Developing Biological Goals for the Bay-Delta Plan: Concepts and Ideas from an Independent Scientific Advisory Panel

A final report to the
Delta Science Program

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EXECUTIVE SUMMARY

The charge to this Panel from the State Water Resources Control Board (Board) was to develop methods for formulating biological goals for narrative objectives in the Bay-Delta Plan. The focus is on evaluating status, trends and responses of targeted species and ecosystems to management actions that include major manipulations of flow and large-scale habitat restoration. The biological goals should be suitable for assessment using data from current monitoring programs and existing data. However, realizing the inadequacy of monitoring for some geographic areas, species, and topics, the Panel also considered additional metrics and methods that could significantly improve evaluation of species and ecosystem responses to restoration actions. The Panel divided the report into four main chapters: 1) Introduction and Background; 2) Ecosystem Structure and Function; 3) Native Fishes and Fish Assemblages; and 4) Salmon and Steelhead. Appendices 9.1 and 9.2 provide answers to specific questions asked by the Board to be addressed in each of these four areas.

Chapter 1 is a broad introduction to topics that are fundamental for understanding the results of this report. The Panel first answers the question "Why has the Board asked the Panel for this report?" The next section discusses why the San Francisco Estuary and its inflowing rivers need to be treated as novel ecosystems, consisting of a mixture of native and non-native species living and interacting in a highly altered environment. Next, the report emphasizes that the combined effects of climate change, increasing water demand, and local modifications are resulting in trends that can have substantial effects on the riverine and estuarine ecosystems and their fishes. These changes should be considered when setting and evaluating progress towards biological goals. Finally, the report briefly discusses the need for experimental (adaptive) management to test the results of management actions. The main purpose in reviewing these challenges is to highlight that making useful inferences about the effects of an action depends on both selection of appropriate objectives and variables to measure and on the experimental designs that determine when and where actions are implemented.

Chapter 2 examines structural and functional components of the aquatic ecosystems of the major tributary rivers and estuary, emphasizing components that support native fishes by providing habitat elements such as riverine flow, physical configuration, water-quality attributes, food, and shelter from predators. The chapter distinguishes between programs of monitoring and evaluation focused on river ecosystems and those focused on the estuary. The estuary has a longer and richer database upon which to develop goals, objectives, metrics, and performance measures, but the need for monitoring and evaluating tributary rivers is strong, with plans for management actions for both flow and habitat currently focused on rivers that flow into the estuary.

Several structural ecosystem elements provide essential underpinning to analyses of fish responses. These include physical (e.g., temperature, flow, geomorphology, turbidity, and conductivity or salinity) and chemical (e.g., nutrients, dissolved oxygen, contaminants) properties. Biotic structural components include the main primary producers that form the base of the food webs (algae and plants) and aquatic invertebrates that provide much of the food for young fish.

Functional components of the ecosystems include rates of primary and secondary productivity that support native fishes. Few, if any, functional components are currently monitored in the rivers or estuary, although some could be determined from existing monitoring data. Functional measurements in the rivers could include bioassays of attached algae and quantifying the abundance and biomass of aquatic invertebrates. In the estuary, phytoplankton primary production and zooplankton secondary production can be estimated from existing monitoring programs. The report emphasizes, however, that only limited monitoring of the kind needed for these assessments now occurs in the rivers, and even in the estuary some important components are not assessed, such as vegetated margins and wetlands of the estuary that support young Chinook salmon. These gaps should be filled to support the overall focus on salmonids and other native fishes.

Generally, the ecosystem attributes the Panel recommends will be most useful in assessment and support of goals set for fish populations, rather than in being goals themselves. Two main exceptions to this would occur: 1) where food web support for fishes is weak (e.g., parts of the estuary), in which structural and functional measures may form the basis of goals; and, 2) where the actions involve alterations of physical habitat.

In Chapter 3, using fish metrics directly, the Panel addresses the question "How do we know if large-scale management actions taken to improve conditions for native fishes, either individually or cumulatively, are doing any good?" The general method to answer this question for native fishes in the Delta is to use abundance metrics for 36 species that are/were common in the Delta. Fish abundance is based on data collected by 18 different fish surveys. The surveys are briefly described to emphasize the diversity of information and relatively long time series of data that they provide. While this wealth of data was not available for the lower reaches of rivers, the same basic approach was used to develop metrics for them using more limited data.

An important aspect of the Panel's approach for native fishes is to go beyond using just fishes listed by state and federal endangered species acts to evaluate change. To do this, the Panel used responses of unlisted native fishes as well as non-native fishes. The contention is that assessing responses of a wide range of fish species representing assemblages with diverse ecological and life history characteristics should expand the ability to detect and identify mechanisms of change. The collective responses of multiple species can then be related to management of the environment to favor listed species or other native species.

The report presents six potential metrics that use river and Delta fishes for evaluating progress toward biological goals, recognizing that progress is likely to be slow and incremental. These metrics can be either positive or negative indicators of change. The metrics include: (1) species abundance trends from the surveys, either individually or collectively; (2) the distribution and abundance of selected fish species as indicator or focal species; (3) the collective abundance and distribution of warm-water non-native fish species as a negative indicator; (4) abundance and condition (e.g., growth) of fishes using Suisun Marsh as a nursery; (5) abundance and distribution of invasive plant, invertebrate, and fish species as an expanded version of metric 3; and (6) the distribution and abundance of native fish assemblages in tailwaters below dams. Other metrics, such as the Index of Biological Integrity (IBI) were evaluated but found to be less useful. The value of the six abundance metrics would be enhanced if information was also available on health of individual fish, using measures that can respond more rapidly to change than populations: length-weight relationship (condition factor), growth, histopathological condition, diet, and recruitment and mortality rates. In addition, for some well-studied species (e.g., striped bass), Viable Salmonid Population (VSP)-type metrics could be used for modeling the populations dynamics (see Chapter 4 for a VSP description). Finally, incorporating landscape metrics (e.g., extent of aquatic macrophytes in the Delta) could provide additional insights into fish responses to a changing environment. Ideally, a broad-based approach would be used, using several or all of these metrics simultaneously, while models of factors affecting fish abundance and health are developed. This report emphasizes that the diversity and length of the 18 sampling programs that focus on abundance provides an unusually rich basis for developing a range of fish-based metrics.

In short, given the complex life history patterns and ecological relationships that Delta fishes have with their environment, the most effective approach is one that uses multiple approaches and metrics. This is particularly true because most of the metrics are not sensitive to short-term (one year or less) changes, but are useful for measuring longer-term changes that might allow distinguishing fish responses to management actions from responses to natural environmental variability. Collectively, the fish and other metrics discussed here, especially those based on the 18 surveys, are most likely to tell a meaningful story after 5-10 years of change. Regardless, there is a need for the 18 standard surveys to continue, for expansion of selected existing surveys, and for new surveys and metrics to develop that refine information on the existing focal fishes while also targeting additional fishes.

Chapter 4 recommends that VSP criteria should form the basis for developing biological goals for natural-origin Chinook salmon and steelhead populations. Abundance and productivity are the most important and intuitive metrics for setting biological goals. In addition, diversity and spatial structure are key to population stability and resilience in a variable and changing environment.

The recommended framework for evaluating abundance and productivity of salmonids accounts for density dependence in survival rates by using stock-recruit relationships that quantify the relationship between parent spawners and their progeny (i.e., juveniles or adults). This approach is needed because density dependence can occur even at low salmon densities, and compensatory density dependence provides resilience and stability during periods of declining abundance.

Intrinsic (maximum) productivity can be estimated by using spawner-to-smolt or spawner-to-adult stock-recruit models; the former reflects conditions solely within the natal watershed, while the latter is also influenced by conditions in the Delta and the ocean. Intrinsic productivity determines the rate of population growth at low abundance. An obvious goal for intrinsic productivity is for the number of returning adults produced per parent spawner to exceed one. Trends in intrinsic productivity should be evaluated over time using a modeling approach to determine whether conditions are improving and progress is made towards biological goals. Additionally, the number of natural-origin adult salmon returning from the parent spawning population should be examined over time to evaluate whether a population that is heavily supplemented with hatchery spawners is sustainable if hatchery fish were removed from the spawning grounds.

A key goal for abundance is to have sufficient spawners to maximize future production of juveniles or adults, as estimated from the stock-recruit relationship. Additional recommendations are provided in Chapter 4. Ultimately, however, the Panel recommends tracking trends in productivity and abundance in response to management actions rather than setting specific targets for future abundance or productivity. Positive trajectories would be a key indicator of success.

Diversity and spatial structure contribute to the viability of salmonid populations in a variable and changing environment, and their attributes are reflected in stock-recruit relationships. A key practical biological goal for diversity is to reduce the number of hatchery-origin salmonids that spawn in rivers as a means to allow natural-origin populations to adapt to the environment and productivity to increase. Spatial structure of salmonid populations could be enhanced by restoring access to previously occupied habitats. A practical goal is to ensure viability of natural populations in existing habitats.

The Panel recommends a covariate stock-recruit approach for quantifying population-level responses of salmonids to management actions and to inform development of biological goals.

To demonstrate this approach, an example using juvenile and adult spawner abundance data for fall-run Chinook salmon in the Stanislaus River is provided. Analyses such as these may provide an indication of key factors affecting survival, but quantification of population-level responses to management actions may take 20-30 years (4-6 generations) depending on the strength of the effect of the management action, natural variation in survival rates, imprecision in monitoring, and generation time. Detection of juvenile salmonid responses to management actions in tributaries may be quicker than for adults.

Monitoring of natural-origin Chinook salmon in the Central Valley has improved in recent years, but key metrics need to be consistently measured each year in every watershed of interest. Additional effort is needed to estimate returns of natural-origin Chinook salmon if viability of the natural population is to be evaluated in relation to management actions. This means that abundances of hatchery-origin salmon in harvests and spawning grounds must be accurately estimated. Steelhead monitoring has improved in a few watersheds, but data for natural-origin steelhead are mostly inadequate and may not improve given their low abundance. Ultimately, data are needed to create "brood tables" for each population to support stock-recruit analyses.

More generally, the Panel emphasizes that well-designed programs of assessment should accompany any major actions in the future. The cost of major actions is high, as is the uncertainty of responses of fish populations to purposeful changes in flow or habitat. Therefore, evaluation of any major action should be embedded in an experimental management program that includes predictive modeling, adequate monitoring and ancillary investigations, and rigorous field-based assessment of outcomes. Such actions should include a built-in system of assessment to determine whether the monitoring program, expectations for the action, or even the action itself should be amended.

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1 INTRODUCTION

Defining biological goals for managing and restoring aquatic ecosystems is challenging. The scientific literature is replete with publications that consider biological goals for a wide variety of ecosystems. Yet, goals must be tailored and scaled to the problems to be solved. The job is particularly challenging for the complex landscape of the San Francisco estuary (SFE)[1] and watershed, which includes two major rivers, multiple tributaries, and a large and heterogeneous estuary. Biological goals also need to be clear, concise, quantifiable, and achievable within appropriate timeframes. Furthermore, biological goals will vary by broad aquatic ecosystem type (tributary rivers, major rivers, and the estuary), and goals ultimately must be designed relevant to specific ecosystems for which defined biological components have been identified.

This independent scientific advisory panel (Panel) was established to advise the State Water Resources Control Board (Board) on developing quantitative biological goals (see Glossary; Section 1.6) for measuring and assessing ecological responses to actions taken under the Board's Bay-Delta Plan (https://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/). Biological goals are intended to be linked to on-going monitoring programs and to build upon long-term datasets. However, the Panel also can recommend areas where additional monitoring and targeted studies, coupled with analysis, synthesis, and integration, might be needed to enhance understanding of these novel and changing ecosystems.

The Board charged the Panel to develop recommendations for formulating quantifiable biological goals for three topics: (1) ecosystem structure and function; (2) native fish species; and, (3) salmonids. The Panel was not asked to explicitly create biological goals, but to provide guidance on methods or approaches to formulate biological goals. The spatial scope that the Panel was asked to consider includes the San Joaquin River and its major tributaries (including the Merced, Tuolumne, and Stanislaus rivers), the Sacramento River including Sacramento River tributaries and Delta eastside tributaries (Mokelumne, Cosumnes, and Calaveras rivers), the Delta, and Suisun Marsh (see Section 1.3).

Healthy, self-sustaining riverine and estuarine ecosystems provide crucial ecological and social services that are essential for human populations (Palmer et al. 2005). As a consequence, billions of dollars are spent on the restoration of these ecosystems, but most restoration

[1] In this report, "Delta" is used to mean the legal Delta, but an expansive view of the charge is taken to include as much of the estuary as appropriate for evaluating potential effects of proposed actions on habitat, processes, and species.

projects go unevaluated (Bernhardt et al. 2005). Numerous restoration projects within ecosystems covered by the Bay-Delta Plan are in some stage of development, but few include detailed programs for long-term monitoring and evaluation or a rigorous adaptive management plan. Clear and concise quantitative biological goals are required to evaluate whether these restoration projects are effective.

1.1 Why has the Board asked the Panel for this report?

The Board sought recommendations from the Panel for scientifically defensible methods that can facilitate the formulation of biological goals to assess progress toward achieving the narrative objectives in the Bay-Delta Plan (Plan) and to inform adaptive management and future changes to the Plan as necessary. This is a broad and challenging charge. Board staff requested that the Panel provide written recommendations for formulating quantifiable biological goals that are designed to assess both: (a) the status and trends of representative salmonids and native fish assemblages; and, (b) the ability of ecosystems to support native fishes. The Panel was not charged with creating the biological goals themselves, but to provide guidance and recommendations on the methods or approaches that should be used to formulate and evaluate biological goals.

Well-crafted biological goals are intended to serve as quantitative benchmarks for assessing viability, recovery or decline of populations of native and non-native fishes and to characterize biotic and abiotic conditions within aquatic ecosystems where these fish reside. Biological goals should be consistent with current scientific knowledge, including information regarding viable populations and recovery plans for listed species. The biological goals should be clearly stated, such that progress towards achieving the goals or an understanding of why goals are not being met can be assessed through regular review, analysis, and synthesis. Proposed updates to the Plan will facilitate adaptive management that uses active experimentation to improve implementation measures and inform future revisions to the Plan, including adjustments to the biological goals and improvements to monitoring programs.

The Board is considering management actions that involve both modifications to river flows and non-flow actions (e.g., habitat restoration, predator control, barrier installation and operations, water quality enhancements) to improve habitat quality and the status of native fish species in the estuary and connected rivers. There is considerable debate among stakeholders about the effectiveness of flow and non-flow management actions for improving fish populations. In practice, some non-flow actions such as habitat restoration are strongly influenced by characteristics of flow, including magnitude, frequency, duration, timing, and rate of change.

1.2 Charge Questions

The Board developed a series of charge questions for consideration by the Panel that addressed the three topics. The Panel in turn made suggestions to rephrase some of the Board's charge questions while maintaining the intent of inquiry requested by the Board, thereby facilitating more direct and helpful answers to the questions. Direct answers to these detailed charge questions and 19 additional questions involving salmonids are provided in Appendix 9.1 and Appendix 9.2, respectively. Briefly, the major themes for the charge questions under the three topic areas, and how the Panel approached them, are as follows:

(a) Questions concerning ecosystem structure and function are directed towards how to measure the effectiveness of flow regimes, habitat restoration, and other non-flow actions (collectively “management actions”) for improving ecosystem condition. More specific questions ask what ecosystem processes should be monitored. Some important differences exist between the estuary and tributary rivers, including distinctive hydrodynamics, variable salinity distributions, the role of floodplains, and the duration and extent of monitoring and research programs. Methods and approaches to develop or enhance monitoring and evaluation programs are presented both for the estuary and for the tributary rivers.

(b) Questions about native fishes other than salmon and steelhead focus on developing biological goals using existing monitoring programs. More specific questions ask if viable salmon population (VSP) metrics are applicable and useful for other native fishes and whether such metrics exist, if there are other metrics to consider using existing monitoring data, and what additional parameters should be monitored for native fishes. The Panel determined that just monitoring native fish species was insufficient, especially for the estuary, because: (1) some native species, such as delta smelt, are so rare that inferences from these data are likely to be very weak; (2) monitoring data or biological knowledge of most other native species is insufficient to support meaningful evaluations; (3) some non-native species such as striped bass have been comparatively well studied and are more likely than most native species to show responses to management actions that would indicate the condition of the estuary or river; and, (4) some abundant non-native species are associated with habitats that are unfavorable for native species, and could thus serve as negative indicators of responses. The Panel also determined that using fish assemblages that encompass a wide array of native and non-native species had promise for determining broad-level responses to change.

(c) Questions about natural-origin Chinook salmon and steelhead populations focus on the use and relative importance of VSP criteria for: (1) evaluating status and trends; (2) measuring population-level responses to management actions; and, (3) setting biological goals. VSP criteria include abundance, productivity, diversity, and spatial structure. More specific questions ask about the influence of hatchery-origin salmonids, and the ability of current monitoring programs to track salmonid population responses to management actions and to track progress towards biological goals. The time frame for detecting population responses is discussed.

Assessment of monitoring methodology and the selection of metrics to monitor are common themes throughout the charge questions. Many types of monitoring have been applied to aquatic ecosystems to evaluate current condition, trajectories of change, and the efficacy of management actions. For example, McDonald et al. (1991) defined seven types of monitoring (trend, baseline, implementation, effectiveness, project, validation, and compliance) for river and riparian condition in forested catchments, with parameters grouped into six categories (physical and chemical constituents, flow regime, sediment characteristics, channel characteristics, riparian zone assessment, and aquatic organism studies). Similarly, Crawford and Rumsey (2011) recommended guidance for monitoring salmon recovery. The monitoring being conducted in various parts of the basin covered by the Plan already includes most of these elements (broadening the assessment of flow regime to include tidal flows), and monitoring design depends on management goals, biological communities, and species of interest.

Determining the full extent of the purpose, types, and details of extant ecosystem and fish monitoring throughout the entire SFE is beyond the scope of work for the Panel. The extensive monitoring for fish in the estuary, however, is documented and summarized here. Where ecosystem monitoring and evaluation programs are less well documented, we refer for guidance in this report to several successful, well-designed, long-term, and large-scale ecosystem monitoring programs that evaluate biological and ecological goals associated with major restoration projects and water quality upgrades in other parts of the world. For salmonids, we recommend approaches that can be applied to any population where data are available, and we provide one specific example.

1.3 The River-estuary System: A Novel Ecosystem

The SFE and its watershed comprise about 40% of the area of California. The current condition of this system and the myriad alterations that have brought it to this condition are well documented in many published reports (e.g., Nichols et al. 1986, Brown and Moyle 2005, Cloern et al. 2016, Whipple et al. 2012). This report does not describe these alterations here except to highlight key features that interact with the relationships among flow, habitat, and ecological structure and function for fishes and other aquatic organisms. This is a novel ecosystem that could be improved for the benefit of native species but not returned to its former state.

Several features of the SFE and watershed are prominent in our discussion about these distinct areas. The rivers and the estuary are connected by movements of water, substances, and organisms, but the two differ substantively because key elements of ecosystem structure and function are very different. Nevertheless, both rivers and the estuary are subject to anthropogenic and other changes that are effectively permanent. They also are affected by shorter-term, more ephemeral variation such as that in flow and temperature. In Chapter 2, salient features of this overall system and some key differences between the rivers and the estuary are briefly discussed.

The land-water interface throughout the system was long ago modified to convert low-lying lands to farms, towns, and cities, resulting in channels confined by levees to protect property from flooding. The levees confined flows to channels with steep, armored sides, reducing habitat value for aquatic species, providing locations favorable to predators of native fishes, and eliminating most floodplains and tidal wetlands. The ecological value of these lost wetlands is being elucidated through studies of the Yolo Bypass and other floodplains (Sommer et al. 2001, Crain et al. 2004, Moyle et al. 2007, Opperman et al. 2017), restored riverine and tidal habitats, and of small remnant tidal wetlands (Grimaldo et al. 2009b, Howe and Simenstad 2011). Information from these studies has led to increasing efforts to modify the land-water interface through manipulations of flooding and restoration or reconstruction of wetlands. However, current land uses in many locations constrain expanding the extent of floodplains or the area subject to tidal flows.

