances from the estimated age-specific abundances for 2015. I used these random draws to run the autoregressive equations forward to 2060. I repeated the procedure (including draws for starting abundances and vital rates) 1000 times to produce a distribution of abundance values for each age class at each time point.

I incorporated quasi-extinction into the forecasting algorithm by setting minimum values for each age class. If the estimated abundance fell below the threshold for quasi-extinction, the estimate for that age class was set to zero. This propagated through the forecasts such that if any age class met the criteria for quasi-extinction, abundance estimates for each of the age classes would eventually go to zero. There was no obvious or empirical level to set for quasi-extinction in this model, so I ran the simulation with four quantiles at the low end of the abundance estimates for comparison (

Table 8). In Technical Note 1 (Tobias 2022), I used 1% of the most recent ten years of the surveys as the threshold for quasi-extinction. One way to address this is to use the historical environmental variables to hindcast abundance. This can then be qualitatively compared to the abundance indices to give an idea for how well the estimates predict the observed patterns in abundance. I graphed the results of hindcasting with various abundance quantiles as quasi-extinction values and included the age-0 abundance indices for reference. I used age-0 because they usually have the highest values of the three age classes and have the widest range of values.

Results

Data Exploration

Rates of survival appear to be adequately described by simple rates, rather than an equation for density dependence (

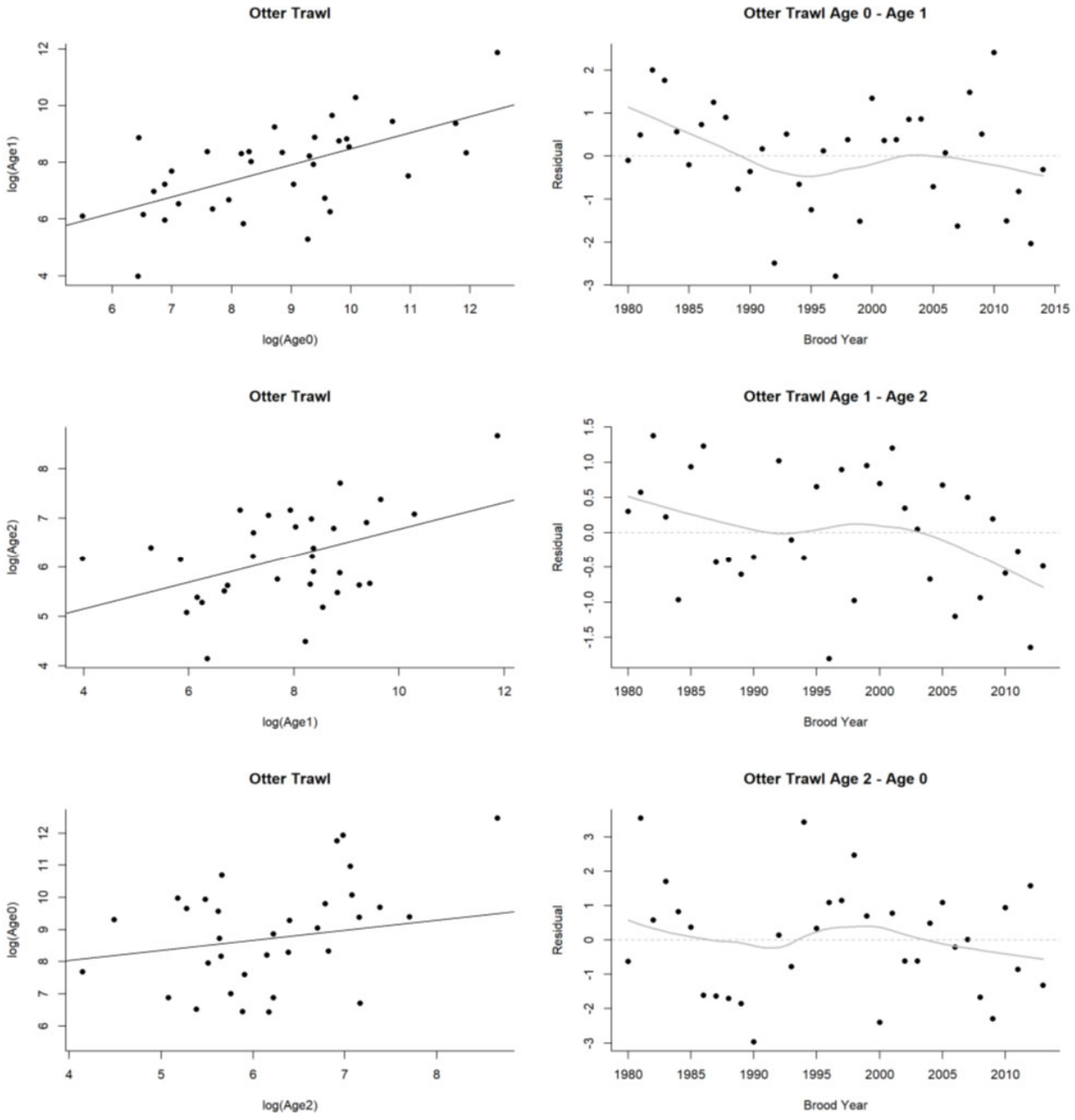


Figure 12 and

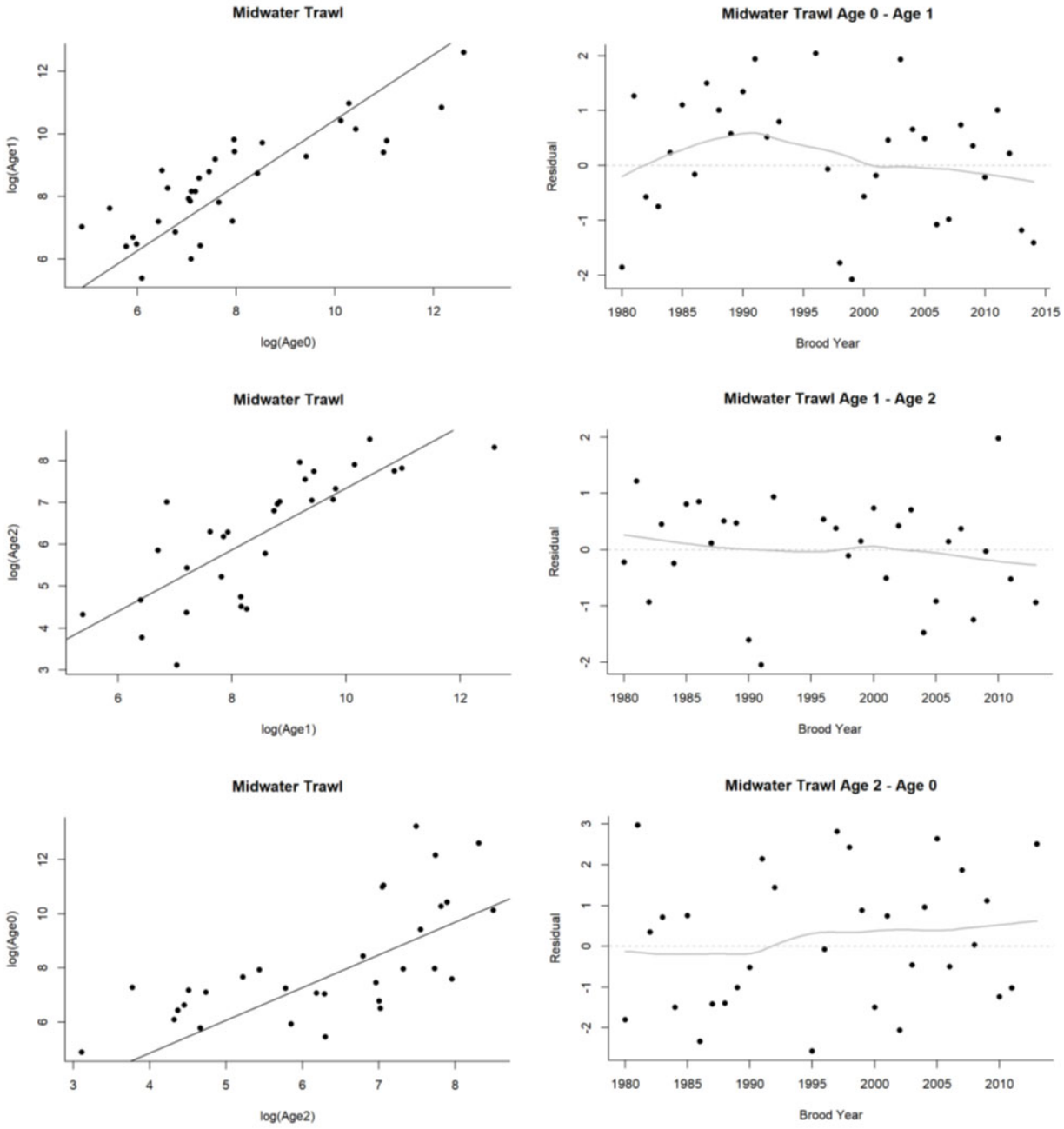


Figure 13). Patterns in the residuals for those relationships suggest that survival rates are not static and that covariates are necessary to fully understand what drives population dynamics. For example, survival rates tend to be lower in more recent years than predicted by the regression analysis that estimated a single static rate of survival for all years. A simple rate does not appear to adequately describe the shape of the reproduction relationship. For the otter trawl indices, the slope of the regression was not significantly different from zero (

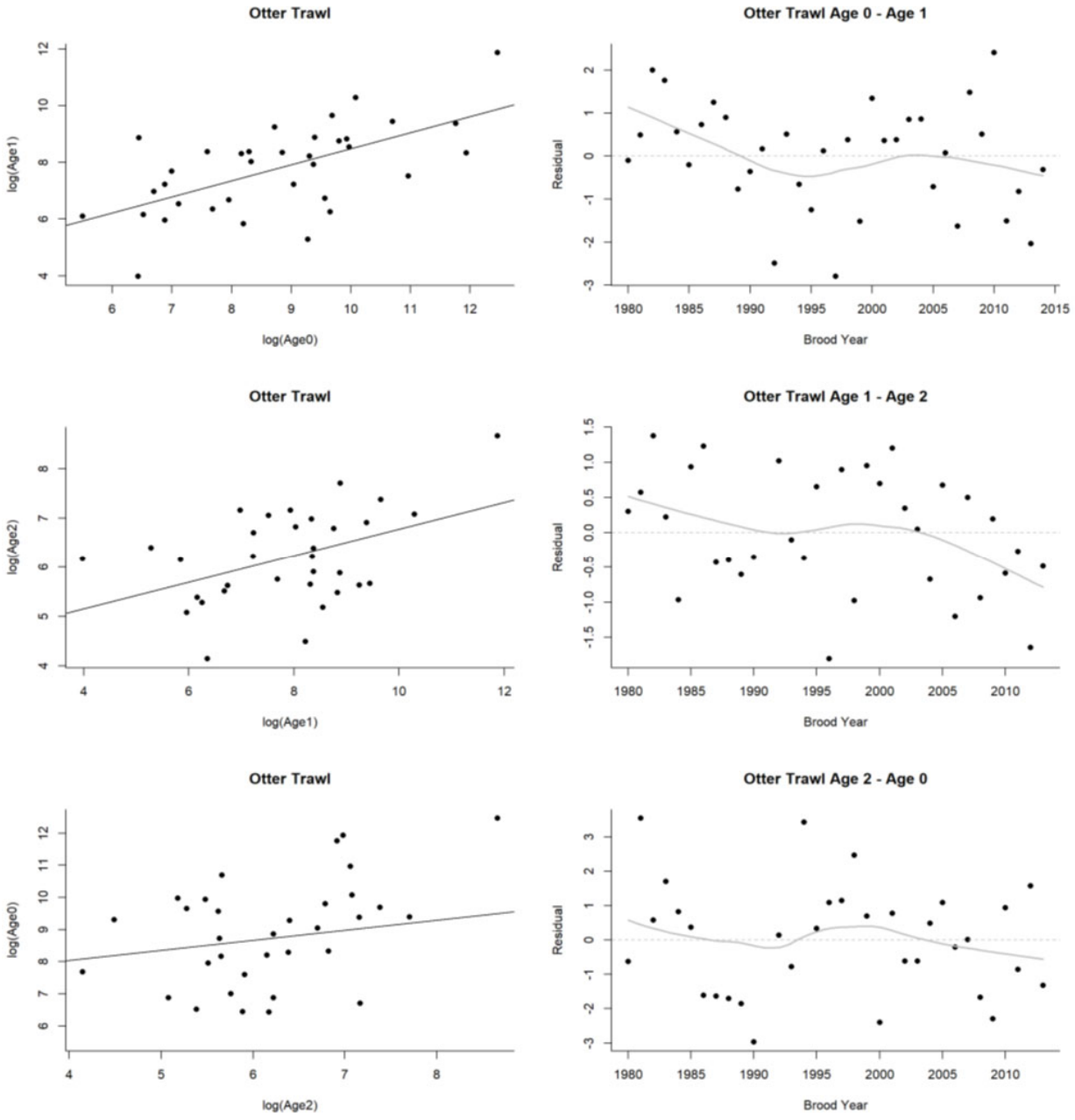


Figure 12); although the slope of the regression for midwater trawl was significantly different from zero, the data points clearly deviate from a linear relationship (

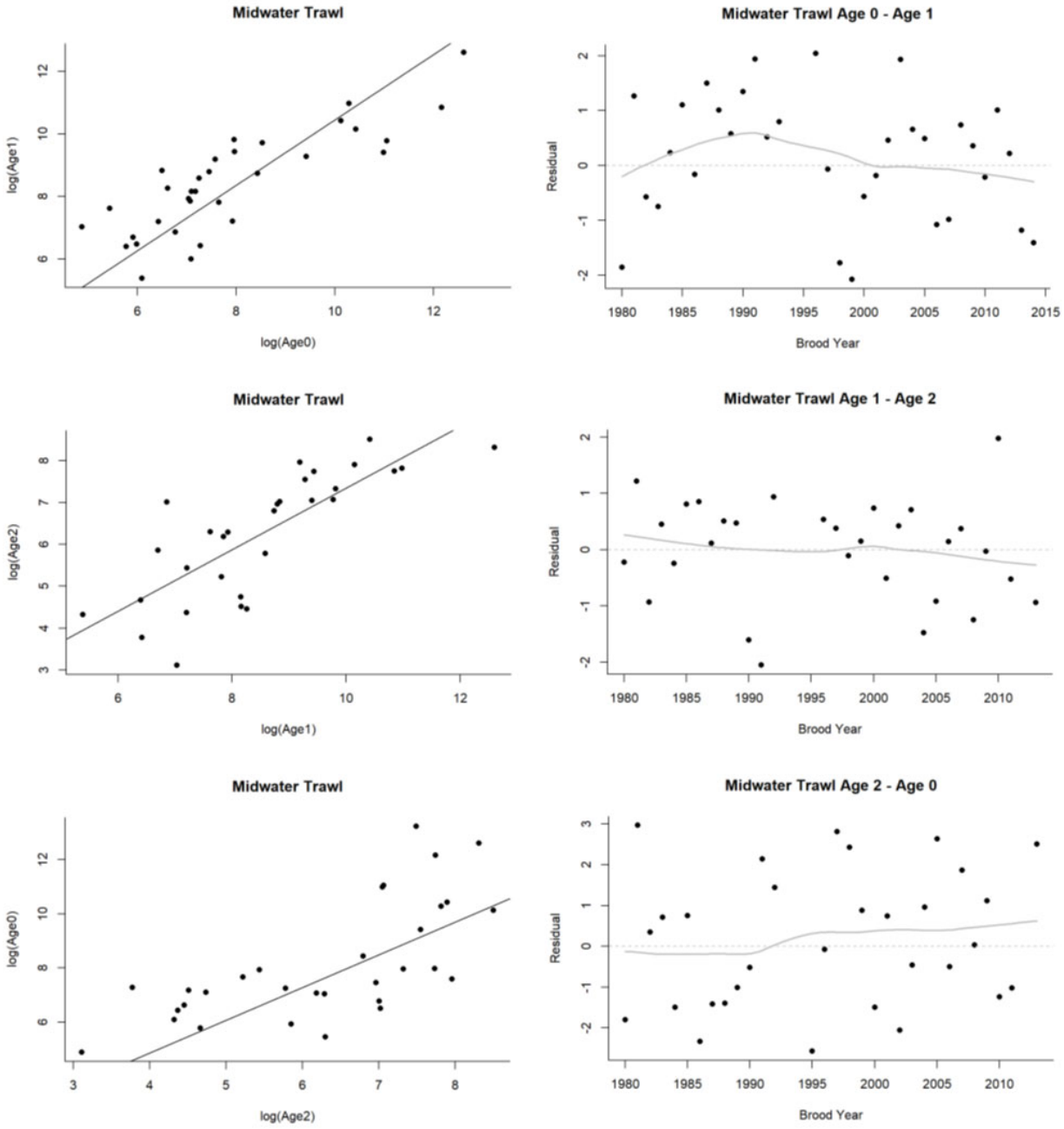


Figure 13). In the data exploration, I use age-2 fish as a proxy for spawning stock because most spawning is thought to occur at age-2. I recognize that some age-1 fish may also be spawning so the relative abundance of spawners may be higher than is represented here. Also note that the age-2 abundance index includes both male and female fish, so reproduction described here is not directly comparable to female fecundity rates. The sex ratio for adult Longfin Smelt is roughly equal (IEP MAST), so multiplying rates by two would be more similar to estimates of female fecundity.

The relationship between the two timeseries of observations created by the two gear types is somewhat complicated (

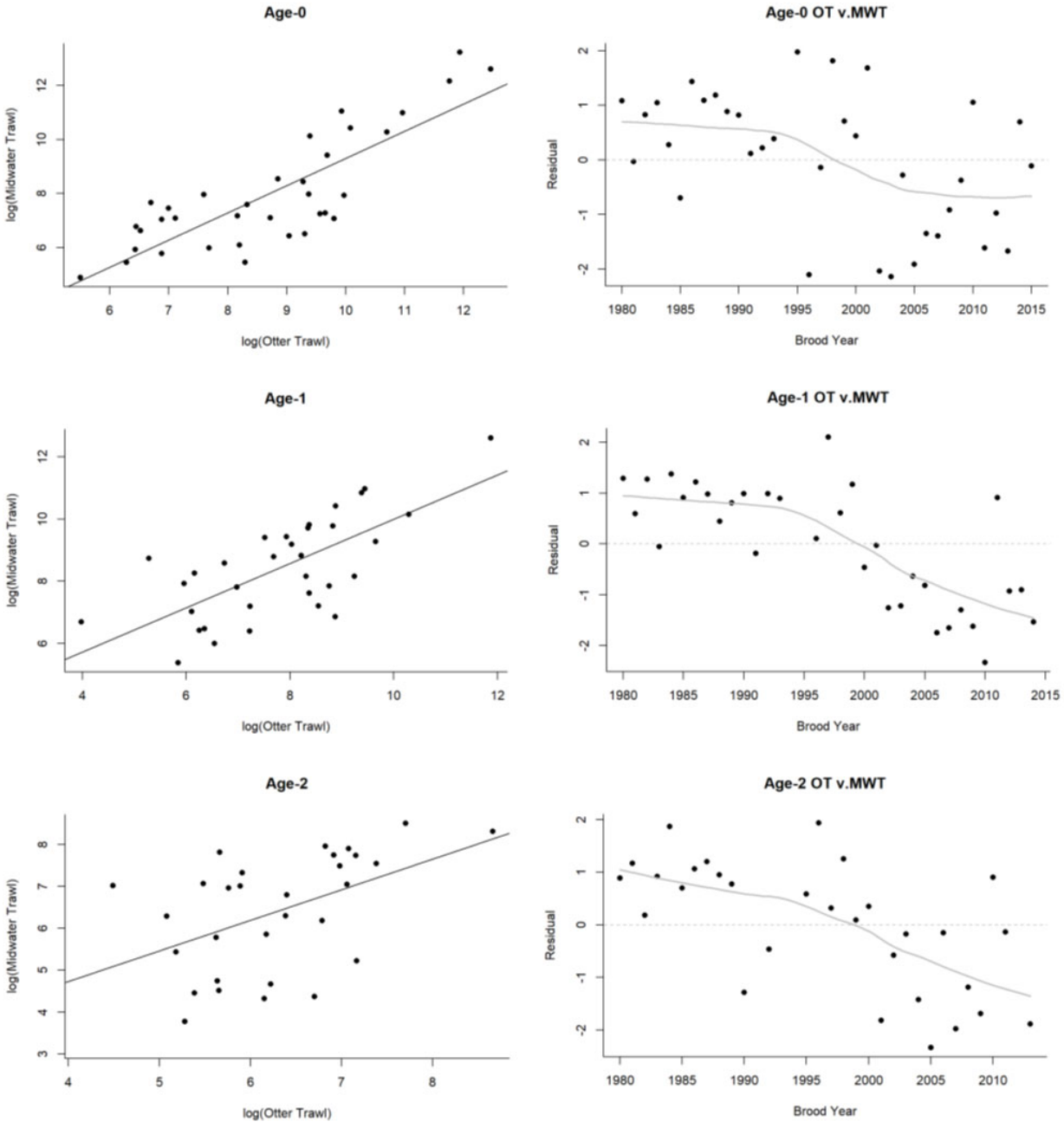


Figure 14). A simple ratio can describe much of the relationship for the gear-specific abundance indices for each of the age classes, but residuals for each age class show a distinct pattern over time.