Except during high-flow events, water infrastructure and operations control flows in the rivers and the net (tidally-averaged) flows in the estuary. Water infrastructure including dams, modified channels, and diversions can be considered permanent features of the landscape, although operations can be modified to manipulate flows. The influence of these structures and operations are central to understanding the effects of flow alteration and management. In the rivers, the hydrograph has been modified across all time scales. For example, the winter-spring runoff peak has been truncated, flows are sustained at artificially high levels into summer, and diurnal fluctuations are unnaturally amplified. These alterations have identifiable effects on the riverine ecosystems by reducing connectivity among habitats, mismatching flow patterns needed for natural establishment of riparian vegetation, removing critical flow-linked cues in the life histories of salmon and other native fishes, and stranding fish during rapid changes in flow.

The direct effects of freshwater flow on the estuary decrease in a seaward direction as tidal flows become more influential in moving organisms and substances. In the Delta, net flows are important, because they deliver fresh water to the south Delta export pumps along with fish, other organisms and diverse substances (Jassby 2005, Kimmerer 2008). The effects of variation in freshwater flow on brackish to saline parts of the estuary occur mostly through movement of the salinity field (represented by "X2", Jassby et al. 1995) with accompanying changes in physical dynamics (Kimmerer 2002). Therefore, even large flow manipulations have minor effects in the more saline reaches of the estuary west of the Delta.

Non-native species now dominate fish, aquatic invertebrate, and aquatic plant communities in both many rivers and the upper estuary (Nichols et al. 1986). Some introduced species (e.g., Brazilian waterweed, water hyacinth, overbite clam, Siberian prawn, Mississippi silversides) exert stronger influences on ecosystem function than less influential non-native species. The effects of invasive species include disruptions of the extant ecosystem following the introduction event and changes in how the ecosystem functions after it has settled into a new state. Species have been introduced through a variety of mechanisms, including release of organisms considered beneficial by agencies and by shipping, aquaculture, aquaria, and live-bait operations. In many areas, habitat degradation has created conditions that favor assemblages of non-native biota over the historically native species assemblages. These assemblages, such as those associated with Brazilian waterweed and water hyacinth, support valuable sport fisheries for non-native fishes such as largemouth bass, sunfishes, and catfishes. The highly altered ecosystems that are dominated by non-native species can be regarded as novel ecosystems that require special management if they are to support native species.

Discharge of contaminants and nutrients affects aquatic ecosystems, although their effects can be difficult to quantify. Numerous chemicals released into the watershed and estuary from agricultural and urban activities have known toxic or other adverse effects on aquatic organisms (Weston et al. 2015). Nutrients are generally considered stimulants to phytoplankton production, although ammonium has been implicated in some of the declines in phytoplankton productivity in the estuary (Dugdale et al. 2007, 2012). Nutrient loading into the Delta also may stimulate periodic blooms of nuisance algae (*Microcystis*) and the invasive waterweeds that choke some Delta waterways and otherwise modify fish habitats (Dahm et al. 2016).

The extreme modification of the river-estuarine system has considerable implications for this charge. For one thing, the past is now only a rough guide to the future. Baseline surveys of fish go back only 60+ years or much less (Chapter 3). Nevertheless, the surveys record major changes to the ecosystem and are still valuable for assessing present conditions and predicting faunal changes in response to environmental change.

1.4 Future, Long-term Change

External change (change due to causes other than the planned actions) may interact with planned actions or confound interpretation of their effectiveness. It is common practice to consider recurrent seasonal and interannual variability in taking actions or conducting analyses.

Longer-term change, including climate change and human activities, poses greater challenges both for achieving goals and for measuring progress toward them.

Long-term changes are expected within the estuary and the catchment. For example, a major role of riverine flood flows is to supply the estuary with sediment, yet water in the estuary has become more transparent as the sediment pulse resulting from historical hydraulic mining has been winnowed out (Schoellhamer 2011). This trend is expected to continue because much of the sediment currently mobilized in the catchment is now trapped behind dams. Ongoing sea-level rise will further inundate wetlands and, where infrastructure blocks landward migration, marshes may drown unless plant biomass production and accretion of sediment can enable them to keep up with rising seas. Thus, the sediment shortage will exacerbate the effects of sea-level rise in restricting the scope for maintenance and restoration of wetlands in some areas. In addition, increased water clarity from reduced sediment transport degrades habitat quality for some fishes, such as delta smelt (Feyrer et al. 2007).

Climate change also will undoubtedly influence responses of ecosystems and fish communities to significant changes in flow and habitat. Effects of climate change vary across the landscape (Cloern et al. 2011). Increased atmospheric warming affects both rivers and the estuary, although water temperature in the more seaward reaches of the estuary is influenced more by ocean conditions than local atmospheric conditions. Warming is already causing a shift to earlier snowmelt runoff in the Sierra Nevada (Roos 1989, Stewart et al. 2004). Earlier snowmelt and a projected increase in the frequency of large storms and more frequent extremes in climate (Dettinger et al. 2016) will reduce water availability for the environment and people because dam operations must balance reducing downstream flood risk during the wet season with water storage for use during the dry season.

Other potential long-term changes include modifications to infrastructure (e.g., further urbanization, new or altered roads or pipelines), adaptation or migration of species in response to climate change, further introductions of new species, revamped fish hatchery operations, failure of levees resulting in long-term or permanent reconnection of subsided lands to waterways, and changes in local and regional wastewater treatment and discharge. Large-scale planned and unplanned changes will have effects that in most cases are difficult to predict, partly because they have not been adequately studied. For example, recent proposals to divert fresh water from the Sacramento River in the eastern Delta (WaterFix[2]) have been developed in part because the current diversion system has unwanted consequences, but the environmental effects of the design and operation of any new diversion still await further critical analyses.

[2] The "twin tunnels," https://californiawaterfix.com/

The combined effects of global climate change and local modifications result in trends that are likely to have substantial effects on the riverine and estuarine ecosystems and their biological species (Cloern et al. 2011). For example, increasing water temperature, decreasing turbidity, and increasing salinity intrusion into the Delta will likely reduce the extent of suitable habitat for delta smelt (Feyrer et al. 2011). Likewise, increasing water temperature, changing hydrographs, and poor survival during juvenile migration through the Delta are growing impediments to the viability of salmon populations (Lindley et al. 2007). Tidal wetland restoration will likely result in improvements in some ecosystem attributes that benefit native fishes through increased food web and habitat support, although the spatial scale of planned restoration is relatively small. Finally, further introductions of new invasive species may further disrupt the system, with unpredictable effects. In fact, the multiple effects of climate change tend to favor non-native species, increasing the difficulties for conservation of native species (Moyle et al. 2013).

1.5 Experimental Management

Ideally, metrics used to track progress toward biological goals should quantify the effects of management actions on target species. The responses of fish populations to management actions are often highly uncertain; therefore, planned management actions may not have the hoped-for effect. Uncertainty in population responses to changes in flow and habitat is shared by some decision-makers. For example, biological objectives for the Stanislaus River are intended to “define success” of management actions and ”measure progress in a transparent fashion” (SEP Group 2016). Advocates for voluntary agreements to define flow and habitat enhancement actions in the Central Valley (CV) emphasize the need to put “a system in place for adjusting this [water] allocation over time as scientific understanding about what works improves” (Mount and Hanak 2018). Such statements embody the main principle of adaptive management (Walters 1997), which is to reduce uncertainty over time by treating management actions as large-scale field experiments, either through explicit manipulations (active) or by observing variation in an experimental framework (passive).

Measuring the effectiveness of management actions, such as increasing flows to a tributary, has some fundamental science requirements: 1) effectiveness must be quantified by measuring variables that relate to objectives of implementation (e.g., response of target species); and, 2) responses to the action must be large enough, and measurements precise enough, to reliably quantify responses. These requirements can be difficult to achieve. For example, many decisions regarding instream flow in the western U.S. in the 1970s and 1980s were based on simple habitat models that predicted depth and velocity (e.g., PHABSIM, Gore et al. 2001). However, the approach largely failed because the responses of fish populations to these simplistic metrics were never adequately quantified, the wrong variables were selected, and uncertainty in model predictions was ignored (Castleberry et al. 1996, Railsback 2016, Williams et al. 2019).

Monitoring design must also account for imprecision or bias in measurements that may mask responses. As one example, a change in smolt production will take longer to detect if annual estimates lack precision and at least a decade of monitoring under each treatment is required to detect a reasonably large effect for most programs (Bradford et al. 2005). In addition, increases in river flow may lead to underestimates of smolt production if high flows reduce detection probability. These issues can largely be resolved by designing programs that anticipate the magnitudes of responses, implementing robust monitoring programs, and accepting that outcomes of management experiments at the population level can sometimes take decades to resolve.

Alternatively, responses can be measured over short periods of the life cycle of target organisms, such as the incubation period of salmon eggs or short-term segments of fish passage from watersheds through the estuary. This approach eliminates some extraneous variability (e.g., that in subsequent survival), but requires an assumption that improved survival at one life stage does not impair subsequent life stages.

In controlled laboratory settings, a cause-effect relationship between one or a few treatment variables and a response variable is determined by randomization, replication, and keeping all other factors constant. Such experimental designs are more difficult to implement in field settings. For example, to evaluate potential effects of increased flow in a tributary, average salmon smolt production could be measured during years before and after flow increases or with and without management actions to increase flow. However, no two years are identical, so attributing interannual changes in smolt production solely to differences in flows is an overreach. One solution to this problem is to use a BACI (Before/After and Control/Impact) design (Underwood 1992) where a similar river with similar attributes is used as a control or where a restored river reach is compared to an unrestored reach of the same river. The difficulty of interpreting diagnostic metrics is amplified if flow is increased and habitat enhanced at the same time, making it difficult to separate two likely interacting effects. On a more optimistic note, natural flow variation will likely provide contrasts over periods long enough to make inferences about the effects of purposeful flow manipulations.

The main objective in reviewing these challenges is to highlight that making useful inferences about the effects of a management action depends on both the selection of appropriate objectives and variables to measure and on the experimental designs that determine when and where actions are implemented.

1.6 Principles and Criteria for Developing Goals

Biological goals should be developed from agreed-upon principles and sets of criteria to assess success and to ensure transparency of intent in the development of goals. Without clear and accepted criteria, there is limited incentive for those restoring waterways to assess and report progress and outcomes. Palmer et al. (2005) presented five criteria for assessing success of river restoration, and the Panel has paraphrased and adapted these criteria for the purpose of developing biological goals more broadly:

- The design of goals and restoration projects should be based on a specified guiding image of a more desirable ecosystem state.
- The ecosystem's ecological condition should be measurably improved.
- The ecosystem should be more self-sustaining and resilient to external perturbations with as little ongoing maintenance as practicable.
- Management actions toward achieving biological goals should not impose lasting harm to the ecosystem.
- Biological goals should be developed so that responses to management actions are assessed in a framework of experimental management with results and data made publicly available.

Throughout this document, the development and assessment of metrics of condition for the ecosystem and native fishes in general and salmonids in particular are emphasized. The reason for this emphasis is that quantitative biological goals must be based on metrics that can be reliably measured. Therefore, the key to this approach to developing goals is to determine metrics likely to be relevant either for assessing movement along desired trajectories (especially reversing declines in fish populations) or for determining the causes of movement along or away from those trajectories. The following criteria were applied in selecting potential metrics to be used for evaluating progress as well as selecting methods for developing broad system goals, more specific objectives, and quantitative metrics. Therefore, metrics for assessing progress toward goals should be:

- Relevant: Metrics should be related to the abundance or productivity of the target species or to the availability and quality of their habitat including shelter, flow, and food. This will force assessments to be directly relevant to the goals.
- Responsive: Specific actions should be clearly linked to attributes that can be manipulated or that will respond to the variability in external factors.
- Measurable or estimable: Ability to measure or estimate attributes will continue to grow with technology and methodological improvements (e.g., use of environmental DNA), but some may remain unmeasurable in either a strict or a practical sense (e.g.,

primary productivity by benthic microalgae, predation rates on pelagic fish) and will have to be approximated or bounded using proxies.

- Statistically sensitive: Changes in a measured or estimated attribute must be estimated in the context of often high spatial or temporal variation. For example, because delta smelt are caught in very low numbers in monitoring programs, the data do not allow for robust estimates of population trends.
- Readily interpretable and understandable: Esoteric or overly complex attributes, or those that do not link clearly to the goals, will fail to win support. Goals may be ambitious, but they should be realistic. If goals are not achievable, they will lead to cynicism.
- Based on scientific principles: Findings need to be based on the methods of science, but with enough flexibility to accommodate new scientific methods and understanding.
- Measured in existing monitoring programs: Metrics tracking goals should rely mainly on existing programs in the estuary. Monitoring in the rivers is spotty at best and may require augmentation with measurements of key structural and functional components.
- Multi-purpose: Attributes that track progress towards goals can serve more than one purpose and provide efficiency. Cost almost always constrains monitoring programs, and multi-purpose metrics allow greater information at reduced cost.
- Associated with milestones: Assessment should be made against a backdrop of temporal change. There should be clear expectations for the trajectory of the attributes, and checkpoints for evaluating and displaying progress and revising the program if necessary.

No one approach is best for all ecosystem types, and guidance can be found in well-regarded, long-term, successful projects regionally and worldwide (e.g., Boxes 2.1 and 2.2). Keys to success include engaging the best scientific expertise for the ecosystem under consideration and involving a broad suite of stakeholders in the planning, implementation, and evaluation process. Attributes that track progress towards goals are used to make inferences about the current status of the ecosystem, the biological populations therein, and the effectiveness of management actions intended to improve ecological status. Linkages between goals, objectives, metrics, and decision-making are discussed next.

1.7 Making Decisions Based on Biological Goals

In this section of the introduction, a brief review of common terms and approaches used in environmental decision-making with respect to explicit biological goals and how they contribute to the decision-making process is provided. A logical start to this discussion is defining differences among goals, objectives, and performance measures. Goals define a desired outcome in a very broad and general way, while objectives are more specific and measurable. Performance measures explicitly define the way in which progress towards objectives is tracked and assessed. For example, a goal for Chinook salmon in the CV is to increase their natural production. An objective, as defined by guiding policy documents, is to double natural production, relative to estimates during a baseline period, within three generations (USFWS 2001). Feasible performance measures for this objective could be annual estimates of naturally-produced Chinook salmon prior to any removals due to fisheries or annual estimates of naturally-produced spawner abundance. The Panel notes that the term "biological goals" used in the Board charge is by definition an objective. Thus, the term "biological goals" is used in this report in order to be consistent with the charge.

Decision tables (also known as consequence tables) are a useful and simple tool for summarizing how management actions or policy alternatives influence the objectives. Development of a decision table is a key element of Structured Decision-Making (SDM), which is an approach for analyzing problems to reach decisions (Martin et al. 2009). Decision tables consist of a series of rows that identify performance measures and columns that identify a series of policy alternatives or management actions (Table 1.1). In an information-rich setting, reliable models may be available to predict the response of some performance measures to policy alternatives. However, as we discuss below, this is rarely the case for biologically-based measures. The utility of each policy alternative can then be computed from the scores for a predicted performance measure in each cell of the decision table, the weight each decision-maker place on each performance measure, and the weight of each incremental change in their values.

SDM can be a particularly helpful procedure for organizing and clarifying objectives and performance measures. As an example, doubling naturally-produced Chinook salmon abundance in three generations is termed a "fundamental" objective because it is closely tied to the goal. A "means" objective is a way to achieve the fundamental objective. In this example, one means objective would be to "increase survival rates of juvenile salmon in tributary x". In this report, methods for defining performance measures for both fundamental and means objectives are identified. The discussion in Chapter 2 relates to means objectives needed to achieve fundamental abundance objectives for other native fishes (Chapter 3) and Chinook salmon and steelhead (Chapter 4). However, the ecosystem chapter also identifies potential metrics for some fundamental objectives on water quality and food webs.

It is more challenging to establish cause-effect relationships between management actions and performance measures that are tracking fundamental objectives than it is for measures tracking means objectives. For example, a means-based performance measure that tracks survival rate between two juvenile life stages of salmon will respond more quickly and detectably to management actions than a performance measure that tracks the abundance of naturally-produced adult salmon. In addition, it will be easier to establish cause-effect relationships for survival between two juvenile life stages because there will be fewer confounding factors to account for. As a result, performance measures for means objectives strengthen cause-effect inferences for fundamental objectives. For example, if juvenile fish abundance increases during the specific times of year when flows were purposefully increased, this will strengthen the inference for a cause-effect relationship between abundance and flow. The ultimate decision to adopt a policy may have to wait until the response of the fundamental abundance-based performance measure is directly observed because there is no guarantee that an increase in survival between two juvenile life stages will translate into a similar increase in adult abundance. However, means objectives and performance measures provide a rapid way of determining whether a policy is on track to potentially result in a positive response in a fundamental objective. Conversely, if juvenile salmon survival rates fail to increase following a management action, then a positive response of abundance to that action is unlikely. Such an outcome provides decision-makers with the opportunity to pursue more viable policy alternatives.

SDM also requires that decision-makers provide weights for each performance measure and define utility functions that describe how they value incremental changes in values of performance measures. For example, it is likely that a fisherman or representative from an environmental agency may attach more weight to salmon abundance as a performance measure, while a representative for agricultural interests would put more weight on a performance measure that describes how different flow regimes influence agricultural production or economics (Table 1.1). Within a performance measure, some stakeholders will consider that the utility for increasing salmon abundance may increase linearly with abundance, while others may indicate very little utility to increases in abundance until it reaches a level at which a target is achieved or exceeded (Figure 1.1). In this report, we make no comment on utility functions for objectives and performance measures because these functions are determined by the values of decision-makers and the mandates of the agencies they represent. However, we do comment on the scientific basis for some targets that decision-makers should consider when defining their utility functions for specific performance measures.

Trajectories of performance measures for both fundamental and means objectives, rather than attainment of targets or specific goals, are what is needed to evaluate the effectiveness of flow and habitat enhancement alternatives to enhance focal species. Consider the example where the abundance of naturally-produced Chinook salmon spawners in a tributary is steadily declining under current levels of flow, habitat, harvest rate, and hatchery practices (scenario line A in Figure 1.2). Then, consider a range of potential population trajectories to a specific set of management actions beginning in 2020. The effect of this policy alternative may be modest and result in only a slight reduction in the rate of population decline (B), stabilize the population near its 2020 abundance level (C), result in a slow increase in abundance (D), or produce a more rapid increase in abundance to the point where a target is exceeded (E). Assume also that means-based performance measures tracking survival rates also increase, so that decision-makers are confident that the population response in the treated tributary is due to the management actions. From a learning perspective, the experiment of increasing flow and habitat is successful under all scenarios because a population level response to the treatment was observed, even though the target abundance was achieved only under scenario E. Some decision-makers might consider scenarios D and E successful with respect to the broad goal of increasing naturally-produced salmon abundance. Those with a pessimistic view of the future of Chinook salmon populations in California might even consider outcomes B and C successful because they have at least bought some time to either find another solution or for environmental conditions to improve through natural variability. The decision to continue the management action in the long term depends on both the responses of fundamental performance measures to the action and stakeholder-specific utility functions and performance-measure weights. However, the effect of the action on the performance measure does not depend on the utility function and therefore does not depend on targets that may define that function.

Thus, in the Panel's view, it is not productive to spend lots of time agonizing about targets or very specific objectives in resource management decisions. The response of performance measures to management actions typically show strong trade-off relationships. For example, increasing abundance of naturally-produced salmon in a tributary through more flow will likely result in less water available to agriculture. Active adaptive management experiments should map out the trade-off relationships so that decision-makers can find a policy by which substantial gains in survival rates of juveniles can be obtained at reasonable costs to agriculture. Solutions are unlikely to be found if decision-makers fixate on attainment of specific targets. This is especially true if the science behind the specific targets is weak or the probability of achieving them is low.

Quantitative models can predict the responses of performance measures to various management actions by providing values for the cells in a decision table (Table 1.1). In some cases, the data are sufficient to make a reliable prediction, such as determining the economic consequences to agriculture resulting from reduced water allocation due to a specified increase in tributary flow. However, the reliability of models predicting most biological performance measures discussed in this report is limited. For example, there are a few quantitative models that project the responses of fish habitat and fish populations to changes in flow in CV mainstems and tributaries. Physical habitat modeling (PHABSIM) has been used to predict the effect of flow on habitat for various life stages of fall-run Chinook salmon (e.g., Aceituno [1993] in the Stanislaus River), but with limited success. Models predicting population responses are highly uncertain but do have heuristic value. For example, the effect of flow on the survival and abundance of winter-run Chinook salmon in the Sacramento River can be predicted using an existing life cycle model (Hendrix et al. 2017). Models predicting the response of the delta smelt population to water exports and Delta outflow are also available (Kimmerer 2008, 2011, Maunder and Deriso 2011, Rose et al. 2013, Newman et al. 2014, Kimmerer and Rose 2018). All these models provide at best ballpark predictions of habitat- and population-level responses to flow (Castleberry et al. 1996, Bradford et al. 2011). However, they do provide an organized framework for identifying key uncertainties that can be used to guide future research and monitoring and to develop a set of initial flow policies to test via adaptive management. Output from these models is much too uncertain to reliably populate the cells in a decision table (Table 1.1), which highlights the need for deliberate field experiments to reduce uncertainties and better quantify performance measure response to adaptive management actions.

Reliable models for some means objectives are emerging. Perry et al. (2018) used a highly calibrated model to estimate how flow in the Sacramento River affects the survival of hatchery-origin late-fall-run Chinook salmon smolts migrating through the Delta. This model could be used to predict a potentially useful means objective that tracks the response of survival rates of salmon smolts in the Delta to changes in flow. A model predicting incubation success of fall- and winter-run Chinook salmon in the Sacramento River as a function of water temperature (Martin et al. 2017) provides another example where a specific means objective (higher survival through incubation) could be reliably estimated. At a minimum, these models can be used to screen proposed policies and eliminate those too small in scale or that may have a negative effect.

Stakeholder Performance Measure Weights			Policy Alternatives					
Agency 1	Agency 2	Performance Measures (PMs)	Status Quo	Increase Flow	Habitat Restoration	Increased Flow & Habitat Restoration	Reduce Exploitation Rate	Reduce Hatchery Production
0.3	1	Abundance of Chinook salmon						
0.2	0.8	Abundance of Steelhead						
0.1	0.5	Abundance of other native fish						
1	1	Water quality		Predictions of performance measure response to each policy from models				
0.2	0.5	Salmon harvest						
1	0.2	Agricultural economics						
1	0.2	Municipal water availability						

Table 1.1. An example of a decision table developed in a Structured Decision-Making process. The table consists of a series of policy alternatives (columns beginning with Status Quo) and performance measures (rows, PMs). Quantitative models are typically used to predict the response of each PM to each alternative. The values of stakeholders and the mandates of their agencies determine the relative importance (weight) of each performance measure (1= most important, 0=least important). Stakeholder-specific utility functions (see Figure 1.1) also need to be defined to quantify the relative benefit of incremental change in each performance measure. This information is integrated to compute the relative utility of each policy for each alternative. This report provides recommendations on approaches for biological objectives that are relevant only to the performance measure examples in the first four rows.

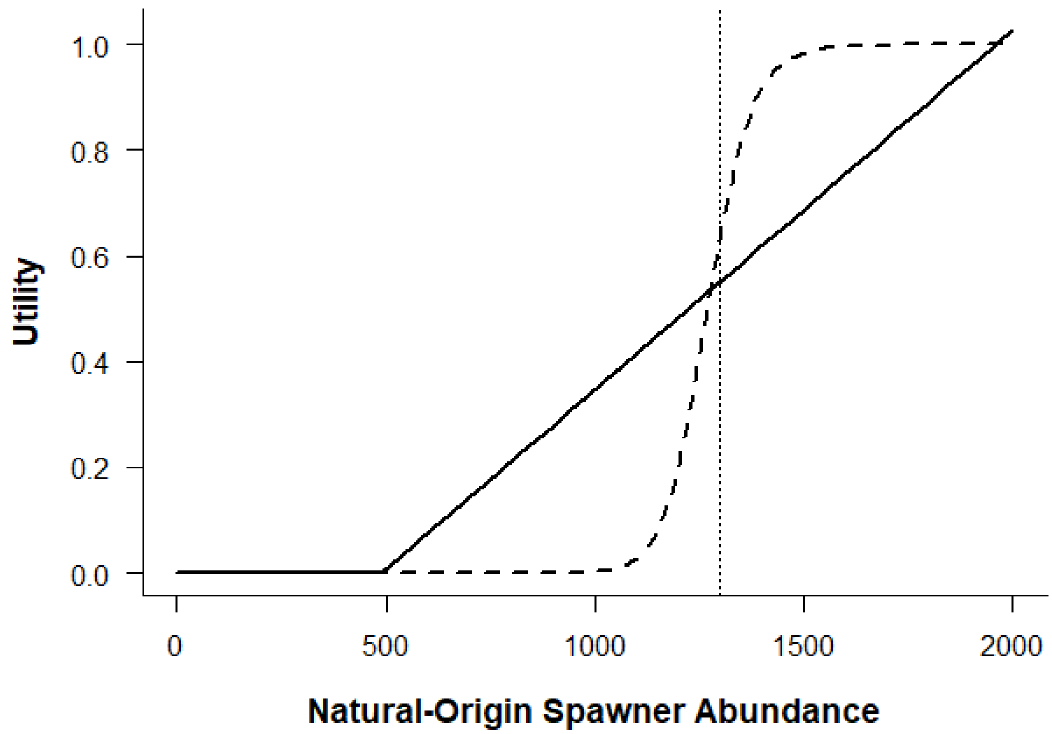


Figure 1.1. An example of utility functions that define the preferences of two different stakeholders to a range of natural-origin Chinook Salmon spawner abundance in a tributary, where the current abundance in the absence of any flow and habitat changes is 500 fish. The solid line represents a stakeholder who has a linear utility function at abundances greater than the current abundance. That is, their preference increases proportionally with increases in abundance. The thick dashed line represents a target-focused stakeholder who assigns little value to abundance until it gets close to a target abundance of 1,300 fish (dotted thin vertical line) and assigns little additional value after it exceeds the target.

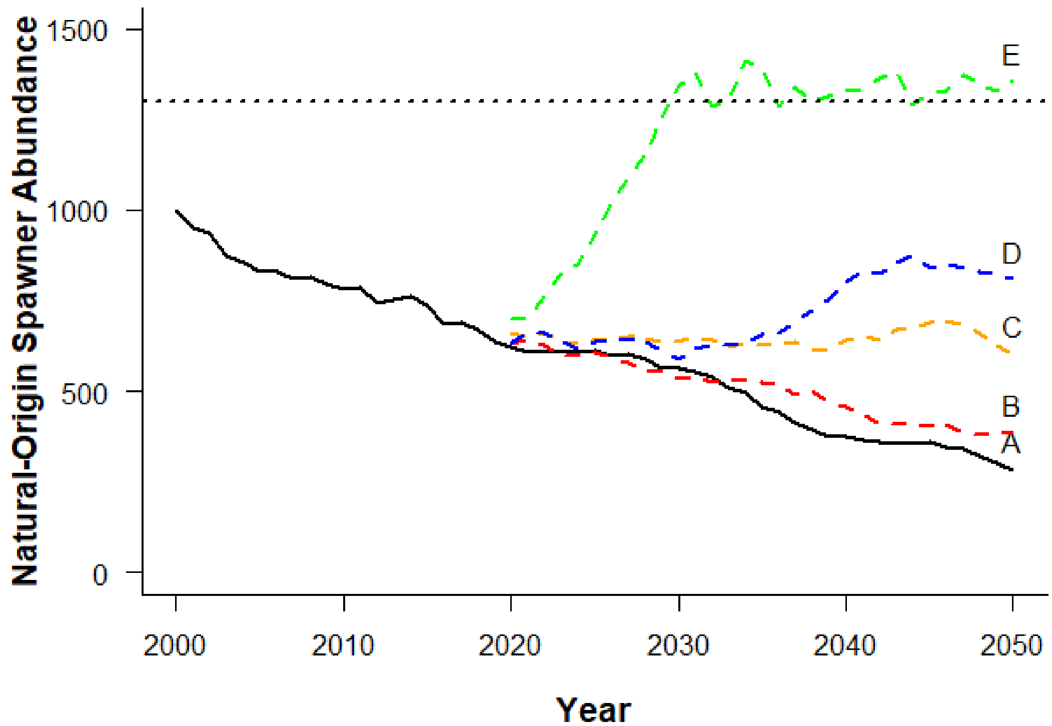


Figure 1.2. Examples of the trajectory of the natural-origin spawner abundance for a Chinook salmon population in a tributary assuming no change in flow and habitat (A), and alternate responses to increased flow and habitat (B-E).

1.8 Report Road Map

This introduction first laid out the background on why the Panel was requested to write a report on approaches for setting quantitative biological goals for the SFE and watershed. A brief review of the charge questions was followed with the concept and evidence for viewing this system as a novel ecosystem much influenced by human alteration, non-native species, and a changing climate and hydrology. Experimental management is an approach for natural resource management through using experimental design coupled with monitoring and evaluation to learn how best to manage complex novel ecosystems. The introduction ends with some general principles and criteria for developing usable goals and a brief discussion on how to make management decisions based on quantitative biological goals.

The core of the report is three chapters focused on setting goals for the estuary and the watershed (Chapter 2), native fishes and fish assemblages (Chapter 3), and the salmonids found throughout the estuary and tributary rivers (Chapter 4). The chapters are not formatted identically, but all chapters are designed to address the charge questions presented to the Panel by the Board. After the three primary chapters, there is a short conclusion section followed by acknowledgements, a glossary and an extensive references section. Finally, there is a series of appendices going into more detail concerning some of the points made in the main chapters. The appendices include 1) brief answers to all the charge questions asked of the Panel; 2) answers to additional salmon questions provided by the Board; 3) tables of native fishes and fish assemblages; 4) an example brood table for salmon and steelhead; and, 5) an example salmon stock recruit covariate model for the Stanislaus River.

2 ECOSYSTEM STRUCTURE AND FUNCTION

2.1 Introduction

This chapter addresses setting goals for the structure and function of a variety of aquatic ecosystem types that support viable fish populations and communities in the SFE and watershed. The conditions within these aquatic ecosystems need to provide diverse fish habitat with crucial attributes such as functional flows, productive food webs, and good water quality. This chapter emphasizes developing monitoring and evaluation programs to examine the responses of these ecosystems to manipulations of freshwater flow and habitat.

The estuary and rivers hold some features in common. Both are dynamic, variable systems that have been greatly altered from their original state, particularly through removal of connections to adjacent shallow habitat (floodplains and wetlands), channelizing watercourses, urban and agricultural influences such as diversions and discharges, and introduction of numerous non-native species. Both rivers and the estuary are influenced by freshwater flow, though this influence plays out differently in the estuary where tides and salinity play a major role. Finally, climate variability has a strong influence on both systems, with a distinctive hydrology driven by a Mediterranean climate and snowpack in the Sierra Nevada.

The estuary and the river ecosystems also differ in some salient ways. The estuary has a long history of ecological research and systematic monitoring programs, some of which extend back 60 years. The river ecosystems have various degrees of monitoring data but lack a long-term, comprehensive program of systematic ecosystem monitoring and evaluation. A much more diverse, if not complete, suite of aquatic organisms has been monitored in the estuary than in the rivers. Resource management and compliance (SWRCB 2000) requirements in the estuary have ensured stable funding for monitoring programs, data management, and scientific publication, while similar efforts for the tributaries are spottier and more discontinuous in time and space. Finally, the theoretical basis for understanding riverine dynamics is better developed than that for estuaries, which vary widely in morphology and river and tidal flows, precluding broad generalization. This makes methodologies for setting biological goals from successful river ecosystem programs elsewhere more applicable than those from other estuaries.

In this chapter, some key general characteristics of the physical, chemical, and biological environments of rivers and estuaries are examined. This chapter begins with a broad discussion of the principles that govern rivers (Section 2.2) and then estuaries (Section

2.3). Next, the chapter focus on effects of freshwater flow and habitat for fishes (Section 2.4). These sections emphasize the ecological productivity of these systems and the habitat they provide for fishes, including food and water quality. Lastly, specific topics for developing goals for pelagic, estuarine nearshore, and riverine ecosystems are discussed (Section 2.5). These topics include abiotic habitat such as landforms and chemical constituents, introduced species, food webs, and predatory environments. This chapter closes with an annotated list of recommended attributes or categories to be considered for measurement to assess ecosystem changes that impact fish through their response to manipulations of flow and habitat (Section 2.6).

2.2 River-specific Considerations

The concepts of structure and function in streams and rivers have a long history of development and application that guide research and monitoring to this day. Cummins (1974) pointed out that studies of streams and rivers focusing solely on taxonomic inventories of biological communities failed to answer key functional and process-oriented questions that define major attributes of streams and rivers. Key processes such as decomposition and primary production define distinctive types of stream and river ecosystems, and strongly influence the structure of the biological communities therein. Hynes (1975) described how the catchment (watershed) rules the river. Structure and function in streams and rivers are intimately linked to the geology of the region that defines the chemistry of the water, the soil types both upslope and in the riparian zone, the dominant vegetation that contributes organic matter to the waterway, the geomorphology of the channels and the connectivity between the flowing water and the floodplain, and the climate that shapes the river hydrograph. Human activity has substantial effects on many of these structuring elements, and function commonly follows structure. These abiotic components of rivers should be characterized and monitored as part of the effort to develop biological goals.

Stream and river research in the 1990s emphasized the functional attributes of flow that could be described using statistics on flow magnitude, duration, frequency, timing, and rate of change. Richter et al. (1996) showed that hydrologic regimes play a major role in determining the biotic composition, structure, and function of rivers and streams. They presented a method (Indicators of Hydrologic Alteration-IHA) to statistically characterize hydrologic variation within each year and before and after major changes to flow (dams, diversions, groundwater pumping, or intensive land-use conversion). Poff et al. (1997) presented a paradigm for river conservation and restoration based on the natural flow regime. They argued that restoring specific components of the flow regime would benefit the entire ecosystem and that restoring some semblance of natural flows should be a cornerstone of management approaches to river ecosystems. Bunn and Arthington (2002) laid out four basic principles and ecological consequences of altered flow regimes for aquatic biodiversity in rivers: 1) flow is a major determinant of physical habitat and thereby biotic composition; 2) aquatic species have evolved life history strategies in direct response to the natural flow regime; 3) maintenance of natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of many riverine species; and, 4) invasion of rivers by alien species is facilitated by altered flow regimes. River flow regimes are the organizing variables for the structure and function of river ecosystems, and flow-related goals are therefore an integral part of a well-designed evaluation program.

Setting biological and non-biological goals for river restoration projects has garnered increased attention as restoration has been embraced in many regions worldwide. For example, billions of dollars are currently being spent annually on river restoration in the United States (Bernhardt et al. 2005). The most commonly stated goals for river restoration in the United States are: 1) to enhance water quality; 2) to manage riparian zones; 3) to improve in-stream habitat for fish; 4) to allow fish passage; and, 5) to stabilize banks (Bernhardt et al. 2005). Based on a study of over 37,000 river restoration projects in the United States, this synthesis showed that fewer than 10% of the restoration projects had any assessment or monitoring linked to the project.

Restoration efforts in the tributary rivers to the Delta require rigorous evaluation with clear goals. Palmer et al. (2005) described standards for ecological river restoration with five criteria for ecological success: 1) a guiding vision provides a dynamic ecological endpoint that is used to guide restoration; 2) the ecological conditions of the river are measurably enhanced; 3) the river ecosystem is more self-sustaining than prior to restoration; 4) implementing the restoration does not inflict irreparable harm, and 5) some level of both pre- and post-project assessment is conducted, and the information made available. River restoration, whether explicitly linked to flow or not, needs rigorous evaluation as to effectiveness and whether specific quantitative goals are met.

Recently, the concept of functional flows has been discussed for managing heavily modified rivers (Yarnell et al. 2015). The functional flows approach is a strategy for allocating ecosystem water budgets by capturing crucial processes upon which native aquatic species depend. Highly modified rivers include those that have much of their length channelized, lined by levees, or converted to reservoirs, have much of their annual flow stored in reservoirs, or have a high proportion of their total flow diverted for human use. All major tributaries to the Delta are highly modified. A functional flow is a component of the hydrograph that provides a distinct geomorphic or ecological function (Escobar-Arias and Pasternak 2010). Rather than the entire natural flow regime, an approach using functional flows provides one basis for defining how much water is needed for the environment by targeting components of the natural flow regime that are thought to be most important for key processes. Functional flows might target key geomorphic processes, biogeochemical processes, or ecological processes. Such an approach has potential utility in managing California rivers to improve conditions for native fish species and can be linked to a well-formulated adaptive management program.

The considerations presented for rivers are based on the growing understanding of the structure and function of river ecosystems provide important guidance when establishing meaningful biological and abiotic goals for the highly regulated CV rivers. Some key examples include capturing the concept of river structure and function, focusing attention on interactions of flow with biological species of concern, designing and supporting a long-term evaluation program based on clear goals, and allowing more functional flows for key riverine processes. River-specific considerations can improve the outcomes of monitoring and evaluation programs. An excellent and successful long-term example of such an approach is the Kissimmee River Restoration Evaluation Program in south Florida (Box 2.1).

Box 2.1 Defining Success: Expectations for Restoration of the Kissimmee River in South Florida

The Kissimmee River is the major tributary of Lake Okeechobee, which then provides much of the water supply for the Everglades. The Kissimmee River was channelized in the 1960s for flood control. The channelization had pronounced environmental impacts on the river, the floodplain, and the extensive wetlands in this low gradient system. The biological consequences of the channelization that particularly drove public awareness of the damage included major declines in waterfowl, wading birds, and sport fish. Comparing the channelized river to baseline conditions before channelization showed major changes to the hydrology, geomorphology, water quality, dissolved oxygen, algae, littoral vegetation, floodplain plants and extent, aquatic invertebrates, herpetofauna, fish, and birds. Realization of what had been lost led to the largest and most expensive river restoration program in the United States at over a billion dollars invested.

An integral component of the Kissimmee River Restoration Project was the development of an evaluation program to define success for the restoration project. The development of the Kissimmee River Restoration Evaluation Program (KRREP) began in 1992 well before the start of the restoration construction in the fall of 2000. The first four years of the development of KRREP included pulling together baseline information on the pre-channelized ecosystem, developing and refining biological and abiological goals (expectations) for the restoration project, and securing long-term funding to support KRREP and matching resources with monitoring priorities. An important product from the development phase of the evaluation program was a dedicated issue of the peer-reviewed journal, *Restoration Ecology,* in 1995 that presented the expectations for the restoration project in advance of construction and the start of effectiveness monitoring. Overall, this approach and timeline allowed KRREP to gather multiple years of background monitoring data before phase one of the restoration began using a well vetted, reviewed, and agreed upon program. Now, more than 20 years after the start of KRREP, a rigorous and long-term evaluation program provides valuable information to the public, to decision makers, and to politicians on this major public investment in river restoration.

Developing the expectations (analogous to biological goals but including physical and chemical goals) followed a well-defined process (Kissimmee River Restoration Studies 2006 and Dahm et al. 1995). The overall goal of the restoration projects is to create ecological integrity as defined by endpoints as measured by metrics that can be compared to baseline conditions and/or reference conditions. This process or sequence of steps is used for developing restoration expectations that are quantitative and can be expressed as the difference the baseline condition and the reference condition. A mechanism is proposed that outlines conceptually how the restoration project will lead to achievement of the expectation. In addition, a trajectory or appropriate timeframe is identified for achieving desired responses. Initially, KKREP set 61 restoration expectations in 1999. This set of expectations has been shortened to 25 expectations using external and internal peer review and deleting expectations that lacked adequate reference data. These 25 goals or expectations make up the current evaluation program that has been in place for two decades.