Population Model

The estimated latent states for the relative abundances follow the patterns in the time series of the Bay Study abundance indices (**Figure 17**). The estimated states are also plausible, in that the highest abundance values are generally for the age-0 class and decline across cohorts.

Using the SSVS procedure, a small number of covariates were selected to explain patterns in each of the vital rates for some of the parameters, but for other parameters several variables appeared to have similar chances for inclusion (**Figure 15**). The range of inclusion probabilities for covariates varied somewhat across the vital rates. The density dependence portion of the reproduction equation had variables with the strongest evidence for inclusion while the other vital rates all had inclusion probabilities close to 0.2, which was the mean value for the prior distribution that was set for inclusion probabilities.

The time series for each of the vital rates after refitting the model with the selected environmental variables is shown in **Figure 16**. There are a few patterns that emerge. The mean of the density-independent reproduction parameter (*a*) was steady over time, but the estimates have substantial variability. The degree of density dependence (*b*) appears to increase over the length of the time series but is interrupted by a sharp decline in the mid-1990s. Age-0 survival is high, largely over 0.85, until the early 1990s, after which the mean survival appears to decline. The mean line has several peaks and valleys after the mid-1990s which make the survival estimates appear more chaotic. The biggest valley is around 2008, but smaller dips occur in 1996, 2000, and 2013–2014. Age-1 survival is fairly steady over time, but there are small fluctuations form year to year and valleys are visible in 1993 and 2014.

Forecasting Extinction Risk

The median total abundance (sum of abundance for all age classes) forecasted by the simulations reached a peak within a few years of the starting year and then declined to near zero by the end of the time series (

Figure 18). The confidence bands were larger for larger mean values, and they tapered down as mean abundance declined. These simulations can also be visualized as the proportion of simulation runs that fall below a threshold for quasi-extinction. The proportion of simulations that fall below the quasi-extinction threshold increases over time for all of the thresholds, from 1% to 15% of the population estimates (**Figure 19**). The trajectory for the most conservative threshold (15%) increases steeply for several years before leveling out slightly and reaching a maximum value of approximately 80% in 2060. The trajectory for the riskiest threshold (1%) is less steep and reaches a maximum value of approximately 20% by 2060.

Hindcasting simulations indicated that 20% was too high to be used as a quasi-extinction threshold because simulations uniformly failed to recover from a population crash in 1990, but lower thresholds were able to recover (**Figure 20**). The 1% threshold appears to be over-confident for population increases in the most recent years. For example, small upticks in the abundance indices in 2006 and 2011 correspond with big increases in estimated abundance with narrow confidence intervals. The 15% threshold appears to strike a reasonable balance.

Discussion

Taken together, these trends create a negative feedback loop with decreasing reproduction and decreasing survival. Throughout model development, although some details varied depending on the

model structure or covariates included, the results have always indicated a change in vital rates in the early to mid-1990s. Based on the current model formulation, a change in parameters associated with reproduction occurred at time. The most likely covariates to explain this change are related to hydrology (outflow and inflow), salinity, and food availability. In addition to declining reproduction, declining survival rates over time indicate that the chances that the recruits survive to spawn themselves is also declining. There are several years in the time series when poor conditions converge to negatively affect multiple vital rates simultaneously. For example, 1993 and 2014 saw low survival rates for both age-0 and age-1 Longfin Smelt.

Forecasts of population size using vital rates estimated by the model indicate that it is likely that Longfin Smelt population sizes will dip below recoverable levels within a decade if these recent levels of reproduction and survival continue. Results were similar across a range of quasi-extinction thresholds. I used the estimates of vital rates from all years of the available surveys (i.e., 1980 to 2015). Using all available years rather than just the most recent years may introduce more variability into the forecasted population size than is realistic if there are vital rates that have trends over time and if the earlier part of the time series does not reflect conditions moving forward. In particular, the effect of density dependence (b) and age-0 survival (S_0) may be affected by this choice of years because of the apparent trends in their estimated time series since about 2000 (**Figure 16**). Adding variability to the vital rates affects the confidence intervals on the predictions, but it also may affect the estimates of population viability over time. Populations with highly variable vital rates tend to have lower levels of population viability over time because adding variation tends to make a population grow more slowly over the long term and the population size is more likely to fall below the quasi-extinction threshold than it is for populations with less variable growth rates (Morris and Doak 2002).

Although these results imply that if future vital rates are similar to those observed in the past there is substantial risk of extinction for Longfin Smelt, they also imply potential mechanisms for recovery. These modeling results suggest that increasing reproductive rates in consecutive years can reset the population to higher levels. Results also suggest that environmental conditions may affect several vital rates simultaneously, which would be reflected in the abundance trajectories for multiple life stages.

Acknowledgements

This modeling effort benefitted from the advice and prior work of many people. The Core Team for the Longfin Smelt Species Status Assessment gave feedback on several iterations of the modeling framework and interpretation of results. The Core Team includes J. Hobbs, M. Eakin, and E. Chen. J. Miller compiled environmental data from several sources. J. Hobbs, K. Hieb, and J. Burns provided abundance index data. B. Matthias gave feedback on model structure and processed environmental data for use on an annual time step. Code for the Delta Smelt life cycle model by K. Newman, L. Polansky, and L. Mitchell was also a helpful reference for code formulation.

R code for the analyses and graphs can be found at https://github.com/USFWS/Longfin-Smelt-Population-Dynamics/

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Tables

Table 7: Environmental variables

Category	**Variable**	**Details**	**Data Source**
Hydrology	Outflow	Median annual outflow	Dayflow dataset
	Inflow	Median annual inflow	Dayflow dataset
Water Quality	Low Temperatures	Count of days < 15° at Martinez (MRZ)	DWR water quality dataset
	High Temperatures	Count of days > 20° at Martinez (MRZ)	DWR water quality dataset
	Salinity	Count of days < 2 PSU at Martinez (MRZ)	DWR water quality dataset
Prey Availability	Copepods	Eurytemora gC/m^3 in Suisun Bay (NZ048) in March	Environmental Monitoring Program dataset
	Mysids	Mysid CPUE in Suisun Bay (NZ048) in November	Environmental Monitoring Program dataset
Ocean Temperature	Upwelling Index	Mean upwelling index in August	NOAA dataset

Table 8: Values for abundance quantiles, some of which were used as quasi-extinction thresholds to calculate risk of quasi-extinction in **Figure 19**.

Quantile	Age-0	Age-1	Age-2
50%	4185	1616	161
20%	1801	451	61
15%	1600	338	49
10%	1399	245	38
5%	1171	162	26
1%	868	80	13

Figures

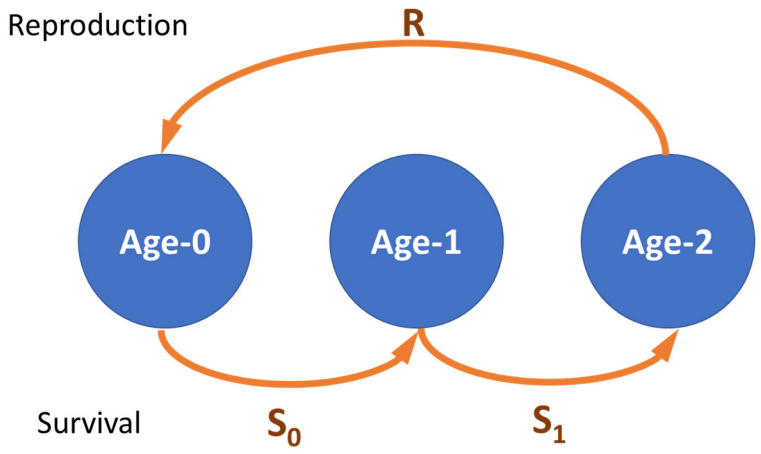


Figure 11: Age-based conceptual model for Longfin Smelt population dynamics.

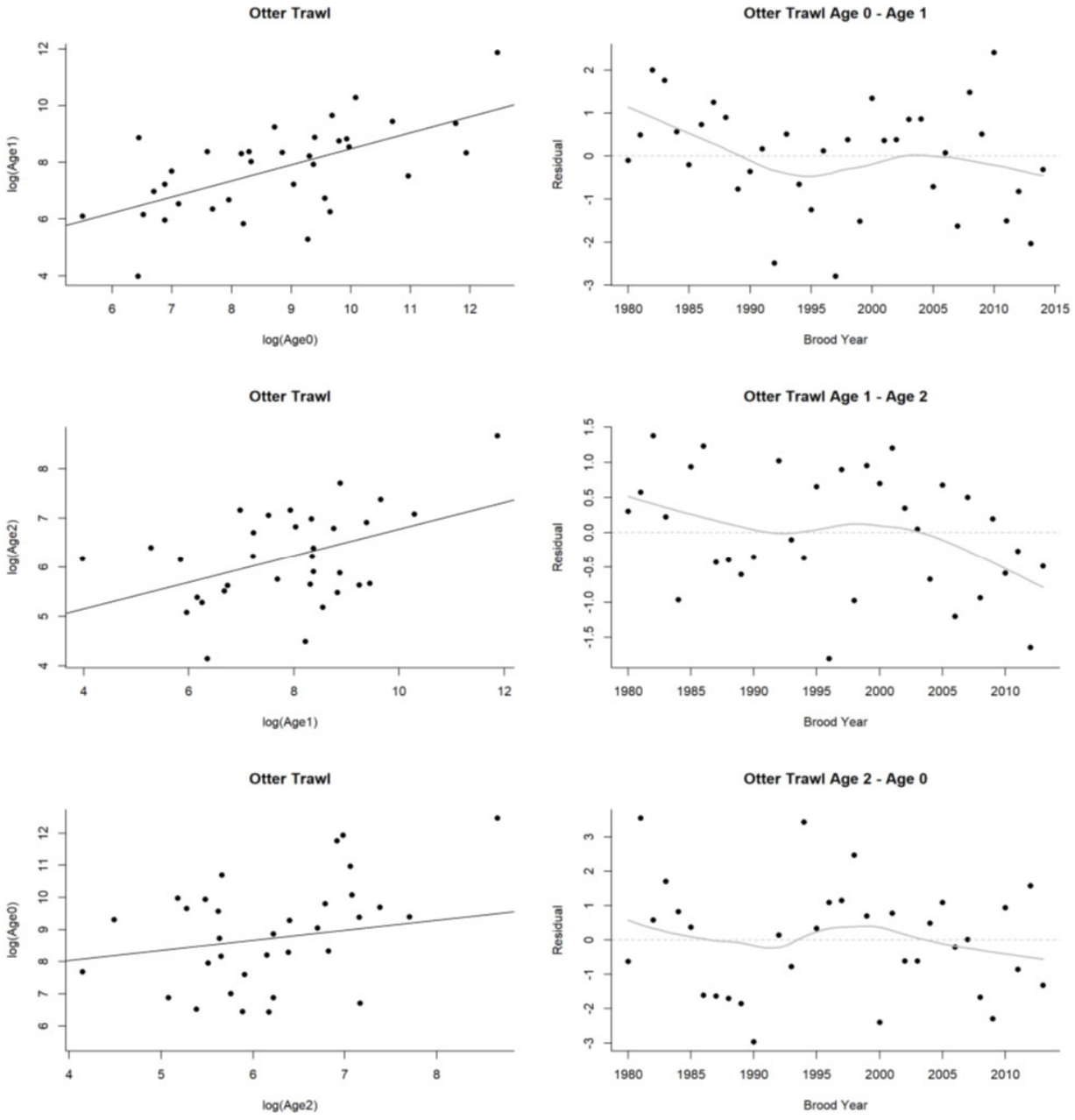


Figure 12: Relationships between otter trawl age class indices on a log scale provide a visual assessment of vital rates. Left column (top to bottom): Age-0 survival, Age-1 survival, and reproduction. Solid lines represent linear regressions. Survival plots have slopes with $p < 0.05$; reproduction plot $p > 0.05$. Right column (top to bottom): residuals for the linear regressions in the left column, plotted over time. Solid lines represent a loess smoother.

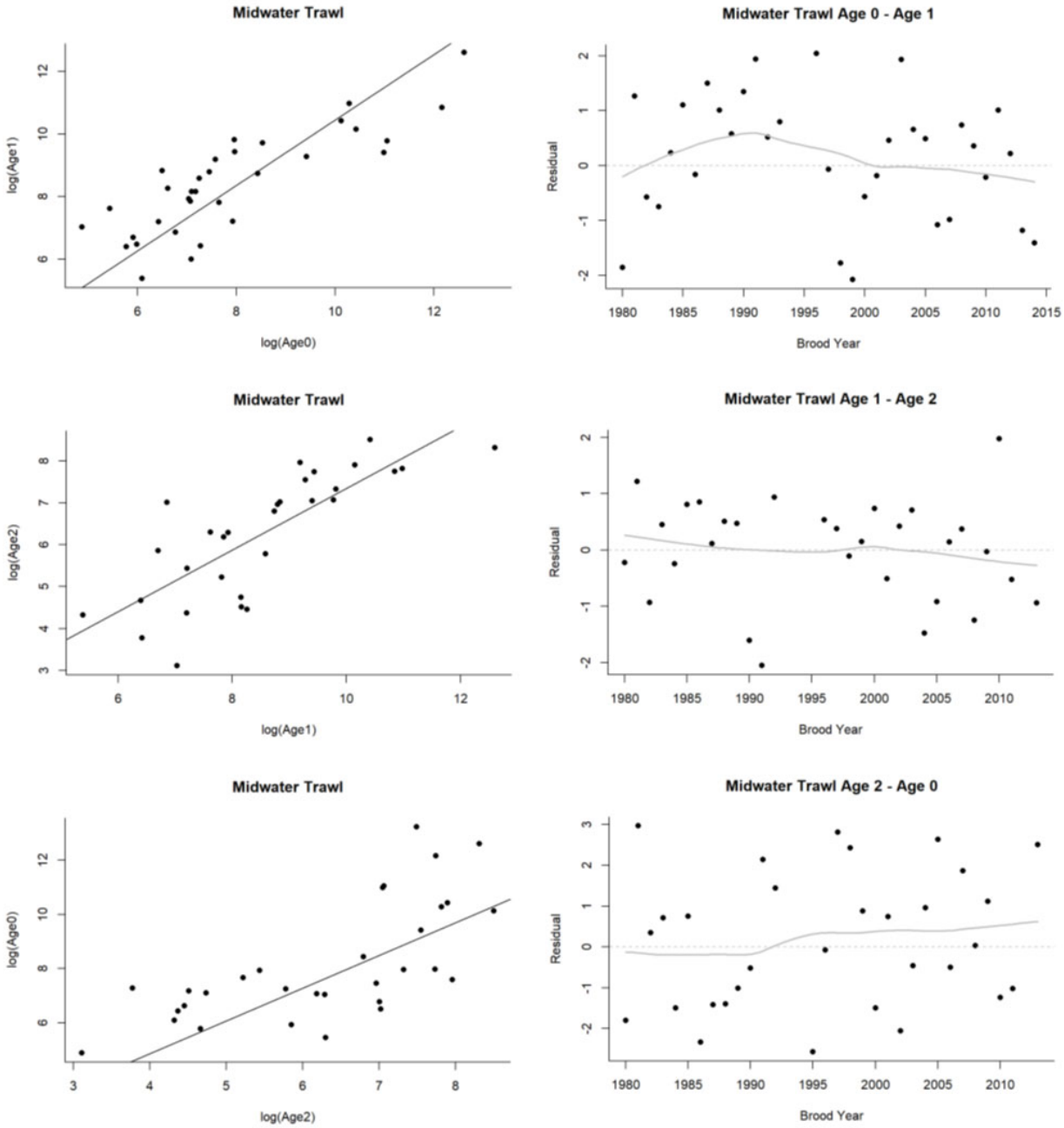


Figure 13: Relationships between midwater trawl age class indices on a log scale provide a visual assessment of vital rates. Left column (top to bottom): Age-0 survival, Age-1 survival, and reproduction. Solid lines represent linear regressions. All plots have slopes with $p < 0.05$. Right column (top to bottom): residuals for the linear regressions in the left column, plotted over time. Solid lines represent a loess smoother.