The 25 expectations in KRREP fall into eight categories. Five expectations involve the hydrology of the river and include eliminating zero discharge days, monthly mean flows that reflect historical seasonal patterns, more natural stage fluctuations, an annual prolonged recession period between the wet and dry season, and a mean range of main channel river velocities throughout 85% of the year. Two expectations each deal with geomorphology and water quality with the focus on bed deposits in restored channels, point bars on the inside bends of river channels, dissolved oxygen concentrations during the wet and dry seasons, and mean turbidity and suspended solids not to exceed levels. Vegetation has five expectations that focus on littoral vegetation distribution, littoral plant community structure, wetland plant cover, broadleaf marsh coverage, and wet prairie community coverage. Four expectations are used to assess the condition of the aquatic invertebrate community including the composition of the drift, the density and biomass of the passive filtering and collecting guild, the richness and species diversity of aquatic invertebrates in restored broadleaf marsh, and the fauna of the river channel bottom being primarily associated with sandy substrates. Two expectations each evaluate amphibians and reptiles and birds. The expectations for amphibians and reptiles focus on restored pasture land converted to broadleaf marsh habitats, and the expectations for birds focus on the density of wading birds on the restored floodplain and the winter densities and species richness of waterfowl within restored floodplains. Finally, the three fish metrics target the annual densities of small fishes (< 10 cm) within restored marsh habitats, the mean annual relative abundance of bowfin and Florida gar relative to redbreast sunfish and centrarchids, and off-channel fish assemblage composition in restored floodplain habitats that is species diverse and has >30% young-of-the year or juveniles.

Assessment of project success considers all the expectations (goals). Collectively, the goals describe the state of the Kissimmee River and floodplain over time after a major multi-phased restoration program. These goals provide feedback for adaptive management by determining whether attributes are recovering as predicted within the specified appropriate timeframes. If not, this triggers additional analysis and experimentation about why an undesired response is occurring. This can lead to modifications to management actions such as fine tuning of flow regimes or reevaluation of expectations. This comprehensive restoration evaluation program of a billion-dollar restoration effort has many attributes that might guide development of structural and function goals (biological, physical, and chemical) for major rivers that flow into the California Delta.


Performance Measures and Restoring the Kissimmee River:
A Successful Large-scale Restoration Project
Anatomy of the Phase I Backfilling
February 9, 2001
Initially 60 Performance Measures
(Expectations) Now Coalesced to 25
• Hydrology - 6
• Geomorphology - 2
• Water Quality - 4
• Vegetation - 10
• Invertebrates - 11
• Algae - 2
• Amphibians - 2
• Fish - 7
• Birds - 11
• Listed Species - 5
Performance measures are linked to an experimental design, locations and frequency of measurements, methods to be used, and ways to analyze and report the resulting data.
Executive Summary of the performance measures used for this project.

CV rivers, most regulated by dams, have flow regimes negotiated as part of Federal Energy Regulatory Commission (FERC) relicensing agreements or other settlements with fisheries agencies. State law *requires* flows that keep fish “in good condition” below dams. However, most of the flow regimes were designed using standardized instream flow methodologies for fall-run Chinook salmon that are not particularly reliable (Williams et al. in press). Incidentally, the designed flows benefitted an array of other native fishes as well; this assemblage of native species is useful for monitoring, even when salmon numbers are not because of uncertainty about hatchery contributions (Chapter 4). Riparian plants and birds as well as native aquatic invertebrates may also benefit from flow releases for salmon.

2.3 Estuary-specific Considerations

Variation in freshwater flow induces a wide variety of physical, chemical, and biological responses in estuaries that are highly dependent on specific attributes of each estuary (Drinkwater and Frank 1994, Kimmerer 2002). Many mechanisms have been suggested for positive (and some negative) effects of high flow (Rose and Summers 1992). These include, for example: 1) increases in loading of nutrients or organic matter into estuaries, stimulating primary and secondary production (the "agricultural model", Nixon et al. 1986); 2) enhanced salinity stratification resulting in phytoplankton blooms (Cloern 1991); 3) suppression of marine organisms including bivalve grazers (Nichols 1985) and predators (Reaugh et al. 2007); and, 4) seaward movement of the salinity field, altering the interaction between salinity and shallow habitat (Jassby et al. 1995, Sommer et al. 1997, Paerl et al. 2014) and patterns of retention of organisms (Kimmerer et al. 2014). These diverse mechanisms, and their positive and negative responses have outcomes that depend on interactions among morphology, ranges of freshwater and tidal flows, sediment inputs, and species composition.

In the SFE, physical responses to freshwater flow are governed mainly by its irregular morphology, with deep channels and extensive shoals, its strong tidal flows, and the large variation in freshwater flow. In tidal freshwater reaches of the estuary (most in the Delta), tidal flows can be much stronger than net (riverine) flows, and both contribute to the movements of substances and organisms. For our purposes, key concerns regarding net flows in the Delta are the transport of fish to the export pumps in the south Delta (Grimaldo et al. 2009a) and diversion of migrating fish off the most direct pathway to the ocean (Kjelson and Brandes 1989, Cavallo et al. 2015). An underappreciated aspect of flows in the Delta is losses of key foodweb organisms due to flow to the export pumps: ~18% of daily phytoplankton productivity estimated for 1975–1993 (Jassby et al. 2002) and ~7%/day of copepods during summer in 1995–2012 (Kimmerer et al. 2019).

Seaward of salinity of ~2, most effects of changing Delta outflow occur indirectly through the compression and movement of the salinity field over deeper water, and resultant changes in stratification and density-driven circulation. The effects of large changes in flow remain strong (Cloern 1991). However, X2 is an accelerating function of flow; for example, moving the salinity gradient downstream from river kilometer 85 (~ Sherman Island) to 75 km (Chipps Island) requires a 72% increase in outflow (i.e., an increase of 137 m^3/s or 4,800 cfs). In contrast, moving the salinity gradient the same distance from 65 km (Roe Island) to 55 km (Martinez) requires a doubling of flow (an increase of 650 m^3/s or 23,000 cfs) (steady-outflow case in MacWilliams et al. 2016).

The direct and indirect effects of varying freshwater flow on the abundance of organisms, particularly fish, in the estuary are key for investigation (Jassby et al. 1995) and the basis for regulation of estuarine flow. Biomass and productivity of phytoplankton in the upper estuary are unresponsive to freshwater flow (Kimmerer et al. 2012). Likewise, abundance, growth, and reproductive rates of ecologically important zooplankton in the brackish to tidal fresh waters of the estuary do not respond to freshwater flow (Kimmerer 2002, Kimmerer et al. 2017). High flow may play a role in subsidizing the Low-Salinity Zone with phytoplankton and zooplankton from fresh water (Kimmerer and Thompson 2014, Kimmerer et al. 2019). Otherwise, there is no evidence that the numerous flow effects on fishes and Bay shrimp (Jassby et al. 1995) occur via transfer of energy up through the planktonic food web, so the "agricultural model" (Nixon et al. 1986) seems to be ruled out (Kimmerer 2002).

The mechanisms for flow responses are more likely related to changes in physical habitat for fish. The most prominent example is that of Sacramento splittail (see Chapter 3), whose abundance increases sharply following periods of flooding on the Yolo Bypass and elsewhere, probably because of the expansion of shallow habitat for spawning, rearing, and foraging (Sommer et al. 1997, Moyle et al. 2004). Mechanisms for relationships of other fishes to flow may include transport of young fish from spawning to rearing habitats (Chadwick et al. 1977, Grimaldo et al. 2017), area or volume of physical habitat (Kimmerer et al. 2009, 2013, MacWilliams et al. 2016), or mechanisms for physical retention of fish and other organisms in the estuary (Kimmerer et al. 2014).

Spring flow standards have been set across a broad temporal scope, with the current outflow or X2 standards lasting from February to June. Although this standard is a blunt instrument for protection of individual species, it was selected to cover times when several different species passed through their early life stages when they are likely most responsive to flow effects (Jassby et al. 1995). Although flow standards that focused on individual species could be made more efficient, the existing broad flow standards are consistent with an ecosystem rather than a single-species perspective. However, for declining fishes such as Delta and longfin smelt, abundance is now too low for statistically robust evaluation of these mechanisms.

2.4 Habitat Restoration

Restoration[3] of wetlands, floodplains, and streams provides a variety of ecosystem benefits (Costanza et al. 1997, Zedler and Kercher 2005, San Francisco Bay Area Wetlands Ecosystem Goals Project 2015, Opperman et al. 2017). Throughout the watershed and estuary, restoration projects of a variety of forms and sizes are underway in efforts to reduce some of the harm inflicted by past practices; a few examples are presented below. Some of these projects are motivated by clear opportunities, such as the removal of dams to allow migrating salmon to gain access to former spawning habitat, while others have broader goals. These projects are using past history as a guide to future conditions, but all of these projects are creating something new. This implies high uncertainty in best practices and anticipated outcomes of restoration, which in turn motivates calls for adaptive or experimental management (Walters 1997, see Section 1.4). These calls were amplified by the CALFED program and later the Delta Science Program and California EcoRestore[4], and many of the larger projects have incorporated at least the structure and terminology of adaptive management in their plans.

Many of the restoration projects in CV rivers are motivated and supported by local watershed groups that have partnered with agencies and stakeholders to plan and carry out the projects. One such example is the Battle Creek Salmon and Steelhead Restoration Project designed to reopen spawning habitat for winter Chinook above the Coleman Fish Hatchery (ICF 2016). Another is in lower Clear Creek where spawning and rearing habitat was improved by removal of a low-head dam and extensive gravel augmentation (Railsback et al. 2013).

Floodplain restoration offers numerous opportunities for improving fish habitat in the CV (Opperman et al. 2017). The Yolo Bypass is particularly notable for supporting agriculture, flood protection, and fish habitat (Sommer et al. 2001). Small- and mid-scale experiments have shown the potential value of more frequent and/or persistent flooding for splittail (Sommer et al. 2002) and salmon (Katz et al. 2013).

[3] We use "restoration" here to mean construction, rehabilitation, or otherwise production of landforms that can provide habitat for aquatic organisms.
[4] http://resources.ca.gov/ecorestore/

EcoRestore proposes to support restoration of 30,000 acres of "habitat" in the Delta and Suisun Marsh, of which about 9,500 acres will be tidal wetland. While wetland restoration will eventually provide a variety of benefits that make it worth doing, numerous challenges impede progress (Callaway et al. 2007, Nagarkar and Raulund-Rasmussen 2016). Moreover, while the total area to be restored is large, the scale of the proposed restoration is still only ~1% of the original area of tidal wetland in the Delta (Whipple et al. 2012). Although a large and valuable effort at the local scale, it seems too small to recover the important role at the Delta scale that historical wetlands likely had in ecosystem structure and function at the scale of the broader estuary.

One of the motivations for wetland restoration is to provide habitat to support fishes such as delta smelt. Wetland plants are highly productive, and that productivity fuels a food web based on plants, epiphytes, and detritus that is distinct from the phytoplankton-based pelagic food web (Grimaldo et al. 2009b, Howe and Simenstad 2011). Moreover, most estuarine fishes, including salmon and delta smelt, use these habitats (Sommer et al. 2001, Grimaldo et al. 2009b, Sommer and Mejia 2013), presumably benefiting from their high productivity.

Another possibility is that wetlands may produce excess zooplankton, exporting the excess to the open waters where the bulk of the delta smelt population lives. This possibility was presented by the Bay-Delta Conservation Plan (now divided into WaterFix and EcoRestore) as a major impetus for restoring tidal wetlands, but the limited evidence does not support this contention (Mount et al. 2014, Kimmerer et al. 2018). However, Hammock et al. (2019) found that stomach fullness of delta smelt was highest in individuals captured in close proximity to tidal wetlands, although they could not determine if the feeding was the result of food (mainly larval herring) being exported from the wetlands or the smelt entering the wetland to forage for short periods.

The outcomes of restoration depend heavily on particulars of location and design; hence, the realized benefits of restoration can be ascertained only through assessment. Yet, assessment appears to be a major part of only a few wetland restoration projects. Even EcoRestore, which presents adaptive management as a core element of its restoration programs, appears to consider this an element of implementation, rather than a key aspect of project design. Furthermore, adaptive management in the Delta and elsewhere in the estuary is rarely incorporated fully into project or program designs and implementation, and the very concept of adaptive management seems to be unclear to some practitioners (Nagarkar and Raulund-Rasmussen 2016).

Two large restoration projects in the upper estuary include efforts to incorporate assessment into their designs and practices. The Dutch Slough restoration project in the western Delta is intended to restore 1,187 acres of tidal wetland using an experimental framework in which the area will be divided into plots to test effects of plot elevation and size on organic matter production and growth and survival of salmon and Sacramento splittail. Planning has been in progress since 2003, grading began in 2018, and the first breach of existing levees is expected in 2021. This project's adaptive management plan (Herbold 2016) has been extensively reviewed. The Montezuma Wetlands Restoration Project[5] in the southeastern corner of Suisun Marsh (a small fraction is in the Delta) is intended to restore about 1,820 acres of wetlands and buffer zones using dredged sediment. This project is in progress, has produced a monitoring plan and several annual monitoring reports, and has an active Technical Review Team (TRT) headed by San Francisco Estuary Institute scientists. The status of monitoring, assessment, and response to findings is not presented in the currently available documents.

2.5 Developing Goals for the Estuarine and Riverine Ecosystems

Societal interest in ecosystem structure and function focuses mainly on the provision of ecosystem services, notably to support the fish populations and communities discussed in the next two chapters. Thus, the Panel recommends goals that may contribute to supporting the fish and not those that might contribute to intrinsic value of the ecosystems. From this perspective, ecosystem attributes can be separated into two categories: 1) abiotic attributes of water quality and physical habitat for fishes and other organisms; and 2) attributes that contribute to biotic components of habitat including introduced species, food, shelter, and risk of predation. Here, the focus is on these categories, first emphasizing direct ecosystem support for fish habitat. Next are considerations of additional ecosystem attributes necessary for the ecosystem to support fish, particularly through food supply.

2.5.1 Physical aspects of fish habitat

The spatial and temporal configuration of the landscape can have a profound effect on the biotic components of ecosystems by providing a diversity of physical characteristics including depths, flow velocities, sediment characteristics, disturbance, and mixing and by connecting channels with shallows. In a relatively undisturbed system this configuration is dynamic, influenced by flow through sediment deposition and erosion

[5] https://ecoatlas.org/regions/ecoregion/bay-delta/projects/1062

and establishment or destruction of patches of terrestrial vegetation, which in turn affect flow patterns. Major changes in flow and physical configuration are required to improve abiotic and biotic habitat for fishes at a meaningful scale.

In rivers, high flow strongly influences spatial and temporal patterns of habitat attributes such as by accelerating river currents, scouring and suspending fine sediments thereby increasing turbidity, depositing sediments in low-flow areas, and improving spawning habitat for salmon. In the estuary, river and tidal flows interact to set the distribution of salinity, which determines species distributions partly through their salinity tolerance but also through other characteristics of physical habitat such as depth and water velocity and by the biotic components of habitat including food and predators.

Diversions of freshwater for human uses throughout the system remove organisms with the water, essentially imposing additional mortality. This is a central topic in controversies over the large diversions in the southern Delta (Kimmerer 2008, 2011, Miller 2011). Although these concerns focus on fish, plankton are also affected, and plans for flow and other alterations should take these losses into account. If California WaterFix is implemented, organism removal amounts will change, but the Panel is aware of no scientific studies predicting what these changes will be. Similarly, numerous smaller diversions, mostly unscreened, operate throughout the system and their impacts on the ecosystem are unknown (Moyle and Israel 2005).

Temperature is a fundamental control variable for all living systems. Several fish species of concern in this ecosystem are subject to thermal stress, notably winter run Chinook salmon and delta smelt. Increasing temperature not only increases the frequency and extent of thermal stress, it is also resulting in a decrease in mountain snowpack and earlier melting (Roos 1989, Stewart et al. 2005, Berg and Hall 2017), which will limit the ability to store cool water behind dams for release when winter Chinook are spawning and rearing. The cool tail waters below dams also provide habitat for a variety of native fishes, and effectively managing the cold-water pool also is critical for these fishes (Chapter 3).

Restoring or creating habitat for fish and other organisms may be constrained by sediment supply. Dams cut off sediment movement, and as sediments in the rivers are moved downstream by flood flows, spawning habitat for salmon below dams can sometimes be maintained only through placement of gravel. The rivers are also delivering much less sediment to the estuary than they once did, and the pulse of sediment from hydraulic mining has been winnowed out of the estuary (Schoellhamer et al. 2013). This has led to concerns about the reduction in turbidity, an important attribute of habitat for pelagic fishes (Feyrer et al. 2007), and the availability of sediment for restoration and maintenance of wetlands and mudflats, leading in turn to calls for an estuary-wide sediment management plan (Baylands Ecosystem Habitat Goals Science Update 2015). An example of new methods for managing sediment is where dredged sediment is being used as fill for re-establishing wetlands, but this sediment supply also is also limited.

2.5.1.1 Goals for physical characteristics

Goals for flow could be set based on a proportion of unimpaired flow. Although this practice would provide some balance in water allocation between human uses and the environment, this approach may be inefficient because it is not tailored to the extant river channel or to ecosystem processes and because the availability of stored water can uncouple deliberate environmental flows from runoff in the watershed. The functional flow concept discussed above should instead be applied in the rivers, using knowledge specific to each river reach to shape and sculpt flows to best benefit biological outcomes (Yarnell et al. 2015).

Flow conditions in the estuary will continue to be set to limit salt intrusion to protect water supplies. Net Delta outflow and export flows also should continue to be controlled for environmental purposes, the former for its indirect influence on fish through salinity and the latter for its direct effects of imposing mortality. The X2 standard used to regulate spring outflows continues to be workable, although longfin smelt and other species have declined in relation to their previous levels at a given flow or X2 value. Better understanding of the mechanisms underlying these relationships may allow for more efficient allocation of water for this purpose. Mortality due to export pumping is poorly known and remains controversial. Mortality rates, however, could be investigated through experimental manipulations.

Restoration to expand physical habitat should take place in an experimental framework to assure that these projects maximize learning and mechanistic understanding of cause and effect. Such restoration in the rivers should be coupled with alterations in various aspects of the flow regime (e.g., magnitude, timing, duration, rate of change, and frequency) as flow and physical habitat are strongly intertwined. From the perspective of this report, goals for habitat expansion should include both the area to be restored (and to what state, e.g., tidal wetland or open water), and an assessment of the likely value of that expansion. The latter should be developed as part of an ongoing sequence of restoration projects with assessments, which should forecast and then assess the uses of the restored habitat by fish species of concern and the conditions within the habitat that are likely to either support or exclude such species in the long run. Exclusionary factors include the colonization of the habitat by invasive plants or clams, the formation of harmful algal blooms, and the establishment of predator hot-spots.

2.5.2 Chemical constituents

Limiting concentrations of mineral nutrients such as nitrogen and phosphorus can limit growth of microscopic and macroscopic plants. In many lakes, rivers, and estuaries, human activities cause excessive nutrient concentrations that in turn cause eutrophication, a systemic condition marked by excessive algal and plant growth and low dissolved oxygen (DO). Many aquatic species are sensitive to low DO, particularly cold-water fishes such as salmon and benthic invertebrates.

Despite high nutrient discharge into the estuary, mainly from municipal wastewater treatment plants and agriculture (Jassby 2005, Novick et al. 2015), eutrophication in most of the estuary has been limited by high turbidity and by high grazing from bivalves (Brown et al. 2016). The principal exception is the shipping channel on the lower San Joaquin River near Stockton, which suffers chronic low dissolved oxygen because of poor circulation, high nutrient loading, and high levels of ecosystem respiration driven by outputs from the Stockton wastewater treatment plant and runoff from cities and farms in the San Joaquin Valley (Jassby and Van Nieuwenhuyse 2005). The Delta ecosystem may respond to high nutrient concentrations through increasing blooms of toxic cyanobacteria (Lehman et al. 2013) and the spread and increasing coverage of aquatic vegetation, mostly non-native (Ta et al. 2017).