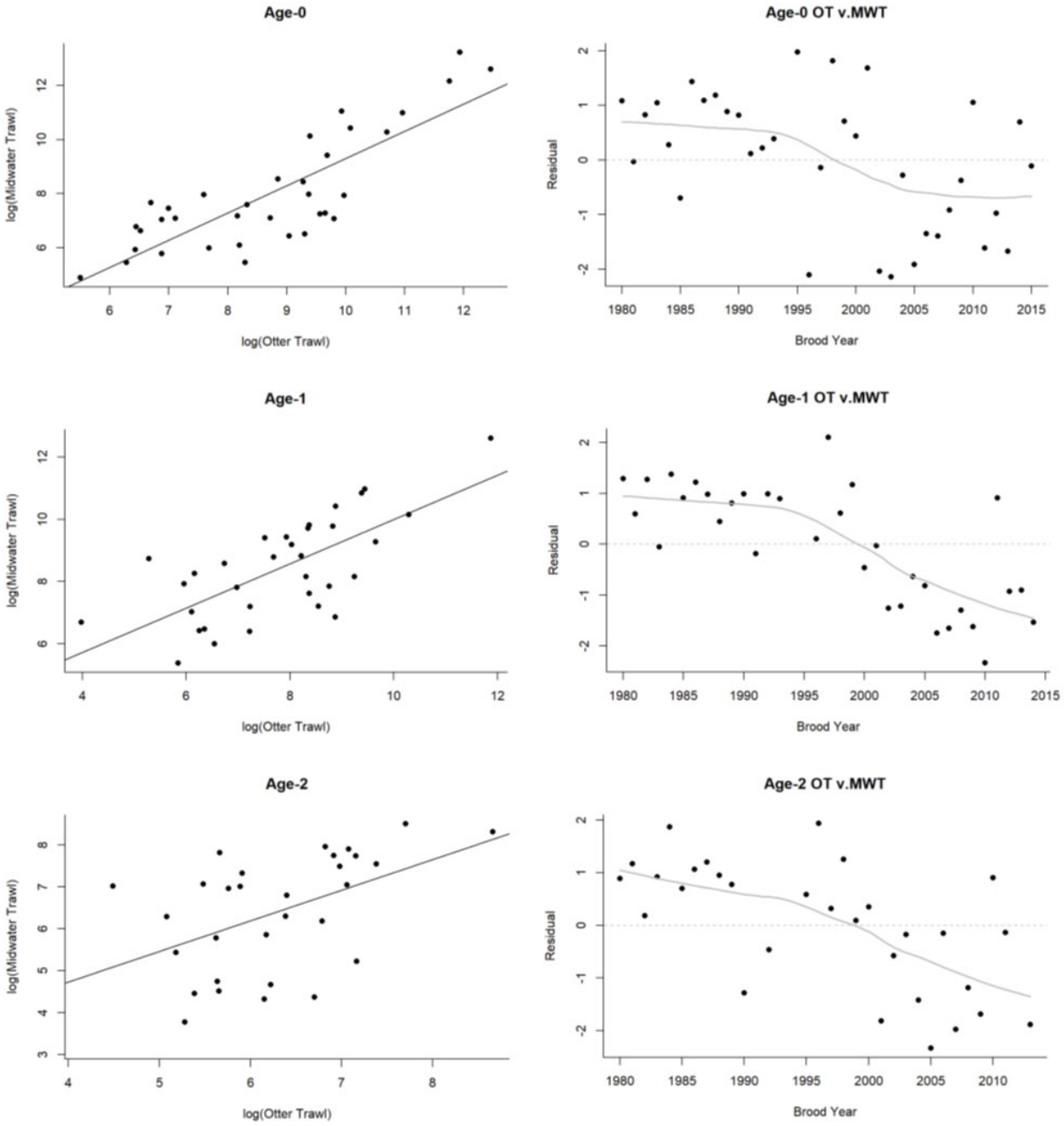


Figure 14: Relationships between abundance indices for the two gear types used by the Bay Study. Left column (top to bottom): data points with a line representing a linear regression between the two gear types for age-0, age-1, and age-2 abundance indices. Slopes for all lines are significantly different than zero ($p < 0.05$). Right column: residuals for the regressions in the left column plotted over time. The solid line represents a loess smoother.

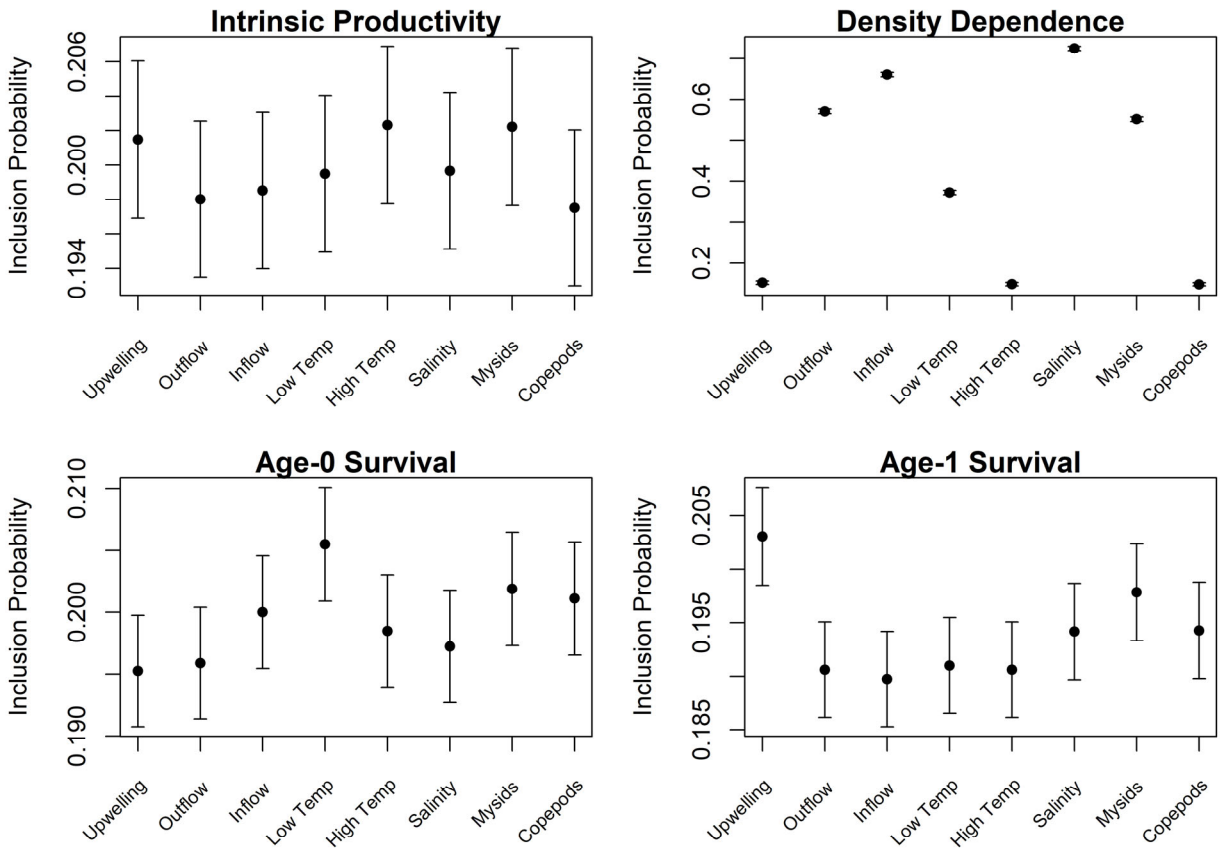


Figure 15: Inclusion probabilities with 95% confidence limits for a set of variables to explain vital patterns in the estimated vital rates. Top row: parameters in the Beverton-Holt equation for reproduction- a is the density independent rate and b estimates density dependence. Bottom row: survival for age-0 and age-1 fish. All environmental variables were standardized. The dashed horizontal line is at 0.2 to indicate a 20% probability of inclusion.

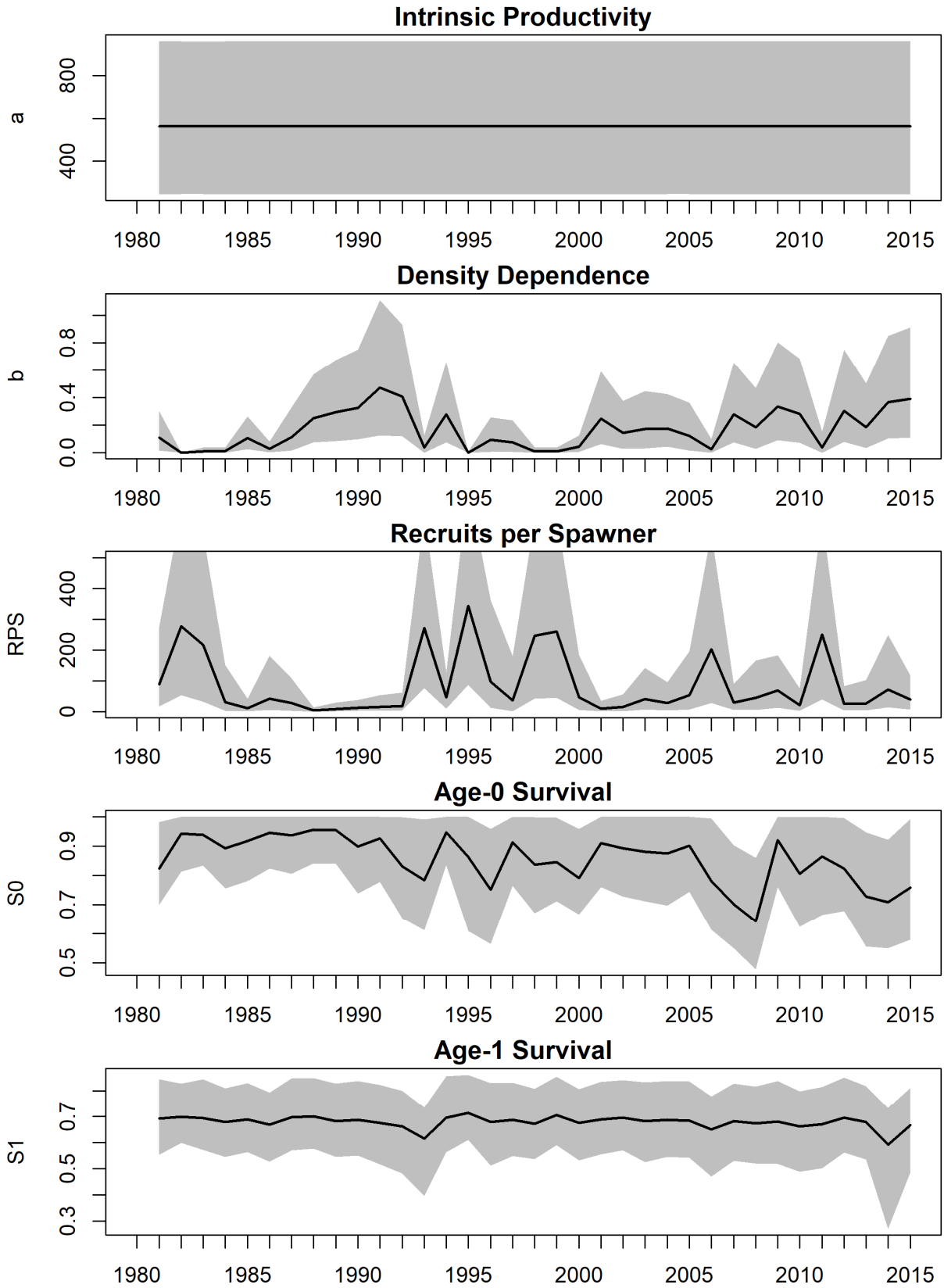


Figure 16: Time series plots for estimated vital rates.

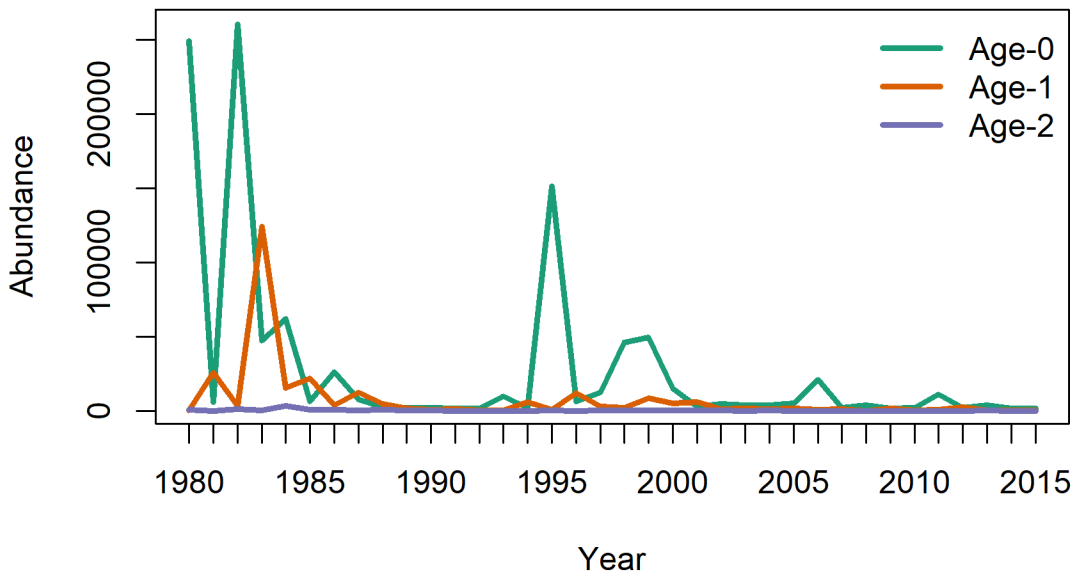


Figure 17: Estimated relative abundance (latent states) for three age classes of Longfin Smelt. Error bars are omitted for readability.

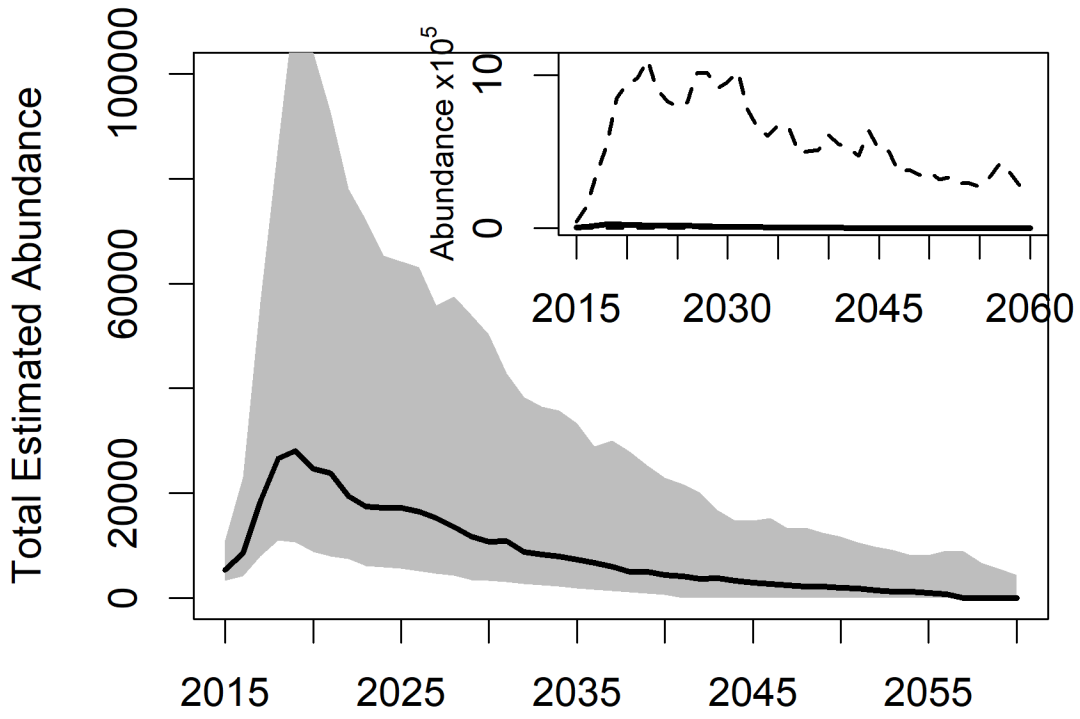


Figure 18: Projected total abundance index values using values randomly sampled from posterior distributions of vital rate parameters from 2005 to 2015, not accounting for the influence of environmental covariates. Black lines in both the main graph and the inset represent the median estimate. The grey envelope in the main plot represents the middle 50% of the estimates around the median. This panel highlights the shape of the estimates over time. The inset shows the median and a dashed line representing the 95% confidence interval. This panel highlights the variability in the estimates.

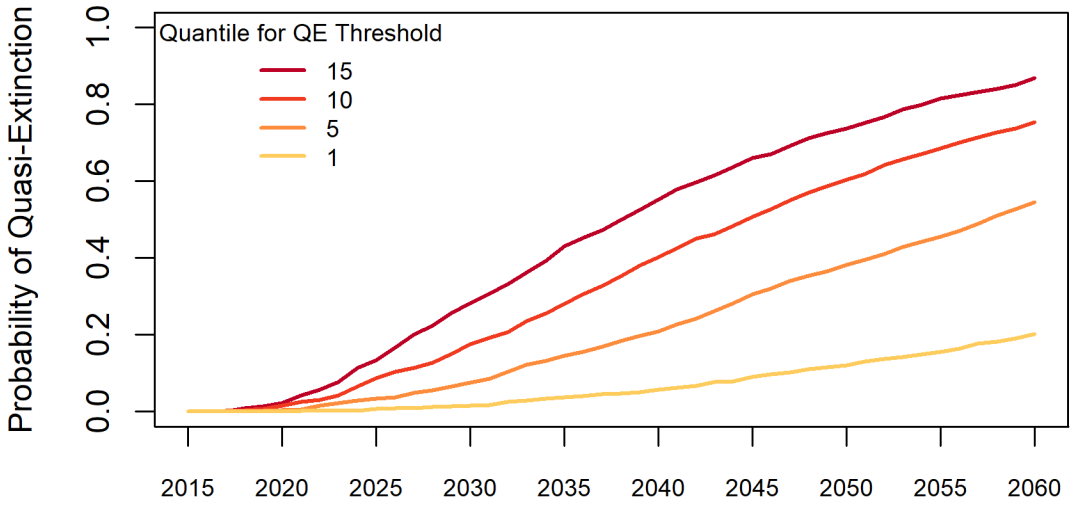


Figure 19: Quasi-extinction probabilities associated with using four different quantiles of population estimates as quasi-extinction values in simulations of abundance index values using values randomly sampled from posterior distributions of vital rate parameters from 2005 to 2015, not accounting for the influence of environmental covariates. Abundance values associated with each quantile are in

Table 8.

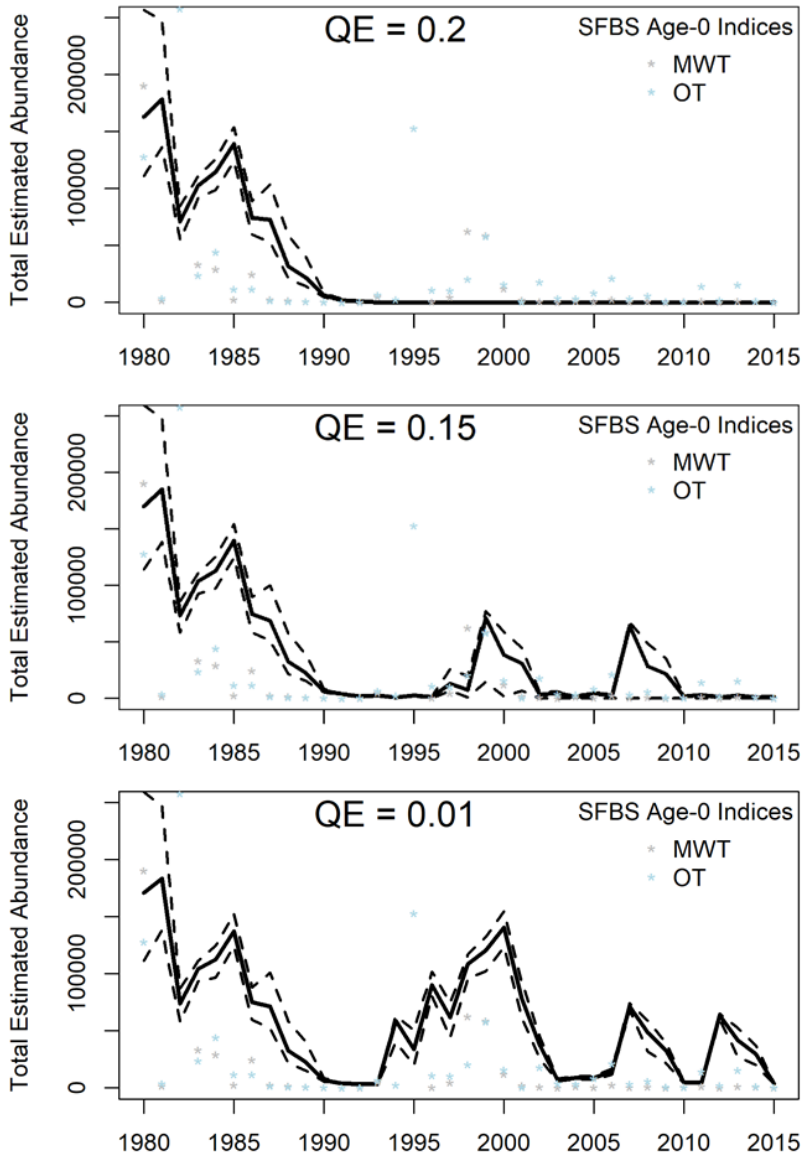
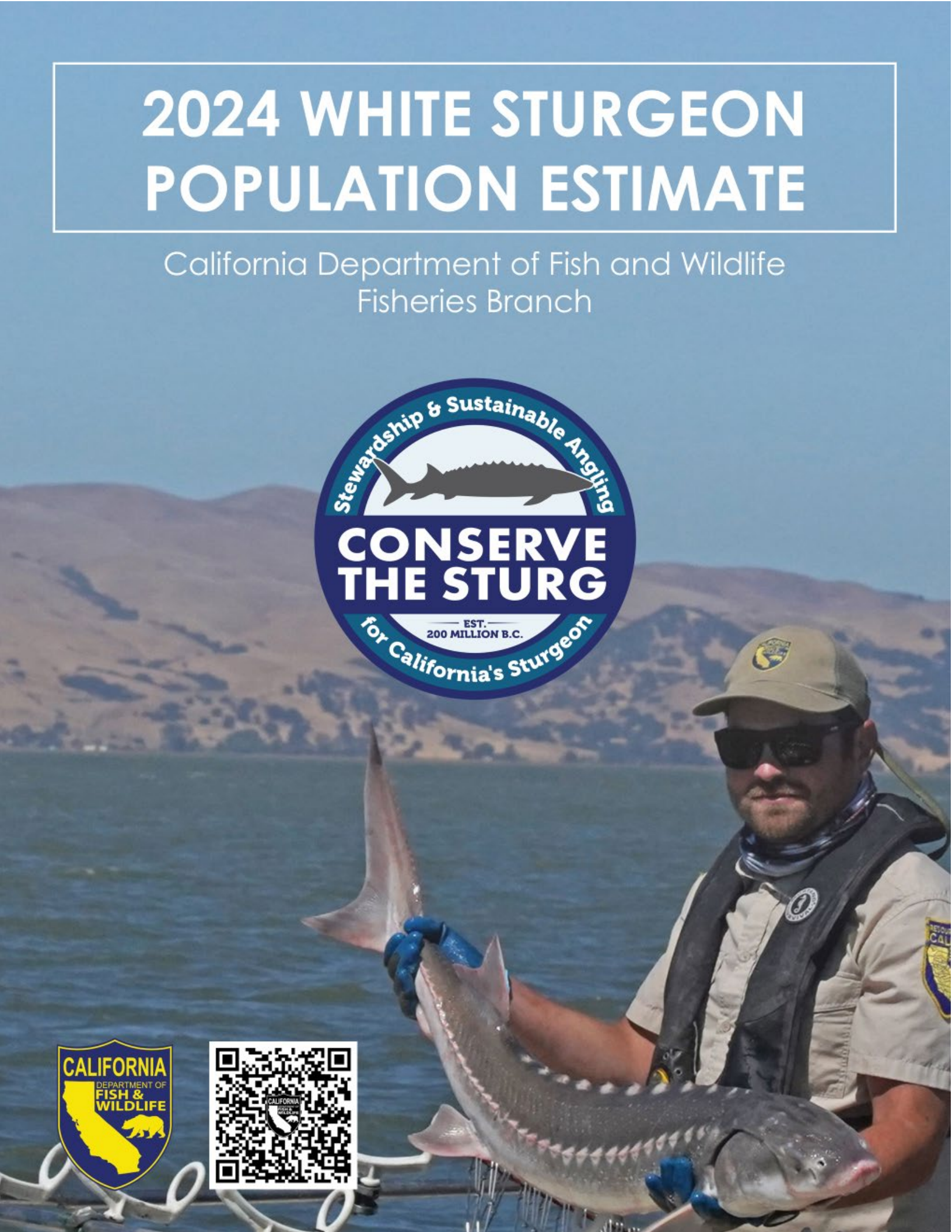


Figure 20: Estimated historical total (all ages) abundance of Longfin Smelt produced by hindcasting with environmental variables for three different abundance values used as quasi-extinction thresholds. The age-specific abundance values associated with the population quantiles are documented in **Table 8**. Values for the age-0 abundance indices produced by the SFBS are also included on the graphs for reference.

2024 WHITE STURGEON POPULATION ESTIMATE

California Department of Fish and Wildlife
Fisheries Branch



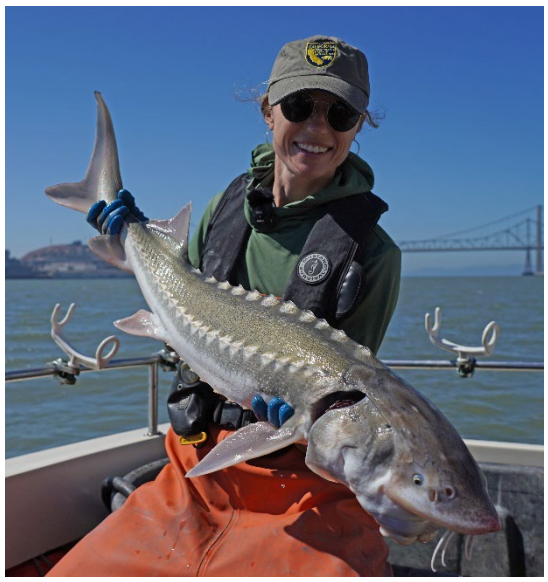


Introduction

In May 2024, CDFW began a new survey program to monitor the white sturgeon population in California. The survey design is based on methods that have been used by Washington and Oregon on the Columbia River since the 1990's, modified to suit the unique environment of the San Francisco Estuary and Delta. A feature of the new survey is that it will provide information across a broad range of fish sizes and ages to help inform management and conservation, whereas the survey methods used in past decades were designed to focus on the size range of sturgeon that could be kept by anglers. The new survey has been peer-reviewed by independent fisheries biologists and sturgeon experts to ensure that the Department is using the most rigorous, scientifically appropriate methods available.

Background

White sturgeon (*Acipenser transmontanus*) are an ancient and iconic fish species native to the San Francisco Estuary and California's Central Valley rivers. If left undisturbed, they can grow larger than 20 feet in length and live for more than 100 years. White sturgeon are an anadromous fish, moving easily between fresh and salt water and spending most of their time in the bays and Delta. Adults sometimes journey into the coastal ocean, but it is not common. Every year, some adults migrate into the largest Central Valley rivers to spawn. Most white sturgeon spawn for the first time after 14–19 years and then spawn again every 2–5 years. Successful spawning and survival of the young requires high springtime freshwater flows.



In June 2024, the California Fish and Game Commission (FGC) responded to a petition to list white sturgeon as threatened under the California Endangered Species Act (CESA). The FGC determined that listing "may be warranted," advancing white sturgeon as a candidate species, and starting the Status Review process. Candidate species receive full CESA protection from harm during the review. All projects that may impact white sturgeon now require CESA permitting and the sport fishery was closed. In August 2024, the FGC supported a CDFW-proposed emergency regulation under Section 2084 in Fish and Game Code Title 14 to allow catch-and-release only. The catch-and-release fishery for white sturgeon opened on October 1, 2024, with new protective seasons, fishing areas, and handling restrictions.



Population Status

The number of white sturgeon has been declining in California since the 1990's. Sturgeon face many challenges, including:

- loss of habitat due to dams
- changes to river flows in the highly altered and managed California water system
- mortality due to toxic Harmful Algal Blooms
- high levels of harvest in the past
- poaching

Estimates from the 1990's and earlier were around 150,000 or more white sturgeon within the range of the legal size to harvest. The exact harvest size limits varied slightly over the years but were most recently 40–60 inches Fork Length (2013–2023). **The most recent 5-year average of this size range of white sturgeon was approximately 30,000 fish**, but these data were collected before a major Harmful Algal Bloom (HAB) in the summer of 2022 that is believed to have killed thousands of sturgeon. Funding for previous studies ended in 2022 so no estimates have been made until 2024.

Historical Monitoring

CDFW started monitoring the white sturgeon population in the 1950's using mark-recapture surveys. Mark-recapture techniques were first developed in 1896 to study fish populations and are a standard fishery tool that is used around the world. In these studies, fish are caught in a marking period, marked with a unique identifier such as a numbered tag, and then released and given time to mix back into the population. Fish are then caught during a recapture period and checked for a mark or tag. The proportion of unmarked fish to marked (recaptured) fish in this second period is used to estimate the size of the overall population.

Until 2022, CDFW caught sturgeon using trammel nets designed to target harvest-sized fish. External reward tags, ranging from $20 to $150 per tag, were placed on fish as "marks" and returned tags were used as "recaptures" in the population models. In 2007, the Sturgeon Report Card program started, providing new data on angler catch and harvest rates that were not available before that time.

There are many different mark-recapture models and fisheries biologists select which models are the best fit for the species and area they are studying. CDFW used the *Lincoln-Peterson method* to estimate the size of the white sturgeon population until 2007 when Sturgeon Report Card data also permitted the use of the *Lincoln Harvest Estimator*. Both methods have been used since that time, but the *Lincoln Harvest Estimator* was considered the best fit for the data.

New White Sturgeon Monitoring

In 2024, CDFW began a new mark-recapture study that does not rely on reward tags and angler participation. To assist with field work and fishing expertise, CDFW contracted with Pacific States Marine Fisheries Commission (PSMFC) and partnered with fishing captains and guides from the white sturgeon sportfishing industry. The new survey is conducted from San Pablo Bay to Rio Vista and the areas in between. These locations were picked based on the Report Card fishing zones with the highest reported catch of white sturgeon. Each zone was then restricted to waters >4 feet deep where the setlines can be safely deployed and divided into 3.5 square mile sampling units (Fig. 1).

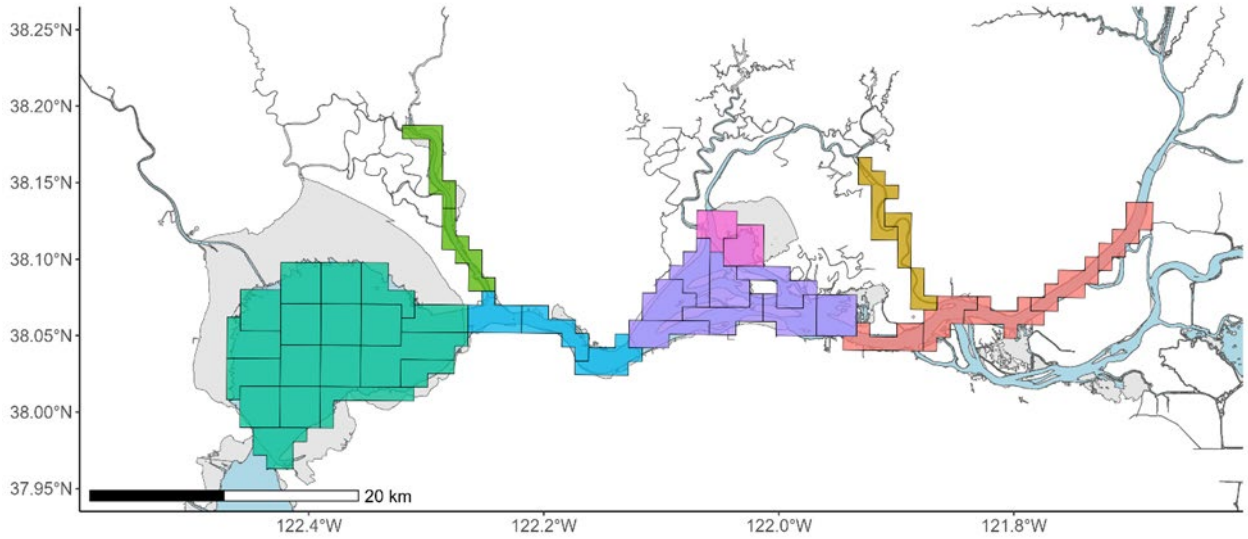


Figure 1. Map of the San Francisco Estuary and Delta indicating the survey region. Different colors indicate Sturgeon Report Card fishing zones (restricted to waters >4 feet depth), black lines show the boundaries of 3.5 square mile sampling units.

White sturgeon are captured using setlines that are anchored in sampling units overnight. Each setline is 600 feet long and has 40 baited circle hooks with three hook sizes to capture a wide size range of sturgeon. Setlines are deployed by two to three boats contracted from the sturgeon fishing guide fleet. The guide captains supply their expert local knowledge of where to find sturgeon while deckhands and CDFW and PSMFC scientists deploy and retrieve gear, measure and tag fish, and record data.



Tiny electronic tags called Passive Integrated Transponders (PIT), like those used to "microchip" pets, are inserted into captured fish as marks. The tags can last for the entire life of a sturgeon and provide a unique code that allows identification of a fish each time it is recaptured and scanned with a handheld electronic wand. Marking is conducted for 32 days in spring and the recapture phase occurs over 32 days in late summer. The survey timing is set when it is least likely to interfere with migration and spawning.

This study currently analyzes the collected mark-recapture data using the *Lincoln-Peterson method* to produce annual population estimates. When multiple years of data are available, the *Cormack-Jolly-Seber (CJS) method* will also be used. CJS will provide additional information on survival, mortality, and emigration and will refine previous population estimates by accounting for survival and mortality.

Preliminary Setline Survey Results from 2024

- Total number of longlines set: **1,009**
- Total number of hooks deployed: **40,243**
- Total number of white sturgeon landed in the first year: **1,449**
    - **961** white sturgeon were caught and marked in the marking period
    - **488** white sturgeon were caught in the recapture period. Of these, **24** were recaptures from this study.
- Mean Fork length: 43 inches (range 10–87 inches)
- Bycatch: primarily sharks and rays in San Pablo Bay and striped bass in inland areas; mortality was low. No federal Endangered Species Act protected green sturgeon or salmon were caught in 2024.

Preliminary White Sturgeon Population Estimate for 2024

Based on the first year of data, we estimate that there are **6,447 white sturgeon between 40–60 inch FL** (95% confidence interval 3,816–11,640) in the California population (Fig. 2). The setline method caught a wider size range of fish than the historical trammel net studies which allows us to estimate the size of a larger portion of the population than in the past. We estimate that there are **18,777 white sturgeon between 10–87 inch FL** (95% confidence interval 12,787–28,664) in the California population.

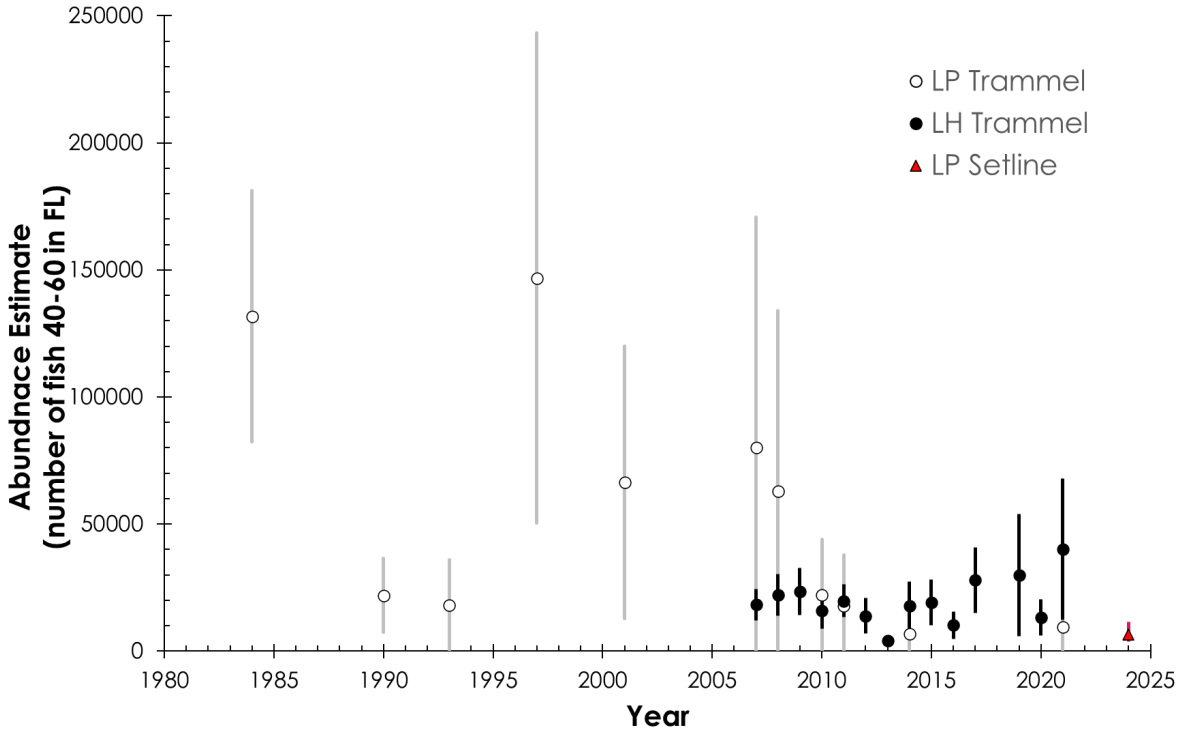


Figure 2. CDFW estimates of the population size of legal-size (40–60 cm FL) white sturgeon. Open circles = trammel net data, Lincoln-Petersen estimator (LP); black circles = trammel net data, Lincoln-Harvest estimator (LH); red triangle = setline data, Lincoln-Petersen estimator. Vertical lines indicate 95% confidence limits. Years with missing data were either not sampled or did not contain sufficient tag returns to generate an estimate.

Next Steps

- Starting this year, CDFW will be conducing sonar-based surveys of spawning white sturgeon in the Sacramento and San Joaquin rivers. This will provide additional information on spawning activity, timing, and location and allow CDFW to monitor this critical event.

- The best way to improve population estimates and reduce the range of possible errors is to tag as many fish as possible during the marking phase of the survey. To achieve this, the Department has added another boat to the marking phase for 2025. We are also working with charter guides to help tag the fish that they catch with clients.



- The numbers presented here are preliminary and are based on the first year of a pilot study. The 2024 season represents the most rigorous survey of white sturgeon conducted in California in many years; however, the survey methods are still being refined. Confidence in the accuracy and reliability of this study will improve as additional years of data are collected.

- A detailed, peer-reviewed description of the methods employed in this study is available on the Department's web page (scan the QR code below) or by request.

- The Department secured funding for three years (2024–2026) to implement and refine this pilot effort. Additional funding will be needed to establish a long-term monitoring program for white sturgeon in California.

Contact Us

This summary report was prepared by staff of CDFW Fisheries Branch. For more information about sturgeon in California, scan the QR code.

Please direct questions to: Sturgeon@wildlife.ca.gov

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November 7, 2025

Matthew Dumlao, Ph.D., Executive Officer
California State Lands Commission
100 Howe Avenue, Suite 100 South
Sacramento, California 95825
Matthew.Dumlao@slc.ca.gov

RE: San Francisco Bay and Delta Sand Mining Lease Applications

Dear Dr. Dumlao:

The Port of San Francisco supports and concurs in the above-referenced lease renewals under review by the California State Lands Commission ("SLC"). Lessees Martin Marietta, Lind Marine, and Suisun Associates currently produce commercial-grade sand from the San Francisco Bay and western Delta on submerged lands and under mineral leases granted primarily by CSLC. The Central Bay leases are within submerged lands at Point Knox and Alcatraz Shoals granted to the City and County of San Francisco by the State of California (e.g., Chapter 1333 of the Statutes of 1968, as amended).