Contaminants include hundreds of compounds from a variety of chemical classes with a variety of sources and effects. No part of the river-estuarine system is free from at least some of these effects, which can be widespread (e.g., mercury contamination in sources of drinking water and in organisms, Domagalski 2001), spatially and temporally variable (e.g., histopathological evidence of contaminant effects in delta smelt, Hammock et al. 2015), and insidious (e.g., low concentrations of pyrethroid insecticides from non-point sources had measurable toxic effects in bioassays, Weston et al. 2015). A thorough treatment of these effects is well beyond the scope. However, contaminant effects can alter the outcomes of management actions without being detected. Therefore, the potential for these effects must be considered and efforts made to resolve causes of unexplained variation in abundance or condition of biological populations of interest.

2.5.2.1 Goals for chemical constituents

Neither nutrient concentrations nor contaminant inputs suggest goals that meet most of the criteria enumerated in Chapter 1 (Section 1.5). For one thing, actions reasonably achievable by the Board lack a direct link to both nutrients and contaminants. Also, any changes in nutrient concentrations and chemical form due to Board actions would pale in comparison to the likely effects of the impending upgrade to the Sacramento Regional County Sanitation District wastewater treatment plant due for completion in 2021. Contaminants are under the aegis of the Regional Boards, but many of the contaminant problems are non-point-source or legacy, and they involve hundreds of compounds, all of which precludes a linear path between a goal (e.g., to reduce occurrences of toxic effects) and actions to achieve that goal.

2.5.3 Non-native species

Some organismal groups in the SFE and watershed consist largely of non-native (introduced) species. This is true of the fishes from the warmer reaches of rivers (Chapter 3) to the more saline waters of the lower estuary; the dominance of non-native species is also true of the aquatic plants of the Delta and rivers, the zooplankton of the upper estuary, and the benthos throughout the estuary. These taxa form what appear (over our limited time scale) to be persistent non-native assemblages. However, the principal reasons to distinguish native from <u>established</u> non-native species are cultural, reflecting society's interest in conserving native species. For ecological purposes these assemblages can be considered in terms of their species-specific traits without the need to distinguish between native and non-native species.

By contrast, introduction events can have unforeseen and potentially disastrous consequences, as with introduction of the overbite clam, *Potamocorbula amurensis* (Alpine and Cloern 1992, Kimmerer and Thompson 2014, Kimmerer et al. 2019). The rate of non-native species introductions appears to have abated, though risks remain. Any influential species introduced in the future could have a substantial impact on the ecosystem and how it responds to management actions. Although some introductions are likely and their impacts can be speculated upon (e.g., quagga and zebra mussels), it is difficult to anticipate what will happen if an influential, unanticipated species becomes established. The systems of management of the estuary and rivers need to be flexible and perceptive enough to adapt. For example, a future introduction could further alter the response of estuarine fishes to flow, in which case either biological targets for flow manipulation would need to be adjusted or the flows themselves adjusted to compensate.

2.5.3.1 Goals for non-native species

Given the risks discussed above, a goal should be to establish a well-defined program to detect, advertise, and possibly eradicate newly arrived (or threatening to arrive) species. The role of detection could be filled in part by existing programs of monitoring and inspection of facilities. This could be supplemented by public engagement encouraged through advertising and outreach, as is currently being done for invasive mussels and the recently arrived nutria.

2.5.4 The pelagic food web of the upper estuary

This food web has been monitored and studied for decades, largely to understand the food supply for fishes. A greatly simplified energy flow diagram (Figure 1) can be used to identify metrics of ecosystem structure and function; discussion for other food webs is expanded upon below. The figure shows a traditional food web with two additions: bacteria and microzooplankton. These important foodweb components are not monitored, but aspects of the other components have been monitored regularly since 1976 or earlier.

Direct foodweb support for planktivorous fish (most larval fishes in the estuary and some juvenile and adult fishes) comes from zooplankton, notably copepods, cladocerans, and mysid shrimp. Feeding by fish is generally positively related to the abundance or biomass of their prey, but varies with prey species; thus, key variables for assessing food availability are structural elements of abundance, biomass, and species composition of the zooplankton. The predominant source of energy for this food web is photosynthesis by phytoplankton (Sobczak et al. 2005), and the availability of this energy is seasonally variable and depends on size and species composition of the phytoplankton, which are therefore useful measures to monitor. However, planktivorous fishes are also abundant in shallow habitats where some of their energy supply may come partly through an aquatic plant- and detritus-based food web (Grimaldo et al. 2009b, Howe and Simenstad 2011).

Although the fish respond to the abundance and composition of their prey, the underlying productivity of both phytoplankton and zooplankton can be useful for tracking how the system overall responds to the environment. Phytoplankton productivity can be estimated with reasonable accuracy using a model relating productivity to chlorophyll concentration, light availability, and light penetration into the water (Cole and Cloern 1987), although the calibration constant should probably be determined periodically (Parker et al. 2012). Copepod productivity has been determined on a number of occasions as growth × biomass; it is generally low in the open waters of the upper estuary and higher in the Cache Slough area, particularly within the Yolo Bypass toe drain in summer (Kimmerer et al. 2018, Owens et al., in prep.). A rough idea of the growth rate of some species can be obtained from chlorophyll concentration, allowing for rough estimates of production. However, copepod productivity, perhaps estimated using biochemical methods (Yebra et al. 2017), could be used as a bioassay for assessing the value of their food resources. This can provide surprises, such as a recent finding of high copepod growth rates in the Yolo Bypass based on phytoplankton not normally considered nutritious (Owens et al. in prep.).

The conceptual model underlying the design of plankton monitoring includes phytoplankton and zooplankton, but the link between the two is rather weak. Microzooplankton such as ciliates play a key role in all pelagic food webs, consuming around half of the phytoplankton production as well as consuming bacteria and providing a substantial and highly nutritive food source for crustacean zooplankton such as copepods (Calbet and Landry 2004, Calbet and Saiz 2005, York et al. 2011). This suggests that at least a modest pilot program of analysis of these foodweb components be undertaken to determine whether the resulting data will help to understand the weak phytoplankton-zooplankton trophic link. This would help to understand why lower trophic levels of the estuary do not respond strongly to freshwater flows.

The benthos is included in Figure 1 mainly to represent losses of plankton to grazing by two species of clam. The Panel does not suggest that this consumption be part of the mix of variables to measure because of its cost and difficulty, but that the extant benthic monitoring program should be maintained to keep track of this loss term to both phytoplankton and zooplankton (Kimmerer et al. 2019). The benthos also should be monitored because benthic organisms, especially amphipods, are important prey for many fishes, such as tule perch, prickly sculpin, splittail, and various non-native fish species.

Although the above measures are generally intended to assess the availability of food for fish, direct measurements of the responses of the fish could complement the findings from more traditional studies. These could include diet studies using morphological and DNA-based identification, stable isotopes, growth rates determined from otoliths (Hobbs et al. 2007) and liver glycogen content (Hammock et al. 2015).

2.5.4.1 Goals for the pelagic food web

The pelagic food web supports native fishes but is not clearly responsive to either flow or habitat restoration. Therefore, goals for this food web would be designed to assess this support as a way of understanding the responses of fishes to management actions, rather than as a gauge for the success of actions themselves. These goals would assess phytoplankton primary productivity as the base of the food web, and zooplankton secondary production as a proximate measure of food available to fish. Primary productivity can be estimated from chlorophyll, light, and turbidity (Cole and Cloern 1987, Parker et al. 2012). Zooplankton productivity can be estimated from biomass (itself determined from abundance by species and life stage and standard biomass values for each of these; Kimmerer 2006) and an estimate of growth rate from temperature and chlorophyll concentration (Owens et al. in prep.).

2.5.5 Food webs of the estuarine margins

Shallow, nearshore areas such as tidal flats, wetlands, and levee margins provide important habitat for a variety of fishes, notably including salmon. In contrast to the pelagic food web, though, investigations and monitoring of food webs in estuarine margins including wetlands have begun quite recently. These regions are difficult to sample because of shallow, obstacle-filled water, extensive aquatic vegetation, and sharp spatial gradients in distributions of organisms. Nevertheless, these areas need more attention because they are under-studied but potentially important to ecosystem function, because of increasing efforts at tidal marsh restoration, and because of expectations that this restoration will benefit fish species including delta smelt and salmon.

Estuarine wetlands and other littoral or shallow areas are important habitat for a wide range of taxa such as wetland plants, migratory birds, and some fishes (e.g., Young et al. 2017, Mahardja et al. 2017). In less-modified estuaries, shallow areas and wetlands also are loci for important biogeochemical transformations. These biogeochemical processes are no doubt less important to the SFE simply based on diminished area.

The current surge of interest in shallow, peripheral habitats in the northern Delta and Suisun Marsh is partly stimulated by the realization that these regions are the most promising within the upper estuary for supporting native fishes including salmon (Chapters 3 and 4). Shallow tidal areas are somewhat isolated from the larger estuarine channels, and likely respond differently to changes in freshwater flow. For example, abundance of copepods in the Cache Slough Complex during summer did not respond to ~five-fold interannual variability in Delta outflow, in contrast to their abundance in the adjacent Sacramento River (Kimmerer et al. 2018).

The need to include measurements in these shallow water zones rests on the substantial use of these areas by smaller salmon that are more likely to be of natural origin (Miller et al. 2010, Chapter 3), as well as other fishes. However, much of the food of these fishes is benthic or associated with aquatic vegetation (Grimaldo et al. 2009b), making quantitative assessments of availability difficult. Therefore, the most useful measure of food availability may be through measurements on individual fish, as discussed in the previous section.

Widespread colonization of shallow areas by invasive waterweeds can limit the suitability of these areas for supporting native fishes. The plants occupy space, reduce current velocities, and provide habitat for many organisms preyed upon by fish, as well as many of the introduced predatory fishes such as largemouth bass (Conrad et al. 2016). Therefore, the area of submerged and floating aquatic vegetation should be examined through a regular program of remote sensing with associated ground-truth determinations (Ta et al. 2017). Moreover, because of the strong association of predatory fishes with vegetated areas, an assessment of the extent of vegetation can be used as a proxy for risk of predation on fishes of interest (see also Chapter 3).

Colonization by filter-feeding clams can also limit the suitability of shallow areas (Lucas and Thompson 2012). Clams consume plankton, limiting the energy supply to the food web in ways that can be difficult to forecast for shallow water bodies (Lucas et al. 2009). Therefore, sampling of shallow areas should be included in benthic and planktonic monitoring programs to improve our ability to forecast conditions in future restoration sites.

2.5.5.1 Goals for estuarine margins

As with the pelagic food web, information on the suitability of the nearshore food webs to support native fishes is incomplete and not yet amenable to clear quantitative goals. The Panel recommends that a goal be established using vegetated shallow areas as a surrogate for predation risk. This could be started with a simple index of proportion of area covered by vegetation (by region), which could later be calibrated to quantitative risks with a series of field studies.

2.5.6 River ecosystems

The major rivers flowing into the estuary lack the long history of diversified monitoring and research that is found in the estuary. There are, however, long-term studies focusing on salmon (see Chapter 3). There also is growing interest in freshwater conservation (Howard et al. 2018) and defining the patterns and magnitude of flow alteration in the tributary rivers (Sengupta et al. 2018, Zimmerman et al. 2018). These rivers have varying degrees of hydrologic characterization, water quality evaluation, and biological assessment (e.g., May et al. 2015, Mazor et al. 2016, Ode et al. 2016), and each ecosystem has specific attributes that must be considered in the design of an effective evaluation program with quantitative goals. Considerable local expertise and knowledge should be engaged in setting biological and abiotic goals and assessing progress. Establishing a strong monitoring and evaluation program to test if explicit expectations are being met is not a trivial undertaking, as demonstrated by the paucity of outstanding examples worldwide (but see Boxes 2.1 and 2.2 for two successful long-term examples). A wide range of professional expertise and local knowledge is required to achieve the best outcomes for the rivers of the CV.

One approach to setting goals for these rivers is to draw upon other, comparable evaluation programs that have been viewed as successful over the long term and have demonstrated positive outcomes for the ecosystems being monitored. A common feature of such programs is that a diversified portfolio of metrics is sampled that link to the main goals and objectives of the overall program. This would be salmon and steelhead in these ecosystems, but might be sport fish, bird communities, or aquatic biodiversity in other parts of the world. Based partly on successful long-term restoration programs elsewhere (e.g. Healthy Waterways Initiative (Box 2.2), Kissimmee River Restoration Program (Box 2.1), and the Grand Canyon Monitoring and Research Center), the Panel recommends that these features be organized into six categories: 1) hydrology and geomorphology; 2) water quality; 3) ecosystem processes; 4) algal communities and bioassays; 5) aquatic invertebrates; and, 6) fish assemblages. Since category 6 is discussed in detail in chapters 3 and 4, this chapter focuses on the other five categories.

Box 2.2 Healthy Waterways Ecosystem Monitoring Program for Southeast Queensland, Australia

Is there a long-term effective monitoring program that encompasses both an estuary and the major catchments draining into that estuary? One excellent example is the Healthy Waterways Initiative that started as a monitoring program for Moreton Bay in 1998 and was expanded in 2002 into a monitoring and evaluation program that included both Moreton Bay and the 15 major catchments flowing into Moreton Bay. Moreton Bay is about 1,523 km^2 and the catchments that flow into Moreton Bay have an area of 22,672 km^2 or about 14% of the size of the Sacramento-San Joaquin catchment.

This region is the fastest growing part of Australia with a population of approximately 2.7 million people (Abal et al. 2005). Southeast Queensland in eastern Australia has subtropical weather with warm water ocean currents. A major feature of Moreton Bay is that the estuary lies at the juxtaposition between the northern portion of Australia where sediments issues predominate producing “brown water” estuaries and the south-eastern quadrant of Australia where nutrient runoff produces “green water” estuaries. The expanding human footprint on the catchment has led to large-scale land use change and extensive urbanization in the Brisbane area. Not surprisingly, these changes led to increased loading of nutrients and sediments into Moreton Bay. The estuarine monitoring program for Moreton Bay began in 1998 due to the growing concern of local councils that population growth was leading to deteriorating water quality, reduced habitats for fish, turtles, and dugongs, and downward trends in overall ecosystem condition.

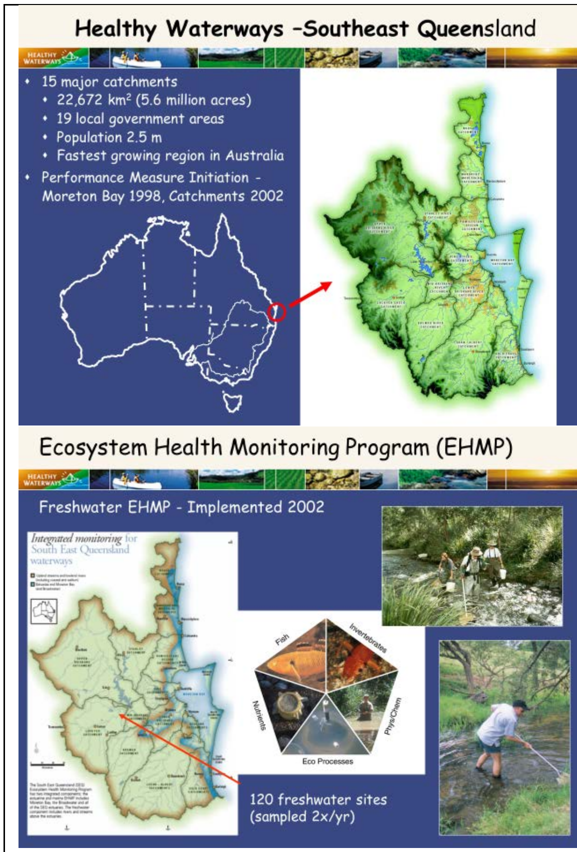
The early stages in monitoring and research on Moreton Bay and the estuarine sections of the rivers entering Moreton Bay focused upon the sources and impacts of nutrients (nitrogen and phosphorus) and sediments, particularly where those sources came from point sources. This led to the recognition that it was necessary to develop an ecosystem monitoring program for the major rivers discharging into Moreton Bay. The Ecosystem Health Monitoring Program (EHMP) was initiated in 2002 to expand the scope of the monitoring program to the major rivers within the basin. One hundred and twenty freshwater sampling sites were established to expand the Healthy Waterways Initiative into the catchments.

EHMP is an important tool for both managers and the public. Through biological, physical, and chemical indicators, EHMP provides an objective assessment of the condition of the ecosystems being sampled. It has been implemented to independently evaluate how effective various environmental protection strategies are for protecting and restoring the rivers, estuaries, and Moreton Bay through integrated monitoring research and management. What then are the key goals (biological, chemical, and physical) of the estuarine component and the freshwater component of EHMP?

The freshwater component of EHMP was developed over a two-year period using 1) conceptual models to identify potential indicators, 2) river and stream classification to characterize different river and stream types, 3) pilot studies to develop and assess indicators, and 4) a major field trial in multiple catchments. Five indicator groups responded strongly to disturbance gradients. These were fish, macroinvertebrates, ecosystem processes, algal bioassays, and physical and chemical indicators (commonly displayed as a pentagon of the five groups of indicators). Specific evaluation tools were used as strong indicators for each indicator group. The specific metrics for the fish are native species richness, the observed over expected fish assemblage, and the percent non-native individuals. The metrics for macroinvertebrates are invertebrate family richness, the richness of Plecoptera (stoneflies), Ephemeroptera (mayflies), and Trichoptera (caddisflies), the PET taxa, in the waterway, and a score derived based on sensitivity to pollution. The metrics for ecosystem processes are gross primary production, ecosystem respiration, and stable isotope signatures of aquatic plants, and the metrics for algal bioassays are limiting nutrients using diffusing substrates and stable isotopes on nitrogen in aquatic plants and macroalgae. Finally, the metrics for physical and chemical indicators are dissolved oxygen (including daily ranges), temperature, pH, and conductivity.

Western Moreton Bay and the river estuaries are monitored at 150 locations as part of EHMP. Water quality is an emphasis at these locations as turbidity, nitrate plus nitrite concentrations, and chlorophyll are responsive to erosion and nutrient loading. Annual monitoring snapshots include the extent and impact of sewage-derived nitrogen on macroalgae and mangroves, seagrass depth ranges, and phytoplankton growth bioassays. More intensive monitoring is done on occasion for such things as contingent planning for wastewater treatment plant upgrades, determining major changes to seagrass distributions, and examining growth responses of harmful algal blooms to nutrient inputs.

EHMP in southeast Queensland recognizes the critical links between catchments, the estuaries, and Moreton Bay, and the importance of long-term support in measuring the structure and function of these aquatic ecosystems. Support of ecosystem models provides the tools for synthesizing monitoring data from EHMP, and monitoring results are communicated through annual report cards that capture the interests of stakeholders and the community. A key component that makes EHMP effective is excellent communication of monitoring and scientific results using a variety of types of media. The methods used in EHMP provide useful and usable insights into developing goals (biological, chemical, and physical) for rivers, estuaries, and bays.



River hydrology and geomorphology characteristics will be specific to the river ecosystem under consideration. Long-term river gaging stations provide the data for evaluating the extent of hydrologic modification (e.g., magnitude, frequency, duration, timing, and rate of change) in a river hydrograph over time (Richter et al. 1996, Yarnell et al. 2015). An important geomorphic attribute is the response to flow associated with floodplain inundation along the river corridor. Other important hydrologic and geomorphic metrics include substrate characteristics, depth distributions, sediment type, flow velocities, and frequency of low or zero flows. Some critical measurements for salmon include temperature regimes during critical life history stages and flows during spawning, egg incubation and rearing, the turbidity and level of fine particles in spawning areas, and the frequency of little or no water on redds.

There are many candidate water quality metrics, but an emphasis on those that clearly affect fish communities should be at the forefront of an ecosystem evaluation program. Temperature is a well-known stressor in the watershed where higher air and water temperatures are becoming increasingly commonplace. Temperature mapping that measures three-dimensional coverage including shallow sediments is of growing importance. Dissolved oxygen is critical to measure as water temperatures increase and rates of river metabolism change. Turbidity is strongly responsive to catchment fire dynamics, forest die-off, and adjacent land use for agriculture or urban development. Other metrics for consideration in a strong and thorough water quality monitoring program are pH, conductivity, and dissolved nutrients. Fortunately, most of these water-quality variables are now amenable to continuous real-time measurement, and the affordable instruments can be co-located with measurements of hydrology.