Marine sands are used for commercial and public purposes throughout the greater San Francisco Bay Area, including construction of residential and commercial projects, public infrastructure (e.g., roads, bridges, parks and buildings), shoreline protection, and replenishment of beaches and wetlands. As a local resource, Bay sands help lower public and private construction costs and reduce associated transportation impacts, air emissions, and greenhouse gas emissions from shipping and trucking commercial-grade sand from outside of the Bay Area (including Mexico and Canada). In this manner, marine sands are important to the Bay Area economy and environment. We support SLC's continuing of the lease for submerged lands within the Bay for the continued harvesting of marine sand consistent with the above-referenced applications and as described in the Supplemental Environmental Impact Report ("SEIR") for the Bay and Delta Sand Mining Project.

Please let us know if we can provide any additional information in support of the above-referenced applications.

Sincerely,

DocuSigned by:
Michael Martin
20E5EEC1DEFF477...

Michael Martin
Assistant Port Director
Port of San Francisco

cc: Nicole Dobroski, Chief
Environmental Science, Planning, and Management Division
California State Lands Commission
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Erika Guerra
Environmental Resources Director, Pacific Region
Martin Marietta
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February 8, 2026

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VIA EMAIL

Dr. Matthew Dumlao, Executive Officer
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CEQA.comments@slc.ca.gov
CSLC.CommissionMeetings@slc.ca.gov

Re: Comments on the San Francisco Bay and Delta Sand Mining Project and Final Supplemental EIR (SCH #2007072036)

Dear Mr. Dumlao and Ms. Dobroski:

I am writing on behalf of Martin Marietta Marine Operations, LLC ("Martin Marietta") (successor to Hanson Marine Operations, Inc.), Lind Marine, Inc. ("Lind"), and their joint venture Suisun Associates (collectively, "Lessees"), to provide further context regarding the Supplemental Environmental Impact Report ("SEIR") and San Francisco Bay and Delta Sand Mining Project. Lessees appreciate the California State Lands Commission's ("CSLC's") work in preparing the SEIR and offer the following supplemental responses to public and agency comments provided on the SEIR, including letters received from the Farallon Institute, Marine Mammal Center, and San Francisco Baykeeper, Inc. on February 3 and 4, 2026.

BRIEF SUMMARY

1. ***CEQA Approach*:** The SEIR correctly applies a baseline that includes existing sand mining operations consistent with CEQA's subsequent review requirements and binding appellate precedent upholding the 2012 EIR. Additionally, the SEIR's alternatives analysis is legally sufficient and appropriately limited to the scope of supplemental review.

2. ***Sand Transport and Geomorphology*:** Substantial evidence confirms that bed-elevation changes from sand mining are localized to the immediate vicinity and do not result in broader geomorphic effects—a finding consistent with the Commission's 2012 EIR. The fact that mined sand is largely "relic" does not, itself, establish any significant impact; mined sand continues to comprise a small fraction of sediment within the Bay and

western Delta.

3. ***Biological Resources and Water Quality*:** The SEIR adequately analyzes potential impacts to marine mammals, marine birds, benthic habitats, and water quality, and correctly concludes that continued sand mining will not result in any new or substantially more severe impacts to these species and habitats. Recent studies demonstrate that impacts to water quality and benthic (Bay floor) habitats are highly localized and short-term.

4. ***Fulfillment of Common Law Public Trust Duties*:** The State has fulfilled its public trust obligations through its comprehensive reevaluation of the Project under CEQA and its updated public trust analysis and findings. As previously upheld by the Court of Appeal, substantial evidence supports the conclusion that continued sand mining will not substantially impair public trust uses, resources, or values.

COMMENTS AND RESPONSES

CEQA Approach

1. **The SEIR Uses the Correct Baseline, Supported by Settled Legal Precedent.** Lessees support the Commission's approach of applying a baseline that incorporates existing sand mining operations as originally analyzed in the 2012 Environmental Impact Report ("EIR"). Master Response C provides a well-reasoned analysis supporting the SEIR's baseline comporting with settled case law surrounding the baseline established in the context of subsequent review under Public Resources Code section 21166 and Guidelines 15162.[1] Moreover, the baseline in the 2012 EIR was specifically litigated and upheld by the Court of Appeal in 2015 in response to challenges brought against the Commission's certification of the EIR.[2] The Court's 2015 decision further supports the Commission's continued reliance on the established baseline.

   Despite the appropriate use of an historical baseline, the Commission was extraordinarily conservative in its approach in the SEIR. For example, while it could have relied on the prior baseline and only evaluated the incremental effects of the Revised Project on Bay bathymetry, sand supply, and coastal morphology, the supplemental modeling conducted by CHE (2025) conservatively modeled and evaluated impacts arising from the full

[1] Unlike the "existing conditions" baseline that normally applies to projects undergoing CEQA review in the first instance, the environmental baseline for existing projects previously approved under an EIR are adjusted under CEQA's subsequent review rules: "In effect, 'the baseline for purposes of CEQA is adjusted such that the originally approved project is assumed to exist.'" (Kostka & Zischke, *Practice Under the Cal. Env't Quality Act* (2d ed. CEB 2025), § 12.23 5; *Sierra Club v. City of Orange* (2008) 163 Cal.App.4th 523, 542; *Benton v. Board of Supervisors* (1991) 226 Cal.App.3d 1467, 1477.)
[2] *San Francisco Baykeeper, Inc. v. State Lands Commission* (2015) 242 Cal.App.4th 202, 217–219.

annual and ten-year peak mining volumes against existing conditions.[3] And still the CHE (2025) modeling concluded that there would be no increase in the severity of this previously evaluated impact.

2. **The SEIR's Alternatives Analysis is More Rigorous Than Legally Required.** Lessees support the Commission's Master Response E that addresses comments suggesting the SEIR consider additional alternatives, including alternatives that involve further reductions in mining volumes. Under the California Environmental Quality Act ("CEQA"), a supplemental EIR need not reconsider new alternatives when only minor technical changes are required to make the previous EIR adequate for the project as revised.[4] A supplemental EIR need only contain the information necessary to address the changes; it is not required to repeat the entire analysis.[5] This is particularly true where, as here, there is no new significant impact or substantial increase in the severity of an impact that cannot be mitigated. Again, the Commission employed a conservative approach in summarizing and analyzing the full set of alternatives within its SEIR.

Sand Transport and Geomorphology

3. **Bed-Elevation Changes are Limited to the Immediate Vicinity of Mined Areas.** Lessees concur in the Commission's Master Response F concerning the formation of local bed depressions within mined areas and the expected rates of recovery through natural infill from ongoing sand transport processes.[6] Lessees also emphasize the importance of recent studies evaluating the effects of sand mining on bed elevation.

   As explained in Master Response F, to assess the potential for local depressions to cause substantial effects outside of the mined areas, the SEIR preparers and CHE (2025) observed and modeled predicted erosion and sedimentation outside of the mined areas.[7] CHE (2025) conducted a bathymetric and hydrodynamic study of sand mining operations, concluding that changes in bed elevation caused by sand mining are likely to be limited to the vicinities of the lease areas.[8] This is consistent with the most recent sand studies conducted by the Bay Conservation and Development Commission's Independent Science Panel (ISP) and Sand Studies Technical Advisory Committee (STAC). CHE's conclusion is also consistent with the 2012 EIR, which found that net bottom erosion due to sand mining had largely been contained within the lease and immediately adjacent areas.[9]

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[3] Final SEIR, pp. II-80–II-81, 3.3-8.
[4] *California Coastkeeper Alliance v. State Lands Commission* (2021) 64 Cal.App.5th 36, 66 [supplemental EIR not required to reevaluate alternatives considered rejected in original EIR].
[5] CEQA Guidelines, § 15163(b).
[6] Final SEIR, p. II-13.
[7] Final SEIR, p. II-14.
[8] Draft SEIR, Appendix D, p. 11.
[9] 2012 EIR, p. 4.3-29.

4. **Relic Sand and Limited Replenishment Do Not Alter the SEIR's Impact Conclusions.** Lessees support the Final SEIR's Master Response F, which addresses public comments on relic sand, limited inputs, and disconnected embayments.[10] As Master Response F explains, new scientific studies conducted since the Commission's certification of the 2012 EIR indicate that mined sand is primarily "relic" sand.[11] This is not inconsistent with the studies relied on in 2012 and 2016, which acknowledged that recovery in the mining areas was generally limited.

Moreover, while the recent sand studies have improved the understanding of sand sources and connectivity (or lack thereof) between subembayments, the transport pathways within each subembayment identified in prior research are consistent with transport pathways described in the sand studies and SEIR. As the 2012 EIR and SEIR acknowledge, sand mining removes sand at rates greater than replenishment. The SEIR explains: "As disclosed in the 2012 EIR, sand in the Central Bay is not considered a 'renewable' resource because sand is not thought to be replenished by sand carried by river and tidal currents to the mining lease areas (2012 EIR, p. 4.2.-10, citing CHE 2009)."[12] However, the age or source of the sand is not relevant to, nor does it affect observational and modeling results conducted by CHE, nor does is portend a significant impact in and of itself. Indeed, sand removed during mining is a minute fraction of sediments within particular lease areas, subembayments, and the Bay itself (see accompanying GHD letter).

5. **The Sand Budget Analysis Cannot be Extrapolated to Demonstrate a Significant Impact.** Lessees appreciate the Final SEIR's Master Response F, which addresses public comments about the sand budget analysis conducted by McKee et al. (2024) and reviewed by the ISP (2024), and whether that analysis indicates any more significant impact.[13] It does not. Master Response F appropriately discusses the limitations of the McKee et al. (2024) sand budget analysis. Lessees offer additional context on the sand budget analysis.

The sand budget analysis shows that sand mining removes a certain amount of sand compared to the net flux of other inputs and outputs (e.g., net sand entering or exiting the Bay at the Golden Gate Bridge). However, this should not be interpreted or extrapolated to conclude that sand mining constitutes a significant impact without additional evidence of an actual impact on Bay processes. The sand budget is an accounting approach to contextualize components that add and remove sand at the embayments scale and does not indicate impacts or transport pathways on in-Bay or outer-coast beaches. Nor does it suggest any specific impacts to benthic habitats or other public trust uses or values.

[10] Final SEIR, p. II-16.
[11] *Ibid*.
[12] Final SEIR, p. 3.3-8.
[13] Final SEIR, p. II-18.

Additionally, the sand budget is limited by uncertainties from each component that is included in the budget (e.g., uncertainty in riverine fluxes, bathymetric measurements, etc.).[14] Conclusions drawn from the sand budget are, in turn, subject to the same uncertainties, assumptions, and limited data points. For example, the change in bed storage is one of the largest components used in developing the sand budget, but these estimates are characterized by an uncertainty level of +/- 50% (McKee 2024, Section 4.1.3). Accounting for this uncertainty can result in complete reversals in sediment fluxes between certain embayments—e.g., at the Golden Gate Bridge.[15] Indeed, the very direction of the flux through the Golden Gate Bridge falls within uncertainty bounds of the sand budget, meaning that the sand budget cannot describe with confidence whether sand is flowing into or out of the Golden Gate Bridge on a net basis.

Further, the sand budget analysis does not account for or much less describe the total sand stored within the Central Bay and Suisun Bay lease areas, which represents vast amounts of sand. Without this data, the sand budget analysis only assesses the "significance" of sand mining relative to *active* sand transport in the conceptual sand budget model. The sand budget does not evaluate mining in relation to the overall sand resources available within the lease areas, including those held in underground sand deposits.

The magnitude of sand potentially present in storage is illustrated by prior work. For example, ADEC (2000) evaluated potential borrow sites for the San Fransico International Airport Development Program. Assuming a dredging depth limit of -90 ft MLLW, Point Knox Shoal was estimated to contain approximately 38.5 Mcy of sand—nearly six times the total volume proposed for mining in the Central Bay. That same evaluation identified other sand deposits with San Francisco Bay that were shown to have over 230 Mcy. Importantly, geophysical surveys demonstrate that the Central Bay shoal sand deposits extend to depths well below -90 ft MLLW, suggesting that storage volumes may exceed the ADEC (2000) estimate.

While the volume of sand being mined may be significant to *active* transport, sand mining is likely insignificant by an order of magnitude when accounting for the volume in storage below the active transport layer. The SEIR examined the sand reservoirs in mined areas compared to embayments as an estimate of the total amount of sand in the Bay. Using Fregoso et al. (2025), order-of-magnitude volumes were estimated that the 10-year lease accounts for 1.7% of the total sand volume if the entire proposed volume was mined in Suisan Bay. Similarly in Central Bay, proposed mined volumes for the 10-year lease make up 1.5% of the total volume in the embayment. Again, that comprises

[14] *See* Letter from Aaron Holloway, Senior Costal Engineer, GHD Inc., *GHD Response to Agency and Public Comments on the Supplemental EIR for the San Francisco Bay and Delta Sand Mining Project* (Feb. 6, 2026), p. 2.
[15] *Ibid*.

only a small fraction of sand occurring within those embayments and an even smaller fraction of sediments within the entire Bay.

6. **Policy Questions Regarding Regional Sand Depletion Are Outside the Scope of CEQA.** Lessees support the Final SEIR's Master Response H regarding Mineral Resources.[16] Lessees offer the following additional context to reinforce why broader Commission policies concerning regional sand depletion fall outside CEQA's required analysis.

   The scope of analysis for a supplemental EIR is limited to only new or changed aspects of a project and its environment to ensure that the original EIR is still adequate in light of the changes.[17] A supplemental EIR is not a vehicle to open policy considerations that lie outside the effects of the proposed project modifications.[18] CEQA does not require an agency to analyze long-term resource-management policies that are not proposed as part of the project. Master Response H appropriately confines the SEIR's Mineral Resources analysis to impacts from the Revised Project, consistent with the statutory scope and purpose of supplemental environmental review under CEQA.

7. **Conservative Modeling Supports the Availability of Sand Resources.** Lessees support the Final SEIR's Response A4-11 to comments on the SEIR's analysis of impacts to mineral resources.[19] Response A4-11 discusses the Sand Resources Availability Analysis (CHE 2025), which used bathymetric analysis to estimate available sand in the proposed lease areas in Central Bay and Suisun Bay compared with the proposed mining volumes under the Revised Project.[20] The CHE (2025) modeling approach assumed the entire 10-year cumulative proposed mining volume is extracted at once rather than gradually over the lease term; thus, it does not account for natural replenishment. Moreover, the CHE Availability Analysis assumed mining would remove the peak volumes authorized under the leases and did not account for the additional volume limits imposed by other regulatory agencies (e.g., BCDC). Nor did the CHE Analysis address the substantial reservoirs of sand that will remain in the Bay below -90 ft MLLW. For example, geophysical surveys conducted for the ADEC (2000) study documented significant sand deposits at depths greater than –90 ft MLLW at Point Knox Shoal. Consequently, the CHE Analysis represents an extraordinarily conservative, worst-case scenario for evaluating sand mining relative to available resources and does not acknowledge the substantial sediments that have and will remain in the Bay.[21]

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[16] Final SEIR, p. II-29.
[17] CEQA Guidelines, § 15163(b).
[18] CEQA Guidelines, § 15126.2(a) ["…the lead agency should normally limit its examination to changes in the existing physical conditions in the affected area..."].
[19] BCDC Comment Letter on Draft SEIR (Sep. 29, 2025), pp. 8–9.
[20] Final SEIR, p. II-81.
[21] *Ibid*.

Consequently, the CHE Sand Resources Availability Analysis grossly underestimates total sediments and available sand resources within San Francisco Bay.

Biological Resources and Water Quality

8. **The 2012 EIR and SEIR Correctly Conclude That the Project Will Not Result in Significant Impacts to Marine Mammals.** Several public comments argue that the SEIR should include additional analysis of changes in marine mammal populations and habitat use within the San Francisco Bay due to climate change and other exogenous factors.[22] The Commission addressed these public comments in Master Response I and correctly noted that, under CEQA, global environmental issues are evaluated only when a project could contribute to a cumulatively significant impact or exacerbate existing conditions.[23] Here, there is no evidence that continued sand mining operations would create any new or more severe significant and unmitigated impacts with respect to marine mammals compared to what was analyzed in the 2012 EIR.[24]

The SEIR acknowledges recent population changes in the San Francisco Bay and appropriately analyzes these minor changes within the scope of the SEIR. The Commission prepared Appendix G, Biological Resources Supplemental Information to analyze the potential of special status species, including those listed after the 2012 EIR was certified, to occur in the project area. Appendix G found a "rare-low" potential for gray whale to occur in the lease areas but recognized there has been an increase in sightings of foraging gray whales in Central Bay near the Golden Gate.[25] Appendix G found a "not expected-low" potential for humpback whales to occur in the Delta, and a "moderate-high" potential for them to occur in the Central Bay.[26] Recently, humpback whales have been observed more frequently feeding in the Central Bay.[27] With respect to harbor porpoises, Appendix G found a "low-moderate" potential for the species to occur in the Delta, and a "high" potential for the species to occur in the Central Bay.[28] Harbor porpoises have recently reoccupied the San Francisco Bay after a 60-year absence and can be found foraging for prey in the Central Bay.[29]

Despite these changes in the population dynamics of marine mammals in the San Francisco Bay, the SEIR accurately concludes that continued sand mining operations will

[22] Baykeeper Comment Letter on Draft SEIR (Sep. 29, 2025), pp. 17–18; Farallon Institute Comment Letter on Draft SEIR (Sep. 29, 2025), p. 5; BCDC Comment Letter on Draft SEIR (Sep. 29, 2025), p. 6; Marin Audubon Comment Letter on Draft SEIR (Sep. 29, 2025), p. 2.
[23] Final SEIR, p. II-30.
[24] *Ibid*.
[25] Draft SEIR, Appendix G, p. G-30.
[26] Draft SEIR, Appendix G, p. G-32.
[27] *Ibid*.
[28] Draft SEIR, Appendix G, p. G-31.
[29] *Ibid*.

not result in new or substantially more severe impacts to marine mammals. The SIER's analysis of Impact BIO-1 found that continued sand mining operations would have only minor direct impacts to the free movement or foraging of marine mammals during extraction activities in the Central Bay and western Delta.[30] The potential disturbance to marine mammals remains localized and short-term.[31] As explained in the SEIR, "the potential effects are expected to be minimal since sand mining in these locations occurs for *only a few hours* during each sand mining event, *physically occupies a small area* of the region they are mining in, and during mining, the *equipment remains predominantly stationary*."[32]

Additionally, the SEIR's analysis of Impact BIO-3 addressed potential impacts of sand mining on foraging habitat for marine mammals.[33] The SEIR, which relies on benthic and water quality studies conducted after certification of the 2012 EIR, concluded that sand mining results in short-term changes in habitat composition and associated marine infauna and epifauna in areas where sand extraction has just occurred. However, these changes "*do not appear to result in any detectable changes in infaunal composition or habitat function*."[34] The SEIR concluded impacts to foraging habitat remain less than significant and there would not be any significant impact or substantial increase in the severity of a significant impact related foraging habitat.[35]

The findings of the SEIR are consistent with the direct observations of Lessees. Over the past two decades of sand mining, tugboat and barge operators have not reported a single incident with any marine mammal. There simply is no direct or indirect impact on these species despite a rise in occurrences.

9. **The 2012 EIR and 2026 SEIR Adequately Analyzed Potential Impacts to Marine Birds.** In its comment letters on the Draft and Final SEIR, the Farallon Institute argues that the SEIR does not adequately consider impacts to marine birds on the Farallon Islands and Alcatraz Island.[36] At base, the Farallon Institute has presented no evidence that contradicts the impact conclusions reached in the SEIR.[37] Further, as documented in the Final SEIR, seabird colonies on Alcatraz Island have experienced substantial

[30] Final SEIR, p. 3.2-21.
[31] *Ibid*.
[32] *Ibid*., emphasis added.
[33] Final SEIR, p. 3.2-24–3.2-26.
[34] *Ibid*., emphasis added.
[35] Final SEIR, p. 3.2-26.
[36] Farallon Institute Comment Letter on Draft SEIR (Sep. 29, 2025), p. 4, Farallon Institute Comment Letter on Final SEIR (Feb. 3, 2026), p. 1.
[37] Many of the studies cited by the Farallon Institute in its comment letter on the Final SEIR are outdated, with a few relying on data from the 1980s and 1990s, and most do not address sand mining at all. None of the sources provide evidence directly attributing any observed changes in seabird communities to sand mining activities.

population growth in recent years.[38] This trend indicates that these species continue to have adequate prey availability and suitable foraging habitat. It also supports the conclusion that sand-mining activities are not causing significant adverse impacts to seabird populations. As discussed in the SEIR, only minor direct impacts to the free movement or foraging birds might occur during sand mining activities.[39]

Again, the findings of the SEIR are consistent with the direct observations of Lessees. Over the past two decades of sand mining, tugboat and barge operators have not reported any interference with bird foraging, and tug and barge operations in Central Bay are usually well beyond the areas near Angel Island or Alcatraz Island where marine birds nest and forage. There simply is no direct or indirect impact on these species despite a rise in populations.

10. <u>Recent Technical Studies Affirm the Commission's 2012 EIR—that the Project Will Not Result in Significant Adverse Impacts on Benthic Habitats and Communities</u>. Several commenters, including the Farallon Institute, argue that the SEIR's analysis of impacts on benthic habitats was inadequate. The SEIR presents information from several benthic studies conducted over the past decade, none of which identify any significant adverse impacts of sand mining on disturbance of benthic habitat and macroinvertebrate species.[40]

Lessees support the Commission's position in Responses A4-8 (BCDC) and O9-15 (Farallon Institute) and agree that BCDC's public comment "incorrectly asserts that the benthic community inhabiting a mining lease never fully recovers because mining occurs repeatedly over the same area of the seafloor."[41] BCDC's assertion is contrary to the recent benthic studies, which were developed and implemented to meet the requirements of state and federal permits. The study was developed and conducted collaboratively with input from all the applicable regulatory and science agencies (USFWS, NMFS, USGS, BCDC, and CDFW) and peer revied by experienced and qualified environmental consultants (ESA and AMS). The study reports are presented and discussed in Appendix E of the Draft SEIR and serve as a base for the updated findings presented in the SEIR regarding benthic impacts. The recent studies represent the best scientific evidence available on the effects of sand mining on benthic habitats following mining events.

As explained in the SEIR, the 2012 EIR found that sand mining only results in short-term changes in habitat composition and associated marine infauna and epifauna in areas

[38] Final SEIR, p. II-33.
[39] Final SEIR, p. 3.2-21.
[40] See, e.g., *Benthic Assessment of Sand Mining in Central San Francisco and Suisun Bays Data Report* (NewFields, 2018b; *Technical Memorandum Benthic Assessment of Sand Mining Supplemental Data Evaluation* (NewFields, 2020); and *Benthic Infauna Data Analysis* (AMS, 2024).
[41] Final SEIR, p. II-74.

where sand mining extraction occurs.[42] These changes did not appear to result in any detectable changes in infaunal composition or habitat function.[43] The more recent studies on benthic habitat support the 2012 EIR's earlier findings. For example, three studies assessing benthic infaunal community changes from aggregate sand mining determined that the lease sites in Central Bay and the western Delta that had been recently mined showed no statistically significant impact to the benthic community from sand mining.[44] Both AMS (2009a) and NewFields (2018b, 2020) indicated that the infaunal communities inhabiting the lease sites were characterized as Type I successional communities and therefore are quick to recover and recolonize following physical disturbance.[45] And the SEIR found that benthic recovery of infaunal communities appeared to occur within two years of mining (AMS 2009a).[46] The SEIR and the more recent benthic studies confirm that the impact of aggregate sand mining on benthic habits are less than significant and there will not be any new significant or substantially more severe impacts.

Additionally, the Biological Assessments for Sand Mining in the San Francisco Bay-Delta Estuary prepared by Lessees' biologist Hanson Environmental (2014 and 2026) analyzed the potential effects of sand mining on benthic disturbance. The Biological Assessments reached the conclusion that sand mining contributes to localized temporary impacts to the benthic macroinvertebrate community inhabiting specific locations within the estuary where sand mining occurs.[47] However, the Biological Assessments found no long-term regional impacts on essential fish habitat and managed or sensitive fish species.[48]

11. <u>Turbidity Plumes Are Short-Lived And Do Not Impact Foraging Habitats</u>. Audubon California asserts that the Draft SEIR did not adequately address potential effects of turbidity plumes on eelgrass and seabird foraging habitat during sand mining operations.[49] Farallon Institute, in its comment letter on the Final SEIR, made a similar argument that insufficient attention has been given to potential impacts on herring and their eelgrass breeding habitat in Richardson Bay.[50] Lessees support the Commission's discussion in Response O5-3 and Master Response J and reiterate that the NewFields water quality studies concluded that sediment plumes and turbidity from sand mining are short-lived and only result in minor short-term, or negligible changes in the water column

[42] Final SEIR, p. 3.2-24–26.
[43] *Ibid*.
[44] *Ibid*.
[45] *Ibid*.
[46] *Ibid*.
[47] *Biological Assessment and Essential Fish Habitat Assessment for Sand Mining in the San Francisco Bay-Delta Estuary*, prepared by Hanson Environmental (Jan. 2026), p. 111 [shared with CSLC January 2026].
[48] *Ibid*.
[49] Audubon California Comment Letter on Draft SEIR (Sept. 22, 2025), p. 2.
[50] Farallon Institute Comment Letter on Final SEIR (Feb. 3, 2026), p. 2.

(NewFields 2015, 2018).[51]

At a recent meeting of BCDC's Sand Studies Commissioner Working Group, Commissioners inquired about potential impacts to eelgrass beds in Richardson Bay in relation to the 2018 NewFields Water Quality Monitoring Report. NewFields subsequently prepared a written response explaining that impacts to eelgrass beds in Richardson Bay are not expected to occur. Richardson Bay lies several thousand meters beyond the aerial extent of sediment plumes generated from sand mining operations in Central Bay.[52] NewFields has detected no adverse effects on water quality within the Bay near active mining sites. Consequently, physical impacts to eelgrass or other foraging habitats in Richardson Bay will not occur as a result of sand mining operations.

Fulfillment of Common Law Public Trust Duties

12. **The State Has Fulfilled its Public Trust Duty.** Baykeeper, in its submittal dated February 4, 2026[53], contends that the State has not adequately reconsidered the public trust in light of information gleaned since the 2012 EIR.[54] This same argument was rejected by the Court of Appeal in Baykeeper's prior lawsuit. Further, Baykeeper neglects to acknowledge that, in considering the public trust (i) the State has the discretion (and indeed the duty) to consider and balance competing public interests,[55] and (ii) there is no set "procedural matrix" for doing so.[56] The State's extensive reevaluation of the Project under CEQA, its updated public trust analysis in the February 2026 staff report, and new proposed findings more than satisfy the State's public trust duty.

In the 2018 Baykeeper appeal, as here, Baykeeper contended that the State violated its continuing duty to protect public trust assets by re-approving the Central Bay leases in 2016 despite "new" scientific research establishing a causal link between sand mining and coastal erosion. As the Court of Appeal recognized, "the SLC Staff report concluded that [Lessees'] sand mining activities *would not impair public trust uses* by either substantially depleting the sand resource or substantially interfering with sand transport

[51] Final SEIR, p. II-98;
[52] ERM NewFields, *Response to Questions from BCDC Commissioners Workshop* (Dec. 8, 2025), p. 2 [shared with CSLC Jan. 16, 2026].
[53] Baykeeper Comment Letter on Final SEIR (Feb. 4, 2026), pp. 12–15.
[54] Baykeeper's statement that sand mining is somehow tied to investments in Ocean Beach protection projects is pure speculation and hyperbole. As prior studies have noted, northern Ocean Beach is accreting and southern Ocean Beach is eroding due to natural coastal processes unrelated to sand mining.
[55] *State Water Resources Control Bd. Cases* (2006) 136 Cal.App.4th 674, 777–778 [it is within trustee agency's "discretion and judgment" to balance all "competing interests"]; *Monterey Coastkeeper v. Central Coast Regional Water Quality Control Bd.* (2022) 76 Cal.App.5th 1, 21 [the doctrine is "inherently discretionary" and does not impose the duty to exercise authorities in any particular way.].
[56] *Monterey Coastkeeper*, *supra*, 76 Cal.App.5th at 21; *Citizens for East Shore Parks v. Cal. State Lands Com.* (2011) 202 Cal.App.4th 549, 576–577.

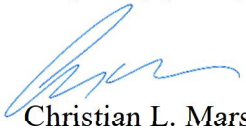
> and coastal morphology at the San Francisco bar and Ocean Beach."[57] The Court of Appeal found that these conclusions were "supported by substantial evidence"—"a trilogy of CHE studies that were discussed in the SLC Staff report."[58] Ultimately, the Court held that "substantial evidence support[ed] the SLC's findings that the project *will not impair public trust uses or values*."[59]

In line with the Court of Appeal's 2018 ruling, the State's public trust analysis set forth in its February 2026 staff report appropriately finds that continued sand mining will not substantially impair public trust uses or resources. In support of this finding, the State has diligently reevaluated a host of issues, including the comments from Baykeeper and other agencies, and conducted its own peer review of the recent science pertaining to biological resources, sediment transport, and Bay and coastal morphology. The fact that Baykeeper or others might disagree with the State's public trust determinations does not take away from the substantial process and discretion the State exercised in reevaluating these issues.

CONCLUSION

The Lessees appreciate the opportunity to provide comments on the Project and SEIR. We respectfully request that this letter be included in the administrative record and that the Commission approve the subject leases for another 10-year term.

Very truly yours,



Christian L. Marsh

Enclosures:
1) Letter from Aaron Holloway, Senior Costal Engineer, GHD Inc., *GHD Response to Agency and Public Comments on the Supplemental EIR for the San Francisco Bay and Delta Sand Mining Project* (Feb. 6, 2026).
2) Letter from Charles Hanson, Senior Fishery Biologist, Hanson Environmental, *Responses to Public Comments on the Supplemental Environmental Impact Report for the San Francisco Bay and Delta Sand Mining Project* (Feb. 8, 2026).

cc: Joseph Fabel, Senior Attorney, Legal Division, California State Lands Commission
Erika Guerra, Environmental Resources Director – Pacific Region, Martin Marietta
John Gillan, Associate General Counsel & Head of Sustainability, Martin Marietta
Bill Butler, Vice President – Operations Planning & ESG Programs, Lind Marine

[57] *San Francisco Baykeeper, Inc. v. State Lands Com.* (2018) 29 Cal.App.5th 562, 580–582.
[58] *Ibid.*
[59] *Ibid.*

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GHD Response to Agency and Public Comments on the Supplemental EIR for the San Francisco Bay and Delta Sand Mining Project (SCH #2007072036)

Dear Ms. Guerra and Mr. Butler:

At your request we have peer reviewed select technical comments provided by agencies and non-governmental organizations on the Supplemental Environmental Impact Report ("SEIR") and the San Francisco Bay and Delta Sand Mining Project. The Final SEIR, released on January 16, 2026, provides detailed responses to comments. GHD has also reviewed the most recent comments submitted by non-governmental organizations on February 4, 2026 related to sand transport, sediment supply, and resource availability. We continue to concur with the SEIR's analysis and conclusions and present these supplemental responses to provide further context regarding the potential effects of sand mining activities on sand transport, supply, and Bay and coastal geomorphology.

SUMMARY OF COMMENTS ON THE SEIR

Below is a summary of points raised in public comments that GHD addresses in this response:

- Public agency comments recommend that the DSEIR reference takeaways from the Stantec (2024) report; however, the Stantec (2024) report does not present any new data or analyses. Further, many of the key takeaways from the Stantec (2024) report are consistent with statements made in the SEIR, which draw upon the most recent the San Francisco Bay Sand Mining Studies ("Sand Science Studies") and reinforce conclusions of the prior Environmental Impact Report prepared in 2012 ("2012 EIR") and supporting technical studies.
- Several public agency comments imply that sand mining could potentially impact outer coast beach areas despite the fact that 1) no pathway has been established connecting mining areas to specific beaches based on multiple independent research studies, 2) beaches closest to the bay entrance are not eroding, and 3) independent researchers have stated the above in public comments.
- Public agency comments often highlight recent science that challenges the previous understanding of bay sediment dynamics. The updated science further informs sand sources and connectivity (or lack thereof) within the Bay, but it does not change the findings related to sand transport at the subembayment scale. The updated analysis and results from the Sand Studies are discussed in the SEIR, showing that prior and recent studies are in agreement that there is no evidence of impact outside of the leased mining areas.
- Public agencies continually describe the impacts of mining as "unknown" or "undefined". The fact that the SEIR and prior studies have failed to identify measurable, demonstrable, and significant impacts on sediment transport and supply beyond the discrete lease areas is itself evidence that mining during a 10-year period will likely not have a measurable or demonstrative

impact on sediment transport or supply outside the lease areas. Characterizing theoretical impacts as “unknown” or “undefined” omits decades of research showing negligible impact.

Detailed responses to paraphrased public agency and non-governmental organization comments are provided below.

Sand Transport, Provenance, and Budget Studies

Re: Comments on lack of information within SEIR that is provided by the Stantec 2024 report, and that the volume of mined sand is significant relative to the SF Bay sand budget.

Response:

Public agency comments imply that this Sand Studies Technical Advisory Committee (“STAC”)/Independent Science Panel (“ISP”) process was the first to integrate scientific understanding of the San Francisco Bay (“SF Bay”) sediment system. However, the body of research and technical analyses prepared for the 2012 EIR, permits, and subsequent litigation was focused on these same issues. Importantly, the prior conclusions regarding less than significant impacts from mining activities are reinforced by findings from the Sand Science Studies as documented throughout the SEIR.

The Stantec (2024) report does not provide any new research or analysis beyond the Sand Science Studies. This report simply summarizes discussions and deliberations among ISP members, which involved discussion around the management questions considering the findings, uncertainties and limitations of the research.

The Stantec (2024) report makes a commendable effort to synthesize the complex Sand Science Studies; however, important inconsistencies remain among the underlying studies, ISP discussions, and the Stantec (2024) report itself. The report was never intended to fully evaluate sand mining impacts and, in several instances, the report goes beyond the available science by speculating about uncertain or unquantified effects in the absence of such data. More details about these issues are provided in Appendix H of the Stantec (2024) report.

Also, the Stantec (2024) report was not updated to account for the revised Sand Budget released in 2025 and presented to the San Francisco Bay Conservation and Development Commission’s (“BCDC”) Sand Studies Commissioner Working Group. The revised Sand Budget corrected an error that was discovered in the USGS bathymetric data and revised conclusions about net sand flux at the Golden Gate. The speculation and deliberation among ISP members were based on an obsolete version of the Sand Budget (2023) and therefore no longer reflects the best available science on this topic. The updated Sand Budget has not been made publicly available at this time.

As the Stantec (2024) report does not offer any new information not covered in the Sand Science Studies, and the Stantec (2024) report does not necessarily reflect the most updated science related to the Sand Budget, the report itself has limitations in directly evaluating or addressing impacts from mining in the context of the SEIR.

The Stantec (2024) report statements that sand is a non-renewable resource are consistent with the 2012 EIR as explained in the SEIR: “As disclosed in the 2012 EIR, sand in the Central Bay is not considered a ‘renewable’ resource because sand is not thought to be replenished by sand carried by river and tidal currents to the mining lease areas (2012 EIR, p. 4.2.-10, citing CHE 2009).” (2025 SEIR, p. 3.3-8).

Both the 2012 EIR and SEIR recognize that sand mining is an extractive practice that does not assume replenishment as a substantial source. While sand budget investigations show that sand mining removes a relatively large amount of sand compared to other fluxes, this should not be interpreted or extrapolated to conclude that sand mining constitutes a significant impact without evidence of this impact. The Sand Budget is an accounting approach to contextualize components that add and remove sand at the embayments scale and does not indicate impacts or transport pathways due to mining beyond capturing the relative scale of flux magnitudes. Additionally, the Sand Budget is limited given large uncertainties from each component that is included (e.g., uncertainty in riverine fluxes, bathymetric measurements, etc.). Thus, conclusions drawn from the Sand Budget are subject to the uncertainties, assumptions, and limited data points.

For example, the change in bed storage is one of the largest components used in developing the sand budget, but these estimates are characterized by +-50% uncertainty (McKee 2024, Section 4.1.3) due to measurement methods. Accounting for this uncertainty can result in complete reversals in sediment fluxes between certain embayments. Indeed, the very direction of the flux through the Golden Gate Bridge falls within uncertainty bounds of the sand budget, meaning the sand budget cannot describe with confidence whether sand is flowing into or out of the Golden Gate Bridge.

Additionally, McKee presented an updated Sand Budget to BCDC's Sand Studies Working Group that made corrections to the 2024 Sand Budget report. The revised Sand Budget (2025) concluded that "net sand flux to the ocean is definitely indistinguishably from zero" (McKee presentation to BCDC Working Group, March 18, 2025). So, the statement "net sand discharge to the ocean" insinuates there is a net discharge of sand through the Golden Gate which is based on an obsolete version of the Sand Budget (2023) and disregards the uncertainty inherent in these estimates and assumptions.

The SF Bay Sand Budget also fails to account for or describe the total sand stored within the Central Bay and Suisun Bay lease areas, which represents vast amounts of sand. Without this data, the Stantec (2024) report's Sand Budget analysis only assesses the "significance" of sand mining relative to active sand transport in the conceptual sand budget model. Since sand mining removes mostly relic sand stored in the SF Bay, the Stantec (2024) report lacks the information needed to make any conclusions regarding sand mining's impact on the overall sand resource availability. The Stantec (2024) report admits to this deficiency, stating that the size of "the pool of relic sand," and therefore any "significance of [the] reduction" in relic sand caused by sand mining, "is unknown." (Stantec 2024, Section 3.2.2.)

As an example of the sand available in storage, the ADEC (2000) report investigated potential borrow sites for the San Fransico International Airport Development Program. Under the assumption that -90 ft MLLW was the dredge depth limit, the Point Knox Shoal was shown to have 38.5 Mcy of sand and is nearly 6 times the total volume of proposed mining for Central Bay. Furthermore, other sand deposits within SF Bay were shown to have over 230 Mcy. Importantly, the sand deposits of the Central Bay shoals extend to depths well below -90 feet, MLLW as shown through geophysical surveying performed along the Point Knox Shoal. Based on these cross sections the volume of sand within this shoal could be 2-3 times greater than the volume above -90 feet, MLLW (i.e., ~100 Mcy within Point Knox Shoal).

While the volume of sand being mined may be relatively significant when compared to active transport, comparing magnitudes to these two shoals highlights that sand mining is likely insignificant by an order of magnitude when accounting for the volume in storage below the active transport layer.

Additionally, the Final SEIR examined the sand reservoirs in mined areas compared to embayments as an estimate of the total amount of sand in the SF Bay. Using Fregoso et al. (2025) order-of-magnitude volumes were estimated that the 10-year lease accounts for 1.7% of the total sand volume if the entire proposed volume was mined in Suisan Bay. Similarly in the Central Bay, proposed mined volumes for the 10-year lease make up 1.5% of the total volume in the embayment.

<u>Re: Suisun bay sand is not replenished, it is a finite resource, and depressions persist.</u>

Response:

The creation of depressions due to sand mining is part of the Project as described in the 2012 EIR and SEIR (Section 2 EIR and SEIR), and the conclusion that sand mining at existing lease areas has a measurable impact on the bathymetry and storage of sand within lease areas is not new (2025 SEIR, pp. 3.4-4, 3.4-6). This conclusion is consistent with the 2012 EIR takeaways and reinforced by the Sand Science Studies detailed in the SEIR (e.g., eTrac, Inc reports 2018a; 2018b; 2023 and Deltares reports 2023b; 2023c).