River ecosystems are often assessed by their structure, although ecosystem theory points to the need to consider both structure and function. Structural approaches have been prevalent in river assessment (e.g. hydrology, geomorphology, water quality, and algal, macroinvertebrate, and fish communities), but river function is increasingly being used to assess effects of environmental stressors (e.g., von Schiller et al. 2017). Examples of functional measures include ecosystem metabolism, organic matter decomposition, secondary production, predation rates, nutrient cycling pathways and rates, and pollutant degradation. Ecosystem metabolism, including both gross primary production and ecosystem respiration, can now be routinely calculated from in situ measurements of dissolved oxygen, light, and temperature (Young et al. 2008, Tank et al. 2010). Stable isotopes of aquatic plants also can be used as a tool for determining long-term growth rates and to infer carbon sources and food web linkages.

Algal bioindicators can be useful components for setting biological goals in CV rivers where nutrient enrichment is a concern. Attached algae (benthic forms) have a long history of use in evaluation programs for assessing the conditions of streams and rivers. Algal community structure is responsive to water quality, river hydrology, and geomorphology. Where these data are available, status and trend monitoring can indicate temporal changes in water quality. Bioassay studies also can reveal nitrogen or phosphorus limitation. In addition, stable isotopes in algae can be used to infer dominant sources of nutrients and organic matter sources to food webs (e.g., nutrient sources from wastewater or agriculture).

Aquatic invertebrates also have a long tradition of use in bioassessment of rivers because they are common, widespread, and easily sampled. They are also sensitive to changes in sedimentation, flow regime, dissolved oxygen concentrations, and land use in the catchment (Bonada et al. 2006). Many streams and rivers in the CV have been sites for aquatic bioassessment studies using macroinvertebrates (Hawkins et al. 2000, Rehn 2009, May et al. 2015, Ode et al. 2016, Mazor et al. 2016). Common indicator taxa include stoneflies (Plecoptera), mayflies (Ephemeroptera), and caddisflies (Trichoptera), known as the PET taxa, which are all important dietary components for fish communities. Aquatic invertebrates also provide an integrated measure of stream condition because their presence indicates that the river has provided adequate habitat including food and good water quality over weeks to months (Abal et al. 2005).

In conclusion, the Panel recommends an approach for setting goals (biological and abiotic) for river ecosystem monitoring and evaluation that draws specific metrics from five categories (hydrology and geomorphology, water quality, algal communities and bioassays, aquatic invertebrates, and ecosystem processes). The chosen metrics should fit the needs of the specific river under investigation and specific actions being considered, and link where possible to knowledge of baseline conditions. Engagement of both scientific experts and local knowledge improves the likelihood that clear goals, responsive metrics, and quantitative expectations are foundations of the resulting monitoring and evaluation program.

2.6 Attributes (metrics) for Consideration for Ecosystem Evaluation

2.6.1 Estuary metrics

Aquatic ecosystems have many attributes that should be measured to fully understand how well they are functioning from an ecosystem and a human perspective. Here, a detailed list of possible metrics is presented; most would be measured frequently at multiple locations in an ideal world. Fiscal reality, however, usually dictates decisions of what gets measured, where measurements are made, and how often measurements are taken. Realistically, a subset of metrics that meet many of the criteria presented in Chapter 1, Section 1.5 should be chosen for monitoring. The first suite of suggested metrics focuses on the Delta. Not all of these metrics are recommended for every situation. This depends on what the management action is and where. Those in bold italics are not now being measured anywhere in the system. Those in regular italics are measured infrequently or only in certain places.

Metrics of physical condition: these should include continuously monitored (CM) data from fixed sites and measurements taken concurrently with all biological sampling.

- Geomorphology: Depth and height distributions (for deposition and scour, modeling, predator hotspots)
- Flow velocities
- Sediment transport and budgets for major rivers and estuary
- Turbidity (either directly or by proxies such as Secchi depth or optical backscatter)
- Temperature (including water column and sediment)
- Salinity (estuary) or specific conductivity (rivers)

Metrics of chemical condition: some of these are now feasible with CMs, while others would rely on individual water samples or bioassays.

- Mineral nutrients
- Dissolved and particulate organic carbon
- Dissolved oxygen
- Selected contaminants
- pH

Metrics of biotic condition: general

- Introduced species (percent by species, numbers or biomass)
- Diets and growth rates of selected fish species (see Chapter 3)

Metrics of biotic condition: estuary

- Phytoplankton: chlorophyll concentration (CM and sampling) and productivity (from chlorophyll, light, turbidity)
- Phytoplankton: counts and biovolume
- Phytoplankton: *Microcystis* abundance index
- Bacteria: counts and growth rate
- Microzooplankton: counts and ID
- Broad-scale microbial survey (High-throughput sequencing)
- Zooplankton: abundance, biomass, estimate productivity
- Epibenthic crustaceans: abundance, biomass
- Bivalves: abundance and biomass (for filtration estimates)
- Aquatic vegetation: extent, distribution, and biomass

2.6.2 River metrics

The second suite of potential metrics is focused on the rivers of the watershed. Which metrics to measure needs to be determined by the main concerns within each catchment, the actions being taken or proposed to improve ecosystem condition, the quality of the baseline information, and the monetary budget available for monitoring and evaluation. The metrics are placed into five general categories: hydrology and geomorphology, water quality, ecosystem processes, algae and algal bioassays, and aquatic invertebrates. This is not an all-inclusive list of metrics; the list, however, is informed by long-term monitoring programs addressing flow modifications and restoration projects. Quantitative goals and expectations can be developed after a thorough assessment of available knowledge and current monitoring within the catchment.

Metrics of hydrology and geomorphology

- Characteristics of flow modifications utilizing long-term gage data
- Extremes of high and low flow; flow duration curves
- Mean velocity structure within the main river channel
- Prolonged recession events along with timing and duration
- Cold water refugia linked to surface water/ground water interactions
- Flow inundation mapping in areas where floodplain connectivity occurs
- Sediment sources and delivery to the channel; sediment size distributions

Metrics of water quality

- Dissolved oxygen profiles (continuous) in critical fish habitat and at gage stations
- Turbidity measurements with emphasis on storm events
- Water temperature with a focus on fish requirements at various life history stages
- pH and conductivity (continuous) to monitor threshold requirements of biota
- Inorganic nutrients and total nutrients to determine eutrophication potential

Metrics of ecosystem processes

- Rates of primary production and ecosystem respiration (river metabolism)
- Stable isotopes of carbon and nitrogen to examine food webs
- Decomposition rates of organic matter
- Limiting nutrient assays using nutrient diffusing substrates
- Pollutant concentrations and degradation rates
- Predation rates on key biotic communities (including fish)

Metrics of algae and algal bioassays

- Algal bioassays to measure primary production rates and community structure
- Stable isotope ratios of nitrogen in macroalgae
- Expected versus observed benthic algal community structure
- Diatom identification for assessment of water quality

Metrics of aquatic invertebrates

- Aquatic invertebrate family richness in various habitat types
- Plecoptera, Ephemeroptera, Trichoptera (PET) richness
- Dominant macroinvertebrates found in the drift
- Expected versus observed benthic invertebrate community structure
- Functional group characterization by invertebrate feeding strategies

Informative potential metrics routinely exceed the budget available for ecosystem evaluation, monitoring and assessment. Difficult decisions on which metrics to use must be made. A variety of tools are available for making these types of decisions (e.g., SDM, logic chains, and developing specific, measurable, achievable, results-focused, and time-bound (SMART) goals). The methodology for selecting the best metrics for an ecosystem evaluation program is best left to those most engaged in the science and management of said ecosystem. Two take-home messages from the successful long-term monitoring and evaluation programs presented in the Boxes (2.1 and 2.2) are worth noting. The first lesson learned is the importance of defining a baseline. Quantitative goals and objectives must be linked to quantitative baselines. Investing the time and resources to develop an informed baseline is crucial where baseline information may currently be limited. The second lesson has three components. The first component is using conceptual models to guide investigations and to communicate to stakeholders. The second component is the use of external peer review on management projects from scoping to the final report. The third component is maintaining ongoing interaction with stakeholders interested in or involved with the monitoring and evaluation. Selecting which metrics to use for evaluating quantitative biological goals is a very important step in informing management actions. Effort and resources applied at the front end of management actions benefit the outcome and lead to the success of future endeavors.

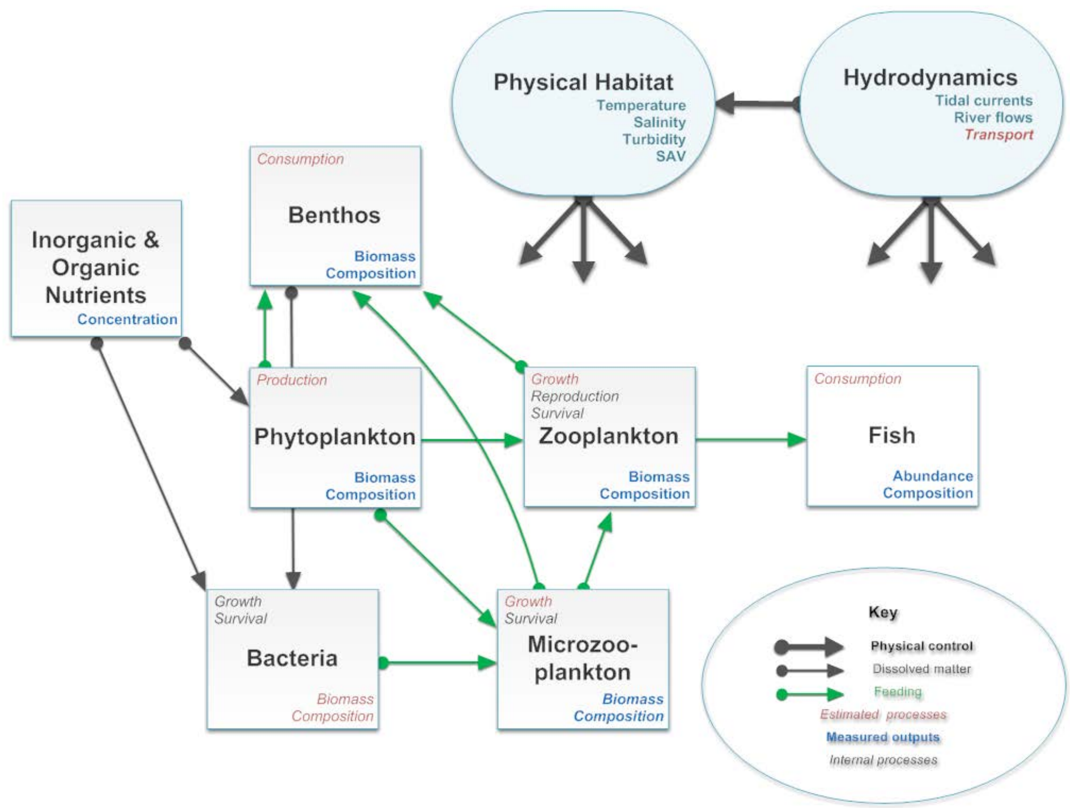


Figure 2.1. Simplified flow diagram of the pelagic food web of the upper SFE. Boxes are major functional groups. Within each box in the lower right are measurable outputs (generally biomass and composition) and in the upper left are internal processes. Blue lettering indicates outputs that could be or are being measured. Red lettering indicates processes that could be estimated from measured quantities. Internal processes (small black) will likely not be measured in the context of a monitoring program. At top, hydrodynamic processes (river, tidal, and density flows, wind waves) influence the spatial distribution of temperature, salinity, and turbidity, and over some time scales directly or indirectly influence the spatial organization of physical habitat and the distribution of submerged aquatic vegetation. All these factors influence the pelagic food web in various ways.

3 NATIVE FISHES AND FISH ASSEMBLAGES

3.1 Introduction[6]

The previous chapter provides ecosystem indicators to determine if large-scale management actions have improved attributes of the underlying ecosystem in ways that provide conditions favorable to native fishes. In this chapter, using fish metrics directly, the Panel addresses the question "How do we know if large-scale actions taken to improve conditions for native fishes, either individually or cumulatively, are doing any good?" In the past, these actions have usually been intended to improve conditions for fishes that are listed as threatened or endangered under the state and federal Endangered Species Acts (ESA) (henceforth “listed species”) or for unlisted runs of Chinook salmon (Chapter 4). However, past and ongoing efforts to improve conditions for these declining species has clearly been inadequate, as demonstrated by continuing declines of listed fishes (e.g., delta smelt, longfin smelt, and winter-run Chinook salmon), as well as by the growing list of species on trajectories to become listed species (Moyle et al. 2011, 2013, 2015, Appendix Table 9.3.2). It is clear that a broader approach is needed to understand and enhance the effectiveness of additional environmental flow releases or other management measures designed to improve the status of native fishes. This chapter shows why a broader approach should involve additional species (including non-native species) as indicator species, especially when grouped into fish assemblages. The latter are co-occurring groups of species with either similar environmental requirements or that, when combined, cover a wide range of environmental conditions. The "health" of such assemblages can also function to indicate restoration success or failure. In this chapter, alternative metrics that can be used to more effectively monitor fishes to show responses to changes in conditions such as improved flow regimes and habitat are suggested. These metrics are designed, directly or indirectly, to provide a “quantitative assessment of progress towards meeting the narrative objectives.” These are labeled as “biological goals” in the charge with the focus on native fishes (Appendix 9.2.2). It is essential to go beyond using just native fishes, including Chinook salmon and steelhead, to meet the assessment objectives. The SFE, especially the Delta, and its inflowing rivers are highly altered and support a wide variety of non-native species, from invertebrates to mammals, which are mostly fully integrated into the present ecosystems. Therefore, it is important to use as many species as possible in order to identify metrics that are sensitive to a broad range of

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[6] See geographic scope section in Chapter 1 for full definition of what area is covered here. However, in this section when we refer to the Delta, we mean the legal Delta with the addition of Suisun Marsh.

conditions and are actually measurable. The approach of using both native and non-native fishes is intended to provide a more comprehensive view of how change, whether through deliberate manipulation of the environment or not, is likely to affect the fishes of the estuary and its inflowing rivers. Non-native species have the particular advantage in that they can be either positive or negative indicators of habitat conditions, depending on the species and the interest group doing the evaluation.

It is worth noting that a major attempt to develop a multi-species approach, using just native species, was the *Recovery Plan for Sacramento-San Joaquin Delta Native Fishes* (USFWS 1996), that was developed in lieu of a delta smelt recovery plan. Seven unlisted species, plus delta smelt, were included in the plan. Their present status is a follows (Moyle et al. 2015, 2017):

- Delta smelt: listed as threatened under the plan; they are near extirpation today (Moyle et al. 2018).
- Longfin smelt: listed by California as threatened and facing extirpation in the SFE today.
- Sacramento splittail: a state Species of Special Concern; it has limited distribution but has benefitted from floodplain restoration projects (Moyle et al. 2004).
- Green sturgeon: listed as threatened.
- Spring-run Chinook salmon: listed as threatened.
- Late fall-run Chinook salmon: state Species of Special Concern.
- San Joaquin fall-run Chinook salmon: state Species of Special Concern, as part of CV fall-run Evolutionarily Significant Unit (ESU); the wild population has been replaced by a hatchery-derived population.
- Sacramento perch: extirpated from Delta, but present in ponds.
- Winter-run Chinook salmon were not treated as part of the report because they were already listed as endangered (1994). They remain endangered and survived a recent drought mainly through a captive breeding and rearing program (Moyle et al. 2017).

Unfortunately, this multi-species approach ultimately failed because most of the effort was placed on delta smelt recovery and the other non-listed species were largely ignored until they too were formally listed or declared species of special concern. In all cases, management efforts have been focused on individual species and not on potential synergisms possible under a broader, multi-species approach. Presumably, if managers had focused on actions that benefitted all or most species (e.g., improved flow regimes, widespread habitat restoration, better fisheries management, improved conditions in south Delta), fewer species would have been listed under the ESAs. In particular, the native fishes of the Delta would have generally been better off if more attention had been paid to the effects of the long-term decline in water devoted to fish and fisheries (environmental water), as described in Reis et al. (2019) and to what Beller et al. (2018) describe as management for landscape resilience. The latter is based on large-scale restoration of flows and habitat, together.

In this chapter some alternative approaches to the current strategy of management by listed species are explored, one crisis at a time. How a wider array of fish species can be used to determine whether or not changes in water and habitat management have created more favorable conditions for native and other desirable species is examined. Salmon and steelhead/rainbow trout are included as part of the approach but not emphasized, for reasons discussed in Chapter 4. Our general approach is compatible with the ecosystem-based approaches described in Chapter 2.

The basic rationale for this chapter is focused on one of the charges given to the Panel: "There are currently no programs that address the decline of other native fish species in the Bay-Delta watershed that are analogous to the SEP salmonid effort. Nevertheless, State Water Board staff will be establishing biological goals for other native fish and aquatic species in the Bay-Delta watershed and its tributaries using data from existing monitoring programs." The charge questions and brief answers are shown in Appendix 9.1.

Direct and indirect responses to the charge questions are organized through the following major sections of this chapter: (a) current surveys for fishes; (b) evaluation of potential indicator species; (c) determination of species assemblages useful for monitoring; (d) discussion of approaches for developing multi-species metrics; (e) recommended species and assemblage metrics; (f) other possible metrics; (g) use of models to support metrics; and, (h) conclusions.

3.2 Surveys for Fishes

The diverse spectrum (18 total) of existing fish surveys for the Delta provides a means to identify possible indicator species and assemblages and possible metrics using this information. Here, answers are provided to the following questions regarding the value of the surveys for monitoring: What are the major surveys that are currently used to assess status and trends of fishes in the Delta? What species are sampled by each survey and how well are they sampled? Are additional surveys needed?

3.2.1 What are the surveys?

Surveys that encompass the Delta, Suisun Marsh, and, more broadly, the SFE, are listed below and described in Appendix Table 9.3.1. Most are also described in detail in Honey et al. (2004). All are useful for the information they provide.

1. CDFW egg and larval survey
2. CDFW Bay Study midwater trawl
3. CDFW Bay study otter trawl
4. CDFW fall midwater trawl
5. CDFW summer townet
6. CDFW 20 mm trawl
7. CDFW Spring Kodiak trawl
8. USFWS beach seine
9. USFWS Chipps Island trawl
10. USFWS Mossdale trawl
11. USFWS Sacramento midwater trawl
12. USFWS Sacramento Kodiak trawl
13. UCD Electrofishing survey
14. UCD Suisun Marsh beach seining
15. UCD Suisun Marsh otter trawl
16. DWR Yolo Bypass fyke net
17. DWR Yolo Bypass beach seine
18. Delta pumping plant salvage

The United States Fish and Wildlife Survey (USFWS) Sacramento midwater and Kodiak trawl surveys (11, 12) are usually treated as one survey but the techniques are sufficiently different to justify treating them separately. The University of California, Davis (UCD) electrofishing survey (13) is based on results from a number of different short sampling programs that use boat electrofishing to sample inshore fishes, as exemplified by Young et al. (2018). The UCD Suisun beach seining survey (14) consists of monthly sampling at one to three locations in Suisun Marsh, since 1980; it is rarely reported as a separate survey. The California Department of Water resources (DWR) Yolo Bypass fyke net survey (16) was included because it is a major effort that focuses on adult migratory fishes of all species (Harrell and Sommer 2003). The Delta pumping plant salvage (18) quantifies the relative abundance of most freshwater Delta species because huge numbers of fish are "salvaged". However, the data are difficult to use because of variable pumping (sampling) rates, high rates of pre-screen loss, and other factors.

3.2.2 What surveys are essential for non-salmonids?

Multiple and diverse surveys are required to evaluate effects of large-scale management changes on non-salmonid fishes. Most useful are those that have a long history and consistently sample a wide variety of fishes (e.g., surveys 2, 3, 4, 8, 11, 12, 14, 15) However, all of the surveys combined provide one of the best pictures of long-term fish abundance and distribution for any estuary in the world. None of these surveys should be abandoned without careful consideration given to the impact loss of a continuous data set would have on monitoring the status of SFE fishes.