It should be noted that while mining removes a measurable volume of sand and from a particular lease area, that does not mean that mining is having a demonstrable impact beyond the lease areas. This is shown by several studies referenced in the SEIR, where observations and modeling efforts found bed elevation changes to be limited to the immediate vicinity of mined areas.

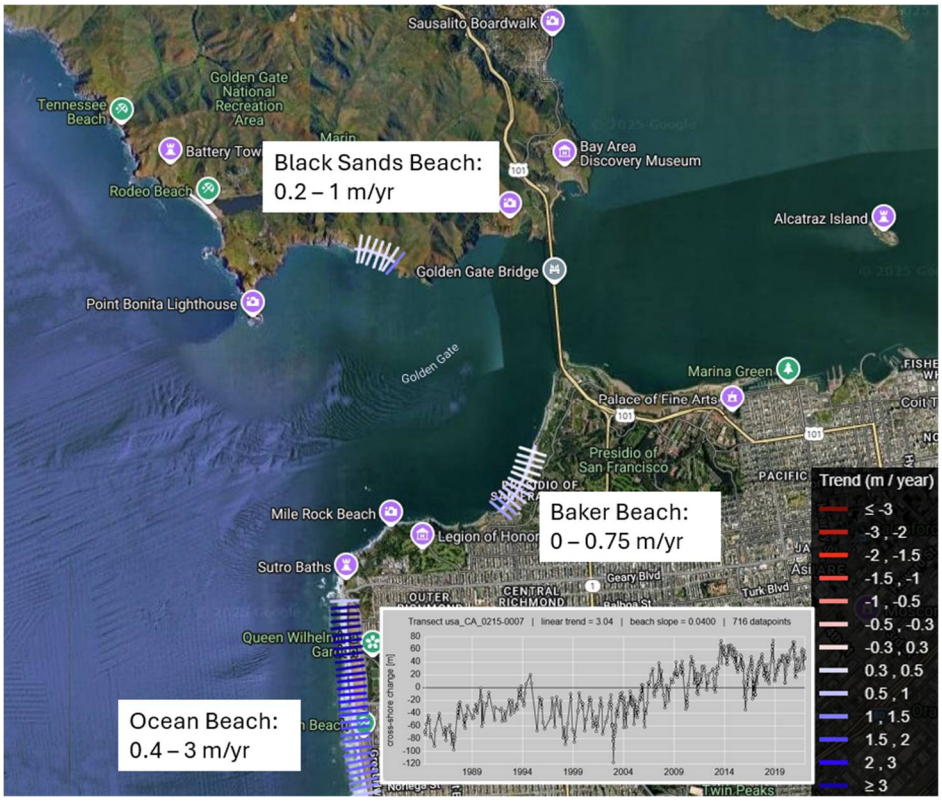
Re: Updated science on bay sand origin, source, and embayment connectivity and effects of mining on beaches.

Response:

The recent Sand Science Studies have improved the understanding of sand sources and connectivity (or lack thereof) between subembayments. However, within each subembayment many of the sand transport pathways evaluated in prior research and analyses (2012 EIR & permitting process) are consistent with transport pathways described in recent Sand Science Studies and the SEIR. As the EIR and SEIR acknowledge, mining removes sand at rates greater than replenishes; however, the age or source of the sand being relic is not relevant to, nor does it affect, observational and modeling results.

The statement about “unquantified effects” fails to acknowledge the effort of numerous research teams, using a variety of analyses and modeling tools, that have been unable to identify any direct link between sand mining activities and effects to beaches in-bay or on the outer coast. Even assuming worst case scenarios (17–20 million cubic yards all in one year), CHE modeling a decade ago, and more recently as part of the SEIR (2025) show no measurable impact on coastal beaches. The study by Anchor QEA (2023) similarly found negligible impacts to coastal beaches (SEIR 2025, p. 3.4-16). Ring analysis based on bed change observations also failed to find detectable change 100 meters outside of mined areas apart from one instance and for one ring, otherwise mining effects were not observed beyond 100 meters (Deltares 2023b,c). These modeling and observational studies reach the same conclusion, that there is no sand transport pathway between mined areas and outer coast beaches.

Additionally, observations from Crissy Field Beach (Barnard et al. 2012) and North Ocean Beach show beaches closest to mining areas are accreting, not eroding. Below is a figure of beach change based on CoastSAT at Black Sands, Baker, and North Ocean Beach—the closest beaches to the entrance of the San Francisco Bay, all of which are stable or growing up to 3 m/yr (+12 ft/yr). Managers of Crissy Fields stated in a BCDC workshop that at “[Crissy Field] … the NPS has had problems with sand accretion near the Golden Gate Bridge” and that “the inlet at Crissy Field does not function naturally, filling with sediment….” (BCDC 2015-The Science of Sediment: Identifying Bay Sediment Science Priorities. Workshop Summary Report). Overall, there is no evidence of erosion along beaches closest to the Bay entrance that would indicate a problem caused by Bay sand mining activities.



The Stantec (2024) report confirms that sand mining impacts outside the active lease area are extremely difficult to quantify or assess. The Stantec (2024) report moves beyond the science and errs when it speculates about theoretical impacts to resources outside the lease areas (including beaches) in the absence of such data. Accordingly, ISP members at the 2023 ISP meetings readily acknowledged that 1) there is no evidence or expectation that sand mining would impact in-Bay beaches, and 2) the Sand Science Studies do not point to any conclusions about long-term effects related to effects on in-bay or outer coast beaches. Further, speculation about impacts of mining on the outer coast was largely based on an obsolete version of the Sand Budget (2023) and extrapolated based on fluxes through the Golden Gate Bridge to assume there is a direct pathway between mined areas and coastal beaches. The revised Sand Budget (2025) concluded that "net sand flux to the ocean is definitely indistinguishably from zero" (McKee presentation to BCDC Working Group, March 18, 2025), indicating less to no connectivity between mined areas coastal beaches. Unfortunately, due to timing, the updated science was not incorporated into the Stantec (2024) report.

Re: San Fransico Bay and the Pacific Ocean share a common pool of sand.

Response:

Prior modeling efforts (2012 EIR, AQEA, and SEIR) all evaluated the impact of mining on this shared pool and concluded that the impact was insignificant. Therefore, the effect of mining on this shared pool has been repeatedly shown to have no demonstrative impact and should not be characterized as "unknown" (e.g., CHE 2025).

Additionally, when comparing mining volumes to the "shared pool" of sand between the Pacific Ocean, the magnitude of that potential "shared pool" should be understood. The San Francisco Airport Expansion Project in the early 2000's (ADEC 2000) investigated available sand within portions of this "shared pool" and concluded that nearly 120 Mcy of sand was available within the SF Bar channel dredge prism and Central Bay lease areas at depths shallower than -90 feet, MLLW. This volume is a fraction of the total amount of sand within the SF Bar (outside ship channel) and Central Bay shoals and at depths greater than -90 feet, MLLW. This investigation of sand resource availability highlights the vast amount of sand volume that is necessary to contextualize when discussing the size of the "shared pool" of sand.

Re: Fregoso et al. study reporting of bay floor change and that greatest rate of bathymetric change were associated with dredging and sand mining.

Response:

The USGS study by Fregoso et al. (2025) reinforces prior findings that sand mining has not produced measurable bathymetric impacts beyond active lease areas. Their analysis of Central Bay bathymetry shows a net accretion of approximately 6.4 million cubic meters between the 1980s and the 2010s in areas outside of mined sites. This result provides further evidence that observed seafloor lowering associated with mining activities is largely confined to the lease areas, with no detectable, system-wide erosion beyond those boundaries.

The SEIR cites McKee et al. (2023) to state that sand mining was the largest loss of sand from the Bay, whose work was influenced by the bathymetric change work by Freqoso et al. (2024). However, this report was revised in late 2024 to correct an error in bathymetric change volumes in Central Bay. The conclusions related to bathymetric change in mining and dredging areas are well known and were evaluated thoroughly in the prior EIR, Sand Science Studies, and SEIR.

Re: SFEI Sand Budget Study (2023) findings that sand mining activities are the largest term in reducing sand.

Response:

The SEIR directly acknowledges this point when citing cites McKee et al (2023): "that sand mining was the largest loss of sand from the Bay" (SEIR 2025, p.3.4-9). Therefore, the SEIR does not fail to acknowledge this point and does not dismiss this information.

As discussed above and in comments provided in Appendix H of the Stantec (2024) report, the sand budget analysis only assesses the “significance” of sand mining relative to active sand fluxes in the conceptual sand budget model.

The SEIR also discusses several studies that focus on the cumulative change, which are all in agreement that sand mining has no detectable impact outside of mining areas: e.g., Deltares 2023b,c where ring analysis showed no impacts observed beyond 100 meters from mining areas; Deltares 2023 where sand mining does not affect bedform dynamics; Anchor QEA 2023 showing negligible change in Golden Gate and coastal areas; CHE 2025 showing no significant impact on sediment transport outside of lease areas. Each of these studies considers cumulative impacts of mining using different approaches, with each study agreeing that there is no evidence of sand mining having impacts outside of mining areas.

Re: Cumulative impact of sanding mining.

Response:

Observations and numerical modeling performed for the prior EIR, Sand Science Studies, and SEIR all arrive at a similar conclusion of insignificant impact of Bay hydrodynamics outside of lease areas. Speculation about theoretical impacts outside of lease areas (including beaches) should be avoided in absence of such data.

Re: Sand is not being replenished in some areas, and undefined impacts where it is being replenished.

Response:

The impacts of sand mining on hydrodynamics and sediment transport are discussed in the SEIR (SIER 2025, p. 3.4-16) based on modeling studies by Anchor QEA (2023) and CHE (2025). Both studies agreed with the finding that “the Proposed Mining is not likely to cause a significant impact on sediment transport and sand budgets in areas outside the immediate vicinity of the lease areas” (SEIR 2025, p. 3.4-18). These studies do not suggest an “undefined” impact, but rather a negligible impact.

The negligible impact of sand mining on areas outside of mined areas is also supported by Deltares ring analysis (2023). As discussed above, impacts were largely confined to 100 meters outside of mined areas.

Re: Update to bay-wide sand transport studies.

Response:

The SEIR discusses several studies that explore geomorphic processes, including bay-wide sand transport, sediment fluxes of sand into the bay, bedform dynamics, and geochemical analysis to identify sand sources (SEIR 2025, p. 3.4-9). The SEIR acknowledges that the results from some of these studies, “are counter to the prior assumption of sediment connectivity thought to apply for the 2012 EIR” and that “[t]he revised hypothesis suggests that Suisun and San Pablo Bays receive limited sand supply from the Sacramento-San Joaquin Delta and instead are largely fed from local Coast Range sources.” (SEIR 2025, p.3.4-10). Despite new developments in Bay sediment dynamics, research detailed in the SEIR and above is consistent in pointing towards negligible impact outside of the lease areas.

Several public comments relate to findings from sand science studies which improve our understanding of sand movement (or lack thereof) within the Bay. The provenance analysis, led by UT Austin researchers, concluded that the Delta no longer provides a significant source of sand that is transported through the Bay to the outer coast. Instead, the Central Bay and outer coast comprise a shared pool of sand derived from tidal re-distribution of relic sand and sand sourced from outer coast erosion. Suisun Bay is comprised of sand from a different source and therefore the effects of mining should be considered separately.

While these results present a departure from the prior hypothesis of a Bay-wide connected system of sand transport, they also reinforce several key conclusions from the 2012 EIR and SEIR which include:

– Central Bay consists of vast deposits of relic sand deposited in the Late Pleistocene that have since been reworked by tides and waves to form large shoals inside and outside the Bay entrance. This fact was well documented in research which informed the 2012 EIR.
– The limited supply of sand from the Delta and lack of sand transport connectivity throughout the Bay further support the findings that mining does not significantly impact regional sand transport patterns.

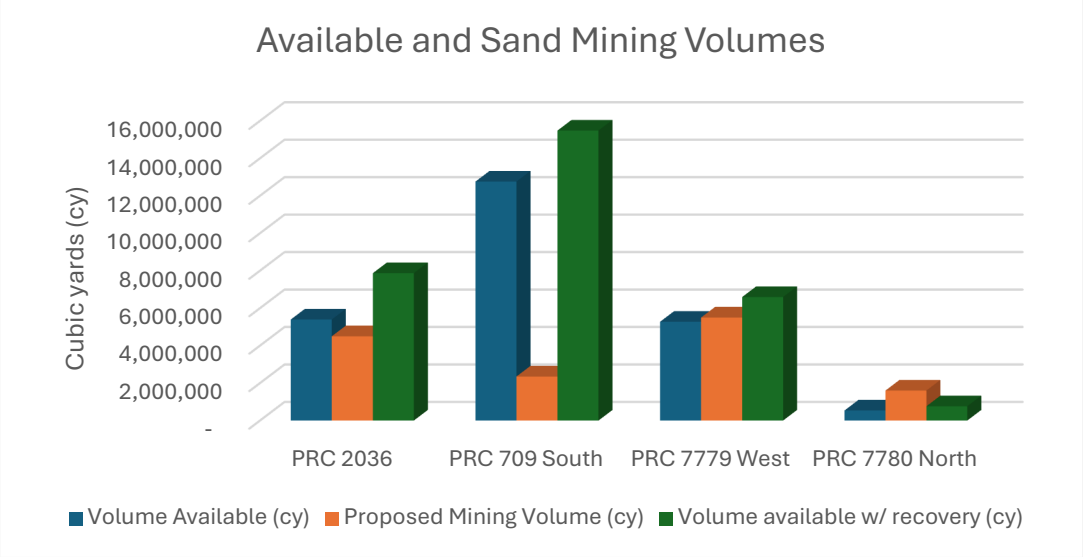
Mineral Resources

Re: Cumulative impacts, CHE report on available mining exceeds proposed volume.

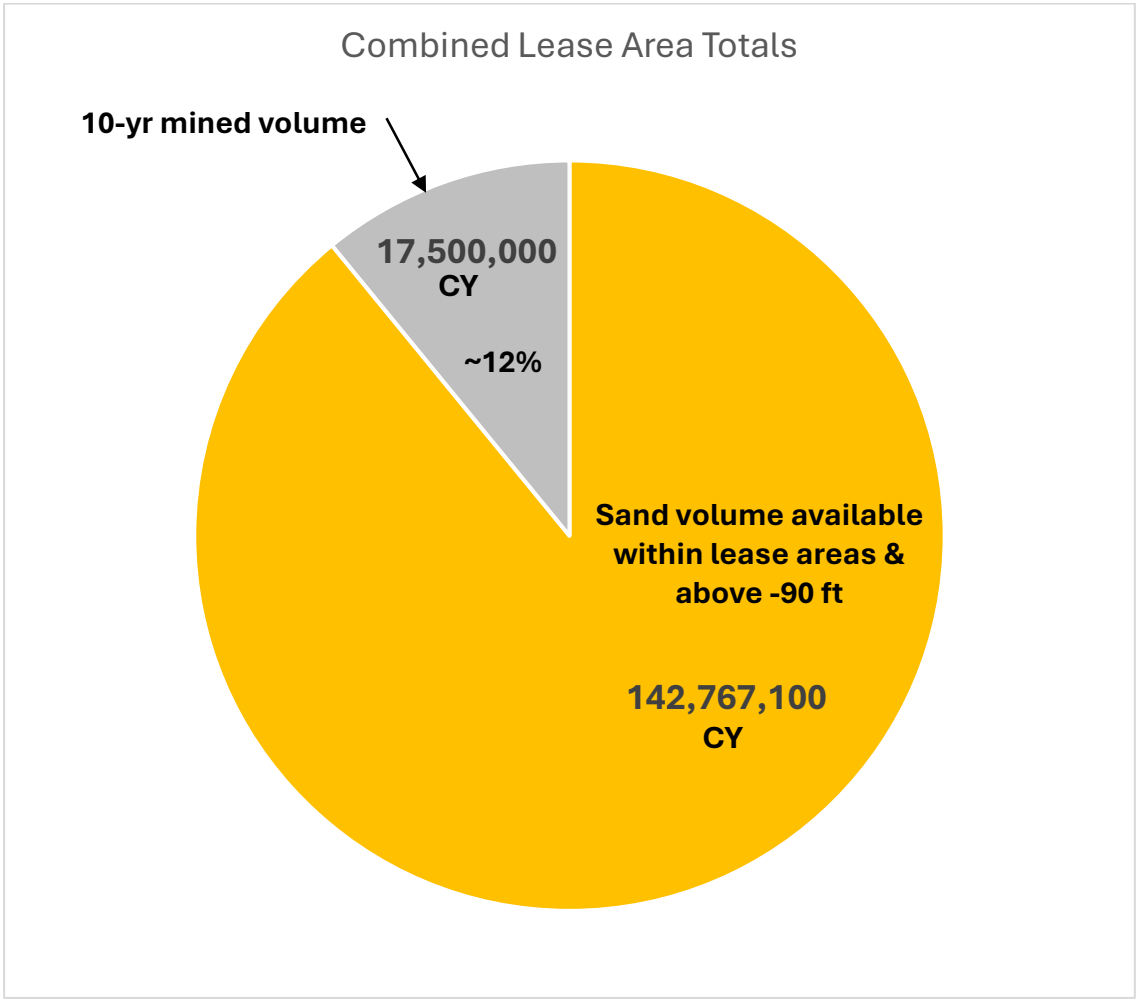
Response:

The SEIR does not dispute the findings from the CHE report (SEIR 2025, p. 3.3-8) but provides important context and limitations to the findings. The CHE modeling used a coarse and conservative approach by extracting the full 10-year cumulative proposed mining volume at one time (versus gradually over the 10-year lease period) and did not account for natural replenishment. Therefore, the modeling assumptions reflect an unrealistic, worst-case scenario to evaluate sand mining relative to the resources available, as appropriately described in the SEIR.

The "ring analysis" conducted by Deltares (2023) estimated recovery rates in mined areas, showing that recovery of mined volumes ranges from 14-116%. These recovery rates can be extrapolated across lease areas to determine a more realistic volume of available sand when accounting for natural recovery. The figure below shows a comparison of the available volume, proposed mining volume, and available volume assuming recovery. As stated by the SEIR, the proposed mining exceeds the available sand volume at PRC 7779 West; but when recovery is accounted for, the available volume exceeds the proposed mining volume.



For additional context, the proposed mining volumes represent ~12% of the sand volume available in lease areas (figure below), as assumed by the CHE (2025) report, without accounting for recovery. This percentage also includes some key assumptions, such as the assumption that mined volumes only represent 10-year totals, sand availability is only considered at lease areas shallower than -90 ft MLLW, and lease areas experience no recovery.



The depth and lease area assumptions associated with the CHE (2025) volume estimates are quite limiting when considering the total volume of sand in the San Francisco Bay. The ADEC (2000) study for San Francisco Airport Expansion Project identified additional sand deposits in the South Bay (Navigation Access Area and East Bay Sites) with potential sand volumes of ~150 Mcy. Geophysical surveying performed for the ADEC study also indicated the amount of sand at depths below -90 feet, MLLW is significant at the Point Knox Shoal, further evidencing that estimated volumes of available sand resources represent only a fraction of the sand within the SF Bay.

Geomorphology

Re: Lack of clarity of new findings that the Bay is disconnected and Bay budget.

Response:

The 2012 EIR and SEIR clearly state that sand mining removes sand from the Bay and that mining is not a sustainable activity as it extracts sand at a rate greater than replenished. As the SIER does not assume substantial sediment inputs, the comparison between mined volumes and local tributary inputs is irrelevant. BCDC's request that the scale of differences is important to highlight is already addressed by the sand budget, and more appropriately put in context with the amount of sand available in the Bay as described above.