Non-native fishes typically dominate captures in most of these surveys, but the surveys have nevertheless been successful at documenting declines of native fishes. However, improved sampling is needed for most Delta resident, non-migratory fishes; the principal source of information on resident fishes (13) is actually a hodge-podge of short-term electrofishing sampling programs that have not been compiled. The USFWS is in the process of establishing a (hopefully) regular program of sampling freshwater resident fishes (B. Mahardja, USFWS, personal communication 2019). Additional effort is also needed to sample areas and habitats known to have populations of native fishes, such as Sacramento hitch and Sacramento blackfish, that are under-represented in all existing monitoring programs. Examples include ponds on Stone Lakes Wildlife Area, the Cosumnes River floodplain, and lower Cache Creek.

3.2.3 Using the heat map (Figure 1)

The green "heat map" (Figure 3.1) is a visualization of the relative importance (abundance) of 36 species of fish in 18 fish sampling programs. Abundance rankings (0-3, which coincide with shades of green on the map) are based on frequency of species catch over time. The goal of the rankings is to determine which species might be most useful for abundance or trend analysis. Factors considered included annual variability in total catch and trends in species abundance. A ranking of “0” corresponds to no catch, “1” indicates that at least one individual was caught but overall catch was low and trends not evident, “2” indicates that a trend was observed but with low overall catch, and “3” indicates that overall catch was high and a trend could be observed.

Figure 3.1 (next page). Green "heat map" of species occurrences in Delta sampling programs. Darker color indicates higher abundance of the species in the surveys. Numbers correspond to colors. “0” indicates no catch, “1” indicates only a few individuals captured, “2” indicates moderate abundance in catch but probably insufficient numbers for trend analysis, and “3” indicates species abundant enough in samples to allow for trend analysis.

	CDFW Egg and Larval Survey	CDFW Bay Study Midwater Trawl	CDFW Bay Study Otter Trawl	CDFW Fall Midwater Trawl	CDFW Summer Townet	CDFW 20mm Trawl	CDFW Spring Kodiak Trawl	USFWS Beach Seine	USFWS Chipps Island Trawl	USFWS Mossdale Trawl	USFWS Sacramento Midwater Trawl	USFWS Sacramento Kodiak Trawl	UCD Electrofishing Data	UCD Suisun Marsh Beach Seines	UCD Suisun Marsh Otter Trawls	DWR Yolo Bypass Fyke Trap	DWR Yolo Bypass Beach Seine	Delta Pumping Plant Salvage
American Shad	3	3	2	3	2	2	3	3	3	2	3	1	0	1	2	2	1	3
Bigscale Logperch	1	0	2	1	1	2	1	3	1	1	1	1	1	1	1	1	3	3
Black Bullhead	0	1	1	0	0	1	1	1	1	1	1	1	1	2	2	1	2	2
Black Crappie	0	1	1	1	1	1	1	2	1	2	1	1	1	3	2	3	3	3
Bluegill	0	1	1	1	1	1	1	3	1	3	1	1	3	1	1	1	3	3
Channel Catfish	0	1	2	2	1	3	1	1	1	2	1	1	0	1	2	3	1	3
Chinook Salmon*	0	2	1	2	1	2	2	3	3	3	3	3	1	1	1	1	2	3
Common Carp	0	1	1	2	1	2	1	2	2	1	1	1	1	3	3	3	2	1
Delta Smelt*	1	3	2	3	3	3	3	1	3	1	1	1	0	3	2	0	1	3
Goldfish	0	1	1	1	1	1	0	2	1	1	1	1	1	2	2	1	1	1
Green Sturgeon*	1	1	2	1	1	0	0	0	1	0	0	0	0	1	1	0	0	2
Green Sunfish	0	1	1	1	1	0	1	1	1	1	1	1	0	1	1	0	2	2
Sacramento Hitch*	0	1	1	1	1	1	1	2	1	1	1	1	1	1	2	1	1	1
Largemouth Bass	0	1	1	1	1	1	1	3	1	2	1	1	3	0	0	1	2	3
Longfin Smelt*	1	3	3	3	3	3	2	1	3	1	1	1	0	3	3	1	1	2
Mississippi Silverside	1	1	1	2	2	1	1	3	1	3	1	2	3	3	2	1	3	3
Pacific Lamprey*	1	1	2	1	1	1	1	1	1	3	1	2	0	1	1	1	1	3**
Pacific Staghorn Sculpin*	1	2	0	2	1	2	1	3	2	0	0	0	0	3	3	0	1	1
Prickly Sculpin*	1	1	2	1	1	3	1	2	1	1	0	0	1	3	3	1	2	3
Redear Sunfish	0	1	2	1	0	1	1	3	1	2	1	1	3	1	1	1	2	3
Sacramento Blackfish*	0	1	0	1	0	1	1	2	1	1	1	1	1	1	1	2	2	2
Sacramento Pikeminnow*	0	1	0	1	1	1	2	3	1	2	1	2	1	1	1	2	2	2
Sacramento Sucker*	0	1	1	1	1	2	0	3	1	2	1	1	1	3	3	2	1	2
Shimofuri Goby	1	2	3	2	2	1	2	3	2	2	0	0	1	3	3	0	2	3
Sacramento Splittail*	0	2	2	2	1	2	2	3	3	3	1	1	1	3	3	3	2	3
Spotted Bass	0	0	1	1	0	0	1	2	1	2	1	1	0	0	0	1	1	0
Steelhead*	0	1	1	2	1	0	3	2	3	3	1	2	0	0	0	1	1	3
Striped Bass	3	2	3	3	3	3	2	2	3	2	1	1	1	3	3	3	2	3
Threadfin Shad	2	3	2	3	3	3	3	3	3	3	2	3	2	3	3	2	3	3
Threespine Stickleback*	1	1	2	2	2	3	3	3	2	1	1	1	0	3	3	1	1	3
Tule Perch*	0	1	3	1	1	1	1	3	2	1	1	1	3	3	3	1	2	3
Warmouth	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	2	3
Western Mosquitofish	0	0	1	1	1	1	0	3	1	1	1	1	1	1	1	1	3	3
White Catfish	0	0	2	3	3	3	1	1	1	3	1	1	1	3	3	3	2	3
White Sturgeon*	1	2	2	1	1	1	0	0	2	0	0	0	0	2	2	2	1	2
Yellowfin Goby	1	2	3	2	1	3	1	3	3	1	1	1	1	3	3	1	2	3

* indicates native species **Rankings**: 0 = No Catch, 1 = Low Catch, 2 = Moderate Catch, 3 = High Catch

Examples of abundance trends in species with different rankings (Figures 3.2- 3.4 below) are from CDFW Fall Midwater Trawl catch data for 1967-2017.

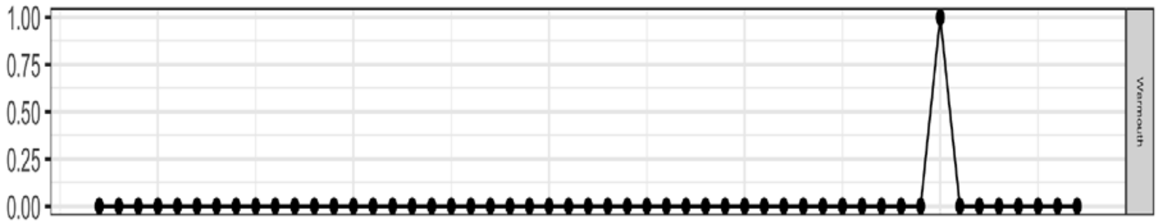


Figure 3.2. Example of rank “1” species. The species (warmouth) was captured only once over the course of the survey period. Note the scale on the Y-axis. Survey is not useful for determining abundance, trends, or dynamics of warmouth in the San Francisco Estuary.

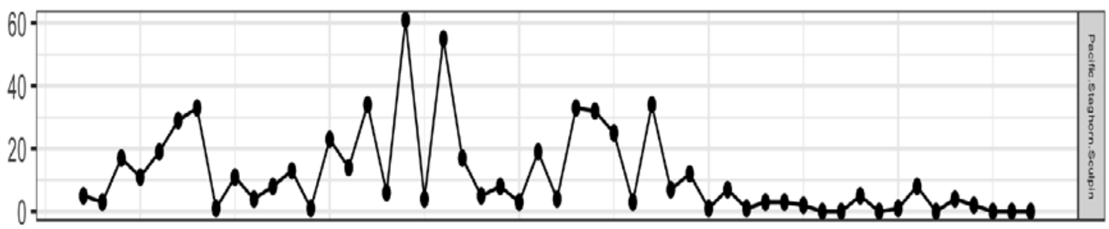


Figure 3.3. Example of rank “2” species. The species (Pacific staghorn sculpin) was captured most or all years of the survey period, but at low abundance. Note the scale (0-60) on the Y-axis. Total catch appears to show a downward trend over the survey period, but yearly catch exhibits considerable annual variation although there appears to have been a major change in overall catch starting around year 2000. This survey could be used to supplement other analyses; however, it is not robust enough to draw strong conclusions in regard to a staghorn sculpin abundance trend in the SFE. This also reflects the inadequacy of a midwater trawl survey for detecting the abundance of a benthic species.

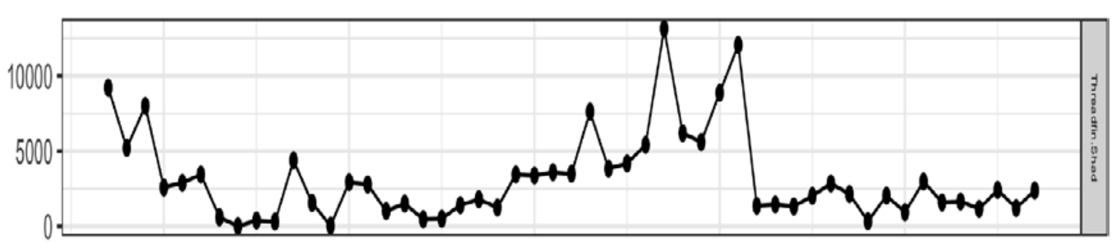


Figure 3.4. Example of rank “3” species. The species (threadfin shad) was captured in all survey years in moderate to high numbers. Strong trends and variability can be observed, with clear inflection points of decline/increase. Note the scale on the Y-axis.

This survey is robust for analysis of threadfin shad abundance and dynamics within the SFE. The reduced catches of both threadfin shad and staghorn sculpin in more recent survey years suggests a pattern worth investigating, however.

3.3 Indicator Species

3.3.1 Introduction

The traditional approach to monitoring the effects of change in the Delta and elsewhere is by tracking populations of individual species, typically listed species or species important to fisheries. The goal here is to identify species that would provide the most information if tracked individually or if tracked with a group of species with similar patterns of abundance. Therefore, the "heat map" (Figure 3.1) is useful because it allows for quick determination of what species are common enough in different habitats in the Delta to be useful for monitoring, beyond just listed species. The first cut is evaluating the 36 species used in the heat map; these are species that all the surveys together indicate are, or have been, common in the Delta. To allow for assembling species in groups that have the same characteristics, Appendix Tables 9.3.1 and 9.3.2 show various measures of status and species characteristics, respectively. The questions then become: Which species are likely to be useful as indicator species? Which species are likely to form assemblages that would react to change in similar ways or that would cover a wide range of Delta habitats?

3.3.2 Choosing Delta indicator species

Most conservation-oriented monitoring in the rivers and Delta has been oriented toward Chinook salmon, steelhead, delta smelt and longfin smelt. This is because they support fisheries, are listed species, or both. However, the monitoring has largely tracked population declines and/or collapse. Here, the role that other, more abundant species might play as indicator species is examined. In this report, *indicator species* are those species that are sensitive to changing conditions and can represent responses of other species, especially native species. The two smelt species ideally would have served as indicator species because they were widespread, abundant, and had short life cycles. However, they are currently too rare to have populations with readily detectable responses to changing conditions in the Delta[7]. Their main value for monitoring is

[7] Longfin smelt are apparently having a resurgence in their population in San Francisco Bay (J. Hobbs, UCD, pers. comm., 2019) but they remain rare in the Delta.

presence/absence to determine whether or not they are extinct in the estuary (or recovering), following criteria of Baumsteiger and Moyle (2017).

Delta indicator species should have all or most of the following characteristics: (a) have year-around presence in the Delta and/or Suisun Marsh; (b) use more than one part of the estuary, usually through different life history stages; (c) have large enough populations, even if in decline, such that responses to change can be detected if adequately sampled; (d) are captured in numbers in multiple sampling programs, so status is not left to the vagaries of a single program; (e) have a life history that is reasonably well understood; and (f) have a life history, physiology, and distribution that are similar enough to those of native species, such that, if conditions favor the indicator species, a number of native species with low abundance will be favored as well. This last characteristic is especially important. For example, Murphy et al. (2011) expressed the opinion that the use of "surrogate species" for making decisions about management of endangered Chinook salmon in the Delta was not justified because of inadequate proof it could actually work. The surrogate species of Murphy et al. (2011) are similar to indicator species defined in this report. The analysis in the next section shows that by carefully choosing indicator species, the last characteristic can be met by a handful of species.

3.3.3 Candidate indicator species

Candidate indicator species, chosen from the list in Appendix Tables 9.3.2 and 9.3.3 because of their potential to meet the above six criteria, are shown in Table 3.1. Species that have the highest number of desirable characteristics are Sacramento splittail, striped bass and American shad. The abundances of these three species are correlated with each other, as well as with those of Delta and longfin smelt (D. Stompe, unpublished, see also Appendix Figure 9.3.1). They would seem to be good candidates for indicator species, where their individual and collective abundance trends would indicate the overall '"health" of the Delta in terms of its ability to support native fishes with similar physiological and life history requirements. Tule perch and Sacramento suckers are also possible indicator species, although their ability to represent other native fishes is less certain. White sturgeon are very long-lived and are captured in specialized sampling programs that are mostly not part of regular sampling programs (except 16), so presumably are not sensitive enough to short-term changes to be an indicator species. One option, however, is to consider all these species together as focal species (see Section 3.6.1).

Table 3.1. Candidate indicator species in the Delta; letters stand for characteristics discussed in text. An "x" indicates the species has the characteristic. ? indicates it may have the characteristic but there is insufficient information. Blank indicates "not a characteristic." (**native species** in bold).

	Presence	Life history	Large population	Multiple gear	Well known	Correlated	
Species/**Characteristics**	**a**	**b**	**c**	**d**	**e**	**f**	**No. characteristics**
Sacramento Splittail	x	x	x	x	x	x	6
Sacramento sucker	x	x	x	x	?	x	5
Green sturgeon		x				?	1
White sturgeon	x	x	x	?	?	x	4
Tule perch	x		x	x	x	?	4
Prickly sculpin	x		x	x			3
Striped bass	x	x	x	x	x	x	6
American shad	x	x	x	x	x	x	6
White catfish	x	x	x	?	?	?	3
Common carp	x	x	?			x	3
Threadfin shad	x		x	x		?	3

3.3.4 Using indicator species for quantitative assessment

After an indicator species is chosen, the first step is to determine if one or more of the 18 sampling surveys discussed above can be used as a reliable measure of abundance trends. If so, then the trends should be graphed (as in Section 3.2.3). Ideally, at least two surveys would be available. If more than one survey is used, the trends should be compared and tested for correlation and if they are not correlated, it should be determined whether this is due to the survey methods or to differences in habitat. If the former, the biased survey should be used with caution. For each survey chosen, it should be determined if the catch trend is generally upward, downward, or stable since the beginning of the survey, preferably by using a simple regression analysis. Examination of the trend graph should show if a change point exists, where the trend makes an abrupt shift up or down. Using the average annual value of the survey, abundance of the species in the years since the action was taken can be used to determine whether the species is higher, lower, or about the same as the value since the change point and/or the value for entire sampling survey. This information could then be used to determine if the species is in decline or not and if the action taken to improve conditions had an effect. If desired the effects of the action could be scored on a numerical scale (e.g. -5 to +5). If multiple species are evaluated in this way, the final score could be based on the number of species with scores greater or less than some number. The species assemblages in Section 3.4 could be scored in this way.

For each species, additional information on status could be incorporated into such an analysis by looking at distribution as well as abundance, by determining presence or absence at each sampling location in each year. Presumably the more sampling stations in which a species is caught, the more abundant it will be (indicating likely correlation between distribution and abundance).

3.4 Species Assemblages

3.4.1 How were assemblages determined?

The previous Section (3.3) focuses on using indicator *species* for monitoring but also suggests that using multiple species as indicators is more desirable. *Fish assemblages* (with **native** fishes in bold below) were developed as alternatives to the single species-oriented monitoring that is largely the rule. The assemblages were initially constructed by grouping fishes with similar habitat and life history requirements and by their co-occurrence in the various sampling programs (see Appendix Table 9.3.3 for species requirements). Groupings were refined by a simple correlation analysis among all species in all sampling programs in which they were abundant (D. Stompe, UC Davis, unpublished). In addition, a multivariate analysis indicated that the fishes fall into broad "natural" groupings that generally coincide with the assemblages (Appendix Figure 9.3.1). Both native and non-native species used in each assemblage included only those scoring "3" in one or more surveys, so many species were too infrequently caught to justify inclusion in an assemblage. The number of surveys in which a species has a score of 3 (abundant enough for trend analysis) is in parentheses. Included species presumably have enough data on their abundance to enable trend analyses. Chinook salmon and steelhead juveniles are treated separately in Chapter 4 of this report and are caught mainly in sampling programs specifically designed to catch them. They are short-term residents of pelagic and inshore assemblages so could be added to those assemblages, if desired.

The assemblages listed below (3.4.2) should be regarded as a "first cut", as those most useful for monitoring. In the following Section (3.5) they are used as the basis for developing recommendations for multi-species metrics, which are presented in Section 3.6.

3.4.2 Possible habitat-based assemblages of Delta fishes

In the assemblages below, species in bold are **native** fishes. The numbers after each species refers to the number of surveys in which it which is rated as a "3" in abundance.

1. *Pelagic assemblage*. Members are typically juveniles or small adults caught together in mid-water trawls in open channels of the Delta. One possible problem is that threadfin shad are so abundant and widespread that they may dominate the assemblage.

   American shad (7)
   Delta smelt (7)
   Longfin smelt (7)
   Striped bass (8)
   Threadfin shad (9)

2. *Inshore, shallow-water assemblage*. Most fishes are juveniles and seasonal in occurrence and they tend to be associated with inshore aquatic vegetation in shallow, tidal water (e.g., tules, aquatic macrophytes). Mississippi silverside is usually the most abundant species; most other species are erratic but regular in occurrence. Splittail and young-of-year Chinook salmon occur seasonally. In the list below a single asterisk ("*") identifies freshwater species and *italicized* text identifies likely resident species.

   American shad (7)
   Bigscale logperch (1)*
   Bluegill (2)*
   Largemouth bass (2)*
   Mississippi silverside (2)
   Redear sunfish (2)*
   S**acramento pikeminnow (1)***
   Sacramento sucker (3)*
   Shimofuri goby (4)
   Staghorn sculpin (2)
   Striped bass (8)
   Threadfin shad (9)
   Threespine stickleback (3)
   Tule perch (5)
   Western mosquitofish (1)
   Yellowfin goby (5)

3. *Benthic channel assemblage.* This assemblage includes adults or large juveniles that are associated with the bottom in the larger channels. Additional sampling (e.g., at night) would probably add other catfish (Ictaluridae) species to the assemblage.

Prickly sculpin (4)
Sacramento sucker (3)
Splittail (5)
White catfish (6)

4. *Inshore vegetation assemblage.* This assemblage is dominated by species associate with large beds of aquatic macrophytes. More intensive or systematic sampling would probably add other species such as spotted bass and golden shiner.

Bigscale logperch (3)
Bluegill (2)
Largemouth bass (2)
Redear sunfish (2)
Tule perch (5)

3.4.3 Native tailwater fishes assemblage

The native tailwater fishes assemblage (see species below) represents a significant contrast from assemblages in the Delta. This is the native fish assemblage found in rivers below dams. The assemblage is not in the Delta although many of the species are present there. Many of these fishes are not even captured in the regular sampling programs (Figure 3.1) that focus on the Delta. For each regulated river, the assemblage contains some combination of 18 native species, although usually not all of them. The native species often constitute 75-90% of the species and biomass, with that percent decreasing with distance from the cold-water source (dam), as well as with reduced velocity and reduced gradient. A typical assemblage would likely include 10-12 native species. In the above list of potential constituent species, double asterisks ("**") indicate species that are listed or are species of special concern. The two-sturgeon species are "big river" fish that occur mainly in the Sacramento and Feather rivers, but individuals are seen on occasion in the lower reaches of most larger tributary streams. This assemblage has been documented (Brown 2000, Brown and Ford 2002, May and Brown 2002, Kiernan et al. 2012), but it is only regularly monitored in a few streams such as Putah Creek (e.g., Kiernan et al. 2012).