The comparison provided above between mining rates and supply from Delta and local tributaries is also misleading as it fails to offer the large sand inflows from adjacent subembayments and outer coast as a comparison. The most recent Sand Budget (2025) estimates the total sand inflows to Central Bay are 1.03 Mt/yr, which is actually greater than the rate of mining. As described in previous responses, the Sand

Budget provides a limited picture of resource availability since much of the sand lies below the active transport layer.

Contrary to several comments, the SEIR addresses the topic of sand sources and connectivity based on studies conducted post 2012: “The revised hypothesis suggests that Suisun and San Pablo Bays receive limited sand supply from the Sacramento-San Joaquin Delta and instead are largely fed from local Coast Range sources. The authors interpret that sand in Central Bay is largely supplied by flood tides and wave-driven littoral transport that convey sand through the Golden Gate and into the Central Bay from the outer coast” (p. 3.4-10). Furthermore, at a Science Panel convened by BCDC in 2014, Dr. Patrick Barnard (USGS coastal geologist) referenced the isopach map of Central Bay prepared by Florence Wong and stated: “[T]he sediment is extremely thick. The volumes are out there…. [T]here are huge volumes of sediment, probably billions of cubic meters.” Thus, sand supplies within Central Bay, and more broadly the SF Bay, are vast, and will experience little depletion over a ten-year span of the current permits and leases. Further, the fact there may be local impacts (e.g., within the mined area itself) does not necessarily mean there is any significant impact on sand transport to other areas. Indeed, as modeled by CHE (2025), even a conservative impact analysis of mining indicates no noticeable impact elsewhere.

We appreciate the opportunity to review and provide this analysis of public comments on the various studies and findings detailed in the SEIR. Please contact me if you have any questions or require additional information.

Sincerely



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February 8, 2026

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Re: Response to Public Comments on the Supplemental Environmental Impact Report For the San Francisco Bay and Delta Sand Mining Project

Dear Ms. Guerra and Mr. Butler,

Thank you for the opportunity to peer review and respond to the recent non-governmental organization and agency comments on the Supplemental Environmental Impact Report ("SEIR") for the San Francisco Bay and Delta Sand Mining Project ("Revised Project").

The commenters assert that changed environmental conditions in the San Francisco Bay-Delta system will be exacerbated by continued sand mining. However, these assertions do not align with the substantial technical record developed since certification of the 2012 Environmental Impact Report ("2012 EIR"). Hanson Environmental has compiled and synthesized an extensive review of the available scientific literature as part of revised Biological Assessments conducted of the effects of sand mining within the Bay-Delta estuary (submitted in January 2026), which is in addition to independent review of available information by the California State Lands Commission (SLC) and its consultants Environmental Science Associates (ESA) and Applied Marine Sciences (AMS) as part of the scientific foundation used in developing the Draft and Final Supplemental EIR's. As documented in the extensive body of studies completed over the past decade, the effects of sand mining are consistently shown to be localized and temporary. There is no evidence of population-level or system-wide biological degradation attributable to sand mining operations. The Final SEIR adequately analyzes impacts on biological resources, including potential effects on special status fish species, benthic communities, and marine mammals. Hanson Environmental provides point-by-point responses to the biological resources comments in the sections below.

Longfin Smelt, White Sturgeon, Green Sturgeon, and Delta Smelt

Comment: Some commenters state that the scope of the effects analysis for aquatic species and their habitats fails to adequately respond to substantial new scientific evidence.

Response: The potential effects of sand mining within Central Bay, Suisun Bay, and Middle Ground Shoal, including an extensive list of operational measures specifically designed to minimize and avoid adverse effects to species such as longfin and Delta smelt, salmon and steelhead, and sturgeon, were assessed in the SEIR. In addition, environmental analyses presented in the 2012 EIR and more recent monitoring and surveys conducted in compliance with terms and conditions of State and Federal permits for sand mining in the San Francisco Bay-Delta estuary included supplemental studies of benthic disturbance (NewFields 2018a, 2020) and water quality and discharge plume dynamics (NewFields 2018b). These studies, in turn, were included in the assessment of potential adverse impacts to native fish and macroinvertebrates inhabiting Central San Francisco Bay and Suisun Bay where sand mining occurs. Results of monitoring over the past decade during sand mining events were also considered in the assessment. The assessment included consideration of the following issues:

- Increased suspended sediment concentrations and turbidity within the overflow plume;
- Contaminants and toxicity within the overflow plume;
- Changes to dissolved oxygen levels;
- Spread of invasive species;
- Entrainment into the suction head during mining;
- Effects of sound pressure (noise);
- Migration and habitat use; and
- Benthic disturbance.

It is incorrect to state that the SEIR and associated review of the biological resources surrounding the Revised Project did not consider more recent scientific evidence. Hanson Environmental has reviewed the more recent scientific references provided by the Farallon Institute and Baykeeper, and none provide any new linkage between sand mining and habitat or species declines.

Habitats of Particular Concern

Comment: Comments were received that expressed concerns regarding adverse impacts of sand mining on intertidal and shallow subtidal eel grass beds. Eel grass beds provide nutrients, cover for juvenile fish from predators, and substrate for spawning by Pacific herring. These habitats are limited to the higher salinity areas within Central Bay and do not occur upstream in the vicinity of Suisun Bay or Middle Ground Shoal.

Response: Eel grass beds occur within the intertidal and shallow subtidal zones of Central Bay. Within Central Bay localized, shallow water subtidal areas have been colonized by eel grass that

serves important habitat and ecological functions within the estuary. As a consequence of the relatively high turbidity and suspended sediment concentrations naturally occurring within the Bay-Delta estuary, light penetration (photic zone) limits the occurrence of submerged aquatic vegetation to relatively small, intertidal and shallow subtidal areas within Central Bay. The distribution of eel grass beds in Central Bay shows relatively small areas colonized in Richardson Bay, adjacent to the Alameda shoreline south of the Bay Bridge, near the Emeryville Marina, and in several locations along the Richmond shoreline. No eel grass beds were identified in the vicinity of the Martin Marietta sand mining lease locations in Central Bay. Further, sand mining in Central Bay is limited to subtidal areas greater than 30 feet deep and there are no known eel grass beds in the vicinity of the lease areas.

Since sand mining activity is limited in Central Bay to water depths from approximately 30 to 80 feet, sand mining activity would not involve excavation of eel grass beds. Further, dissipation of the overflow plume by sediment settling and turbulent dispersion, in addition to the short-duration (a period of hours) is not expected to result in adverse direct or indirect effects on eel grass. Tidal hydrodynamics within Central Bay contribute to rapid movement of the overflow plume on both flood and ebb tidal cycles, which further reduce the duration of increased turbidity. Consequently, no significant direct or indirect impacts to eel grass beds have been identified.

Effects from Exposure to the Overflow Plume

Comment: During sand mining an overflow plume is discharged that includes elevated turbidity and suspended sediments, plankton, and aeration bubbles. According to the commenters, sand mining has the potential to alter or modify subtidal habitats by changing water depth, substrate characteristics, water currents, or water quality (including suspended sediment concentrations) resulting in temporary behavioral avoidance by sensitive species of previously available habitat. These physical changes could affect the quality and availability of suitable habitat and habitat functions for protected fish species.

Response: Information on species habitat requirements, changes associated with sand mining activity, habitat types and functions in the sand mining areas, and the biological response of various species was used to assess potential effects of sand mining in the 2012 EIR, more recent SEIR, and intervening studies. A number of investigations have been performed throughout the world on the environmental effects of dredging and sand mining activities, and specifically on habitat quality and function for fish populations. Results of these investigations provided in the original 2012 EIR and Biological Assessments served as a foundation for use in assessing potential effects of sand mining on subtidal habitats and the protected fish species inhabiting the Bay-Delta estuary.

Based on results of this assessment it was concluded that exposure to the overflow plume during sand mining may affect winter-run and spring-run Chinook salmon, steelhead, white and green sturgeon, and smelt; however, the potential effect is temporary and localized. This conclusion is consistent with earlier analysis and is touched on again in the most recent SEIR and Biological Assessments. At base, the overflow plume is not expected to have any significant effect on fish species or their habitats.

Entrainment Risk

Comment: Several comments expressed concern that fish eggs and larvae would not be protected from entrainment into the drag head during mining and that the positive barrier fish screen would not be effective in protecting larger juvenile and adult fish from entrainment.

Response: The 2012 EIR committed the Lessees to the installation and operation of state-of-the-art positive barrier fish screens (cylindrical wedge wire screen with 1.75 mm screen openings and 0.2 fps design approach velocity) on all sand mining equipment. For example, impacts to larger juvenile starry flounder, English sole, northern anchovy, and other managed fish species greater than approximately 10-15 mm are expected to be negligible based on the effectiveness of the positive barrier fish screens. Planktonic eggs and larvae is expected to be very low given the short duration of sand mining events, the relatively small volumes of water entrained.

Likewise, installation and operation of the fish screens substantially eliminates the risk of entrainment for juvenile and adult fish species listed as threatened or endangered under the federal or state Endangered Species Acts; however, smaller larval fish (e.g., less than approximately 10-15 mm in length) would continue to be vulnerable to entrainment as acknowledged in the 2012 EIR, SEIR, federal Biological Opinions, and Incidental Take Permits (ITPs) issued by the California Department of Fish and Wildlife. For take that might occur of larval-stage special status fish species, the 2012 EIR, CDFW ITPs, and SEIR include compensatory mitigation, which the wildlife agencies have deemed sufficient to fully mitigate the loss and to avoid jeopardy to listed fish species. Such compensatory mitigation will also incidentally benefit other, non-listed species. No further mitigation is required.

Effect of Noise on Marine Mammals

Comment: Baykeeper in its February 4, 2026 comment letter on the Final SEIR, argues that the Final SEIR's analysis of noise impacts on marine mammal communication should have analyzed how increased whale presence, behavioral use, and vessel interactions affect risk under federal and state protected-species frameworks. Baykeeper discussed changes in marine mammal populations (gray whales and bottlenose dolphins) and habitat use within San Francisco Bay since completion of the 2012 EIR.

Response: The 2012 EIR evaluated potential effects of underwater sound generated during sand mining and determined that the potential impacts to fish and wildlife species from increased noise would be less than significant (SLC 2012). Noise generated by sand mining is marginally above ambient levels and only within the range detectable by fish (and possibly marine mammals) over a very small area of the Bay and Delta located immediately around the sand mining operation (SLC 2012). Because such impacts are localized, result in no permanent loss of habitat function or value, do not impede or prevent fish migration, and do not result in any direct loss in population, habitat, or biological diversity, the temporary increase in noise above ambient levels is expected to remain less than significant (SLC 2012). The 2012 EIR's analysis is further supported by the Final SEIR,

which further analyzed potential noise impacts using newly developed thresholds for the onset of auditory injury, including permanent and temporary threshold shifts from impulsive noise sources (e.g., pile driving) and non-impulsive noise sources (hydraulic suction dredging).

Data collected by the Marine Mammal Center and California Academy of Sciences (https://www.calacademy.org/press/releases/california-academy-of-sciences-confirms-24-dead-whales-in-the-san-francisco-bay-area) compile information on the number of whales and other mammals observed in San Francisco Bay, the number of whales that have been killed, and when possible the cause of the mortality. The data suggest that the greatest sources of mortality are vessel strikes and propeller damage. In the past decade, neither Martin Marieetta nor Lind Marine have experienced any vessel strikes by sand mining equipment. This may be attributed to the slow rate of travel by mining equipment in combination with U.S. Coast Guard warning to mariners regarding the presence and location of whales within the estuary. Based on these considerations, it was concluded that underwater sound generated during sand mining, while it may temporarily affect animal behavior in the immediate location of the mining activity, is not expected to result in potential displacement of any special status species.

Impacts to Migratory Pathways

Comment: Comments have been received regarding the potential effect of the overflow plume and benthic disturbance associated with sand mining on the ability of fish and Dungeness crab to migrate within the estuary.

Response: The channel areas where sand mining activity occurs are characterized by relatively high-water velocities that affect the distribution of overflow plume suspended sediments within the water body. Although fish could be exposed to elevated suspended sediment concentrations within the overflow plume, the orientation of the plume offers opportunities for fish to avoid the plume by moving laterally within the channel and therefore, plume exposure is not expected to result in a barrier to winter-run or spring-run Chinook salmon, steelhead, green and white sturgeon, and other fish movement or migration within the area. Further, exposure to elevated suspended sediment concentrations associated with the overflow plume is anticipated to be of relatively short duration (e.g., minutes or hours) and therefore, temporary reductions in the ability to capture prey is not be expected to result in detectable changes in growth rates, condition, or survival of fish inhabiting the estuary.

Habitat disturbance in estuaries, such as the San Francisco Bay-Delta, is a continuous process even under natural conditions. To survive in an estuary, an organism must adapt to a wider range of fluctuations in environmental factors than in the open ocean or in fresh water. Repeated deposition and re-suspension of the same sediments are predominant characteristics in the San Francisco Bay-Delta estuary due to the high energy of tidal influences. Superimposed on typical seasonal cycles of deposition and re-suspension are intense periods of erosion or deposition that can rapidly change the sediment surface.

As with natural conditions under tidal influence, sand mining also results in temporary disturbance to benthic habitat. Sand mining using the methods deployed by Lind Marine results in depressions

in localized areas; whereas, mining methods used by Martin Marieatta result in shallower depressions along the bottom. Dungeness crab have evolved in response to highly dynamic changes in benthic substrate including sand waves, some several meters in height, formed by high tidal currents and freshwater flows through the Bay. Since juvenile crabs have the capability of successfully migrating through these sand wave conditions, their migration to the ocean is not expected to be blocked or delayed by temporary benthic disturbances resulting from sand mining events.

Based on consideration of the effects of sand mining on habitat quality and availability within Central Bay and Suisun Bay, the 2012 EIR already recognized that sand mining may contribute to temporary, localized, and episodic behavioral responses that could affect Chinook salmon, steelhead, sturgeon, and their critical habitat. However, those impacts were addressed through avoidance, minimization, and mitigation measures and there is no evidence linking sand mining activity to blockage or delays in migration of juvenile crabs between the Bay and ocean.

Benthic Disturbance and Sand Transport

Comment: Baykeeper and others have commented on concerns regarding the effects of sand mining on benthic disturbance and offer critiques of the study design, sampling, and data analyses of the NewFields (2018, 2020) and AMS (2009, 2025) benthic sampling studies and analysis.

Response: While criticizing the methods and data produced from the most recent benthic studies, the commentors do not provide any contrary studies or data suggesting any greater impacts on benthic habitats or communities. Since the 2012 SLC EIR was certified, additional studies have been performed affirming that sand mining does not have a negative impact on the benthic community. The recent benthic studies of sand mining effects were developed and implemented to meet the requirements of the terms and conditions of the 2012 EIR and associated state and federal permits. The study design was developed collaboratively with oversight from regulatory agencies and experienced and qualified environmental consultants. Results of these studies have been documented in several reports that were reviewed and evaluated by the state and federal agencies in addition to the expert scientific panel prior to being finalized. The recent benthic studies represent the best scientific evidence available on the potential effects of sand mining on benthic habitat disturbance and repopulation of mining areas by benthic species following mining events.

As part of developing the 2012 EIR for sand mining, AMS (2009) conducted a benthic macroinvertebrate study in areas of Central Bay where sand mining by Martin Marietta occurs. Based on results of these surveys, AMS (2009) reported that no effect of sand mining on benthic communities within the Central Bay lease areas could be detected, which was thought to be attributable to the natural instability caused by high-energy tidal currents and dynamic sediment movement. Similarly, no significant differences were detected in the benthic communities where sand mining occurs in Suisun Bay (AMS 2009). The 2012 EIR concluded that sand mining results in short-term changes in habitat composition and benthic infauna and epifauna communities; however, these changes are temporary and do not appear to result in a detectable change in infaunal species composition or forage suitability.

NewFields conducted a later study in 2016 and 2017 which sought to characterize benthic habitat conditions and assess potential impacts of sand mining disturbance on habitat functions after 12 months. The study collected sediment profile images, sediment grain size and total organic carbon data, and benthic community data (e.g. benthic organism count for species, abundance, etc.). Samples were collected in areas where sand mining is permitted (treatment areas) and in control areas (where mining did not occur). After 12 months, there was no statistical difference between the areas that were subject to sand mining and the areas that were not.

In addition to the AMS and NewFields study results, regulatory agencies and SLC have limited sand mining to specific designated lease areas. Avoiding mining in shallow water areas and within 250 feet of shorelines serves to reduce and avoid the risk of benthic disturbance to sensitive subtidal habitat areas. Limiting the volume of sand that can be mined from a designated area each year further reduces potential adverse impacts from benthic disturbance. Mining by Martin Marietta and Lind occurs in areas with high sand concentration and low percentage of fine sediments, typically associated with high velocity areas and dynamic subtidal habitat conditions (highly mobile and frequently disturbed by tidal currents as reflected in sand wave formation and sand accretions). These naturally disturbed subtidal areas are typically colonized rapidly following benthic disturbance. Concentrating mining activity to relatively small areas limit the potential for benthic disturbance by sand mining to only a small portion of Central and Suisun Bays, and avoids potential benthic disturbance to a majority of the Bay-Delta estuary.

Based on results of these analyses it was concluded that sand mining contributes to localized temporary impacts to the benthic macroinvertebrate community where sand mining occurs; however, no long-term regional impacts have been identified.

Resident Killer Whale Critical Habitat

Comment: Comments suggest that the Environmental Setting and Effects Analysis presented in the Draft SEIR did not address the potential effects of sand mining on the survival and abundance of juvenile Chinook salmon, which is a prey resource for killer whales inhabiting coastal waters outside of San Francisco Bay.

Response: The effects of sand mining on survival of juvenile Chinook salmon has been assessed in the 2012 EIR, SEIR, Biological Opinions, and Biological Assessments for sand mining submitted to NMFS (2014 and 2026). All juvenile Chinook salmon and steelhead produced in Central Valley tributaries migrate downstream through areas where sand mining occurs. The installation of positive barrier fish screens on the sand mining suction pipe, which is designed to meet fish screen design criteria for Delta smelt (e.g., 1.75 mm screen openings, approach velocities less than 0.2 feet/second) and short duration priming and clearing of the drag head and suction pipe in close proximity to the bottom help minimize the risk of adverse effects on salmonids. There is no evidence of juvenile salmonid mortality as a result of operation of the positive barrier fish screens during sand mining events over the past 10-years. Further, the overflow discharge plume characteristics have been evaluated based on current mining activity and have not been found to result in toxicity, depressed dissolved oxygen, or turbidity concentrations expected to result in adverse affects to juvenile salmonids. The overflow plume occurs only temporarily during and after

mining events before dissipating, and there are opportunities for juvenile salmonids to avoid exposure to the plume. The low risk of adverse effects of sand mining on juvenile salmonid survival and abundance support the conclusion that the impact of sand mining on prey resources for Killer Whales or any other marine mammal remains less than significant.

Consideration of New Studies

Comment: How will new scientific information be incorporated into future operations and management?

Response: Martin Marietta and Lind Marine have consistently engaged in a collaborative science-driven process with State and Federal agencies and other interested parties. New information has become available over the past decade that improves the collective understanding of the effects of sand mining on the aquatic species and their habitat within the estuary. I have personally conducted a literature review of new studies related to potential impacts on biological resources from sand mining and believe that the new research supports the conclusions reached in the SEIR.

Sincerely,



Dr. Charles Hanson
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cc: Christine Boudreau, Boudreau Associates LLC
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