As rivers containing the tailwater assemblage flow towards the valley floor, the low-gradient reaches become increasingly warmer with the low flows of summer (often from diversions); the lowermost reaches mainly support non-native warm-water species such as largemouth bass, bluegill, threadfin shad, red shiner, fathead minnow, Mississippi silversides, white catfish, smallmouth bass (Brown 2000, Moyle 2002). For the Sacramento basin, May and Brown (2002) identified four fish assemblage metrics tied to environmental/water quality: (1) percentage of native fish, (2) percentage of fish intolerant of poor water quality, (3) number of tolerant species, and (4) percentage of fish with external anomalies. The first two metrics were associated with higher elevation reaches with cold-water flows, while the latter two metrics were generally associated with poor-quality habitat at low elevations.

California roach**
Chinook salmon (anadromous)**
Green sturgeon (anadromous)**
Hardhead**
Pacific lamprey (anadromous)**
Prickly sculpin
Rainbow trout/steelhead (anadromous)**
Riffle sculpin**
River lamprey (anadromous)**
Sacramento hitch**
Sacramento pikeminnow
Sacramento splittail**
Sacramento sucker
Sacramento tule perch
Speckled dace
Threespine stickleback
Western brook lamprey or Kern brook lamprey**
White sturgeon (anadromous)**

3.5 Factors Useful for Developing Multi-species Metrics

The following are observations that contributed to the search for useful multi-species metrics to evaluate the effects of altered flows, habitat, and management on fishes of the Delta region and CV.

1. Non-native fishes dominate most general surveys in the freshwater tidal Delta; they are most abundant whether measured by numbers or biomass. Native fishes are comparatively minor players in the present Delta ecosystem, although their

importance increases as the water becomes more saline and dominated by marine/brackish water species.

2. The pelagic fishes assemblage is the one most sampled (9/18 surveys), but delta smelt and longfin smelt are no longer abundant enough in the Delta to reliably quantify trends in abundance for this assemblage. The Pelagic Organism Decline (POD) concept brought attention to this assemblage and its decline. However, it included threadfin shad (Sommer et al. 2007) which is now the most frequently caught fish overall; they were captured in all 18 surveys and the overall catch was relatively high, and a trend could be observed over the span of the survey period in nine surveys. American shad were not part of POD but perhaps should have been, given they area also pelagic planktivores. Striped bass, a POD species, are in decline but still common enough to reliably quantify a trend in abundance.

3. The USFWS beach seine survey is the most useful survey for species inhabiting shoreline areas in fresh water, including juvenile Chinook salmon. Many of the fishes are seasonal, mostly as juveniles taking temporary residence in shallow water. The most abundant fish today is Mississippi silverside, a year around resident and fairly recent invader (1970s). The UCD seine survey shows similar trends but samples more brackish water species and has many fewer samples.

4. Data from various egg, larval, and pelagic juvenile surveys (6 total) were not used here because the surveys did not catch a large number of identifiable species. However, given that they sample early life history stages, their utility as indicators of rapid change should be explored.

5. Juvenile Chinook salmon and steelhead were caught in surveys designed specifically to catch them (e.g., 9, 10). However, the best surveys that catch both salmon (including smaller salmon of natural origin) and other fishes are the USFWS seining survey (8) and the DWR beach seine survey (17). Conversely, a distinctive feature of Chinook salmon distribution is that most smolts (i.e., larger juveniles) are caught in midwater trawls in programs designed to catch them. Most of the effort to capture juvenile salmon and steelhead is through screw traps set in the main rivers (Chapter 4).

6. Resident freshwater fishes are best sampled with boat electrofishing. Only one short published survey was used for our list of sampling programs (13) but other, similar surveys have been performed (but not compiled). They consistently show that largemouth bass and sunfishes dominate weedy inshore waters, with few native fish

(except tule perch in some locations) present. A regular (preferably annual) survey is needed and has been proposed by USFWS (B. Mahardja, pers. comm. 2019).

7. The fish salvage data originates from salvage facilities associated with the two pumping plants in the South Delta that supply water for the CV Project and State Water Project. The "sampling" is erratic because the number of fish captured each day depends on Delta inflow, how much water is being diverted, special releases for fish conservation, and other factors. However, the data collected from salvage efforts provides basic background information on the relative abundances of many, if not most, species, because of sheer numbers of fish caught (Figure 3.1).

8. Because of their abundance, non-native species can be important indicators of success of restoration projects, especially if combined with environmental indicators. For example, an increase in the long-naturalized striped bass and American shad populations could represent an increase in the overall "health" of the Delta, as determined by features such as higher turbidity and larger zooplankton populations (see Chapter 2). Conversely, an increase in non-native centrarchids (largemouth bass, etc.), along with an increase in the extent of aquatic plant beds and an increase in water clarity, could represent a negative trend in Delta "health" (although regarded as positive by those who benefit from fisheries for bass, sunfish, catfish, and common carp.

9. Native species that seem to have the most potential for monitoring as indicator species are Sacramento splittail and perhaps Sacramento sucker, white sturgeon and tule perch, especially if included in assemblages (Table 3.1). Other native fishes that might have potential are currently too narrowly distributed or too low in abundance to be very useful (e.g. Sacramento hitch, blackfish).

10. All surveys that capture multiple species of fish, mainly as juveniles, proved to be useful. However, an annual resident fish survey for the Delta is needed, as well as more frequent sampling of all fishes in tributary streams such as the Stanislaus, Mokelumne, Tuolumne, and Merced rivers. Some sampling is no doubt taking place in these tributaries and, if so, the data should be made readily accessible (with other surveys).

3.6 Multi-species Metrics

The following metrics were developed from the information presented above (Sections 3.4, 3.5) and other sources. They are designed to help track progress towards meeting the narrative objectives. The metrics are preliminary and can be treated as alternatives to one another or as symbiotic (validating). Where feasible, multiple metrics should be used for verification of results.

3.6.1 Delta indicator and focal species

An indicator fish species metric would include measures of abundance and distribution of the multiple Delta species to indicate "recovery" of conditions favoring estuarine fishes that require fresh water at some stage in their life cycle: striped bass, American shad, Sacramento splittail, Sacramento sucker, tule perch, delta smelt, and longfin smelt. These species have a diverse set of ecological requirements, so would not necessarily respond similarly to change; the species chosen for the indicator role would be treated individually but discussed together. This metric could have a composite score with each species being rated on an overall abundance trend (increasing, decreasing, stable) over 5, 10, and <20 year periods. This trend, where possible, could be based on multiple surveys. For example, Moyle et al. (2004) used data from seven of the surveys to help determine long-term trends in the splittail population.

Each species could also be rated on how much their distribution through the Delta has changed, based on a grid (occupancy) or number of sampling stations in a particular survey. The starting assumption would be that each species was once found in all stations, at least seasonally. This evaluation could be based on a score sheet used for evaluating the current status of all California inland fishes, native and non-native (Moyle et al. 2013).

It is worth noting that the above use of multiple indicator species is very similar to the *focal species* approach used by scientists interested in conservation of important habitat for birds (Nicholson et al. 2013, Dybala et al. 2017, Shuford and Dybala 2017). For this approach, a group of species are chosen, the focal species, which together have habitat requirements that encompass the range of conditions found in the desired habitat type. For example, to monitor progress on restoring riparian habitat in the CV, Dybala et al. (2017) chose 11 species of birds as focal species. They compiled data from various sources on the densities of breeding individuals of each species as well as estimated population sizes of breeding individuals of each species in four regions in the CV. Figure 3.6 illustrates one result from this study, a diagram that shows estimated population sizes of the 11 focal species compared to projected status of habitat if conservation measures are successful after 10 and 100 years. The habitat improvement reflected in the diagram is presumed to provide habitat for the many other (non-focal) bird species that depend on riparian forests. Such a diagram could be created for aquatic habitat using data from the 18 fish surveys, as a visual "report card."

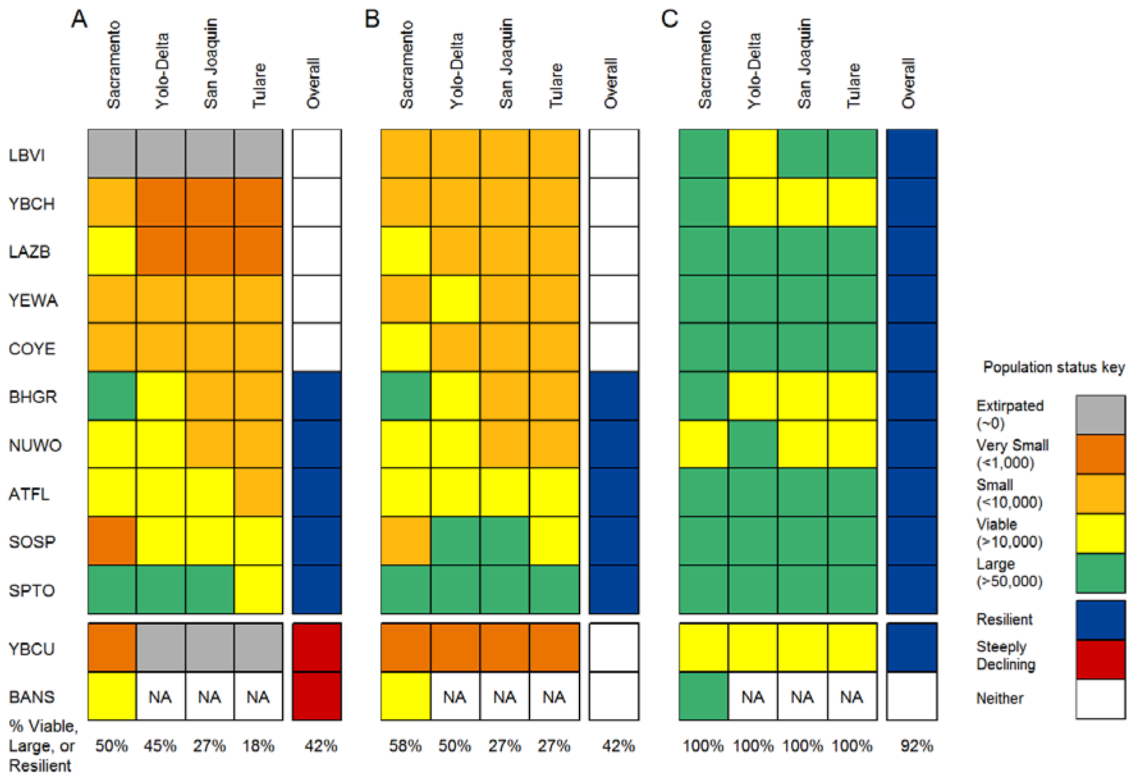


Figure 3.6. Evaluation of regional focal bird species populations for: (A) current status; (B) projected population after 10 years of projected restoration actions; and, (C) after 100 years of projected restoration actions, in four basins in the CV. Abbreviations on left are bird species while colors represent population status. From Dyabala et al. (2017).

3.6.2 Delta fish assemblages

Another potential Delta fish metric is a variation on the theme expressed in the Delta Indicator Species Metric, using the combined results of all the various surveys on an annual basis. The basic questions being addressed would be: 1) has there been a major change in the stability/composition (this would have to be defined) of any or all of the Delta fish assemblages described above?; and, 2) what do those changes mean?

3.6.3 Warm-water resident fish assemblage

A warm-water fishes metric would inherently be a negative one for native species, on the assumption that abundance and distribution of non-native fishes associated with dense beds of Brazilian waterweed (*Egeria densa*) and other non-native aquatic plants (or with shallow warm water in general) reflect a Delta that is not favorable habitat to most native species. Species to be included would be largemouth bass, four sunfish species (bluegill, redear, green, and warmouth), Mississippi silverside, and golden shiner. Scoring would be as for the Delta Indicator Species Metric (3.6.1), but the final scores would be negative. It might be possible to have a metric specifically related to the distribution of invasive aquatic plants as determined by remote sensing *(e.g*. Durand et al. 2016). See also, Chapter 2.

3.6.4 Suisun Marsh assemblage

Suisun Marsh is being increasingly recognized as a major nursery area for native and other sensitive species that rear in brackish water. A Suisun Marsh metric would track capture frequency of young-of-year striped bass, American shad, Sacramento splittail, starry flounder, and staghorn sculpin, and note catch frequency of delta and longfin smelt as well. The focal species approach might work especially well here because there is monthly sampling data from 1980 to present, so annual catches could evaluated at 1-, 5- or 10- year intervals to evaluate status of fish habitat in the Marsh.

3.6.5 Invasive species fish, macroinvertebrate, and plant assemblage

Ideally, management actions that are specifically designed to improve conditions for native species should make conditions less favorable for recent invasive species that have negative effects on native species. An invasive species metric could measure trends in the distribution and abundance of a selection of recent invasive plants, invertebrates, and fishes, on the assumption (hypothesis, really) that non-native species of all types tend to be associated with one another. Possible species include: overbite clam, Siberian prawn (displacing bay shrimp in fresh water), two species of Black Sea jellyfish, Brazilian waterweed, water hyacinth, and Mississippi silverside.

3.6.6 Tailwater river native fishes assemblage

Regulated rivers below dams (tailwaters) in the CV typically have the most complete assemblages of native fishes in the entire region, with the dominance of 8-10 native species determined by flows and temperature. The native fishes, including salmon and rainbow trout, thrive in cold to cool (usually less than 20°C in summer) flows from dams, which very few non-native fishes are able to do. This is in contrast to the Delta, which tends to have environmental conditions that favor non-native species. In the rivers, the water warms as it flows downstream, resulting in non-native species becoming an increasingly important component in the assemblage. Thus, a tailwater metric could involve measures of relative abundance of native species (e.g., percentage of total miles of habitat with 75% native fishes by numbers/biomass) at regular spatial and temporal intervals. Water released from dams to benefit estuarine fishes should take effects on the tailwater fish assemblage into account.

3.7 Other Metrics

In this section, three additional approaches to determining the effects of large-scale management actions on native and other fishes are discussed. The first, *fish health metrics*, originates from the basic concept that the health and condition of individual fish can tell a great deal about how well the ecosystem in which they live is functioning. This metric has considerable promise but data and monitoring of this type are largely lacking for most fishes, and new data is collected only sporadically. The second metric, *outflow sensitive species*, is based on the idea that if species sensitive to Delta outflow are thriving, then the entire estuarine ecosystem is likely to be thriving as well. This particular methodology is too focused on one metric, the response of a few euryhaline species to Delta outflow, to provide the sensitivity needed here. The third metric, *Index of Biotic Integrity*, has proven very difficult to use in California streams (as prescribed) but shows little promise of being useful for the Delta.

3.7.1 Fish health metrics

This chapter has largely focused on ways to use abundance measures as indicators of the status of fish populations and their response to environmental change, which primarily reflects the availability of data. However, if samples are large enough, there are measures of health of individual fish that could be useful, to detect whether the fish are responding to improvements in habitat and enhanced prey resources through restoration projects. These metrics could be supplementary to abundance-based metrics, provided they are done systematically. Some candidates for such metrics include:

1. *Length/weight relationship (condition factor)*. Condition factor is often used as a measure of fish health; plumper fish are seen as healthier fish that are growing rapidly. Most of the sampling programs listed in this chapter routinely measure a sample of fish lengths. Measuring weights could be time consuming but add considerable additional information to each survey. Changing values for condition factors can show whether environmental change has been good or bad for individual fish, even in a fairly short period of time (weeks)

2. *Growth*. Rapid growth (absolute or comparative) in fishes is often an indicator of favorable habitat conditions (e.g., food, temperature). Therefore, determining the relationship between size and age can tell a great deal about the health of individuals in a population, even if it is just for developing a simple relationship such as provided by length frequency distributions. For example, larger females of many species produce more eggs, so females that grow large and reach maturity rapidly may have a much higher lifetime egg production than slower growing individuals. Slow growth may also be an indicator of inadequate food supply or stressful environmental

conditions. Measuring incremental growth often requires sacrifice of samples of fish to obtain otoliths. Analysis of otolith geochemistry (stable isotopes), however, can also indicate where the fish was living at various stages of its life cycle and what it was eating, suggesting potential for being influenced by restoration projects.

3. *Histopathological condition.* A histopathology investigation of individual fish can determine factors such as contaminant and starvation stress. The simplest way of quantifying this is percentage of fish that show signs of having been exposed to unhealthy conditions, through poor body condition, lesions, or other problems.

4. *Diet analyses.* Studies of the diets of fishes are most useful if done in conjunction with the other individual fish metrics as described above, as well as with invertebrate studies as discussed in Chapter 2. For species of interest, dietary studies need to be repeated periodically (space and time) to see how the species fits into the changing food webs in the estuary. For example, if a wetlands restoration project is supposed to export invertebrate food to the main channels, diets of fish in those channels (e.g. delta smelt, juvenile striped bass) could reflect success of the export strategy (e.g., Hammock et al. 2019). Understanding the diets of fishes can also help to determine if food or predation are limiting factors affected (or not) by major management actions. For example, Buchanan et al. (2018) show that almost all juvenile salmon passing through the highly altered central and south Delta are consumed by predators. Using presence of salmon DNA in the guts of predatory fishes as an indicator, they showed a wide array of fishes prey on salmon. Similar findings have been found for Mississippi silverside and other small fishes as predators on delta smelt (Baerwald et al. 2012).

5. *Recruitment and mortality rates.* These are population metrics that are important for life history-based models and are routinely collected for salmonid populations in order to generate VSP analyses. These metrics are lacking for non-salmonids in the system because the data are hard to collect, and require frequent sampling of different life stages and use of methods such as mark-recapture, which are time-consuming and expensive

3.7.2 Outflow-sensitive species

The Bay Institute (Bennett and Rosenfield 2017, draft, unpublished) proposes using responses of species demonstrably sensitive to Delta outflow as a measure of how well the entire ecosystem, especially the Delta, is functioning as a healthy estuary should. Their analysis is based on outflow (X2)-response relationships for “sensitive species” such as starry flounder, bay shrimp, Sacramento splittail, delta smelt, and longfin smelt (Jassby et al. 1995). For each species, an “exceedance probability distribution” estimates the likelihood that a given level of fish abundance or recruitment will re-occur or be exceeded in a given position of X2 as determined by flow and export regimes. A "return period" estimates the number of years that will be required for a given level of abundance (or recruitment) to re-occur. Thus, Bennett and Rosenfield suggest that distributions of biological response can be used to identify minimum population thresholds and "typical" population responses under varying Delta outflow scenarios.

The Bay Institute analysis uses a long time series of data that encompasses times when invertebrate prey were much more abundant than now. It is likely that the X2 relationships developed by Jassby et al. (1995) likely no longer apply, at least for some species. Also, this analysis includes both pre- and post-POD data, and the ecosystem apparently went into a new state around 2000–2002. Note that the delta smelt MAST report (IEP-MAST 2015) showed a positive relationship of early survival of delta smelt to flow (replicated in the Bay Institute Report) that was not discovered earlier, probably because it did not exist (i.e., the response of delta smelt to flow changed after the POD)[8]. The overall problems with this approach are discussed in Chapter 2.

3.7.3 Index of Biotic Integrity

This index is included for the sake of completeness in discussing possible approaches to the problem, although the Panel does not find it likely to be useful. IBIs were proposed by Karr et al. (1986) as a way to evaluate how much the biota of a reach of wadeable stream had changed (usually meaning degraded) from its presumed original condition. Karr and Dudley (1981) defined biotic integrity rather vaguely as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region."

The original IBI focused on the diverse fish assemblages of Midwestern streams and consisted of 5-10 metrics, such as percent species tolerant of poor water quality or percent non-native fish in samples. Metrics were scored on a 1 to 5 point scale and then added for a total score, which was normalized to a 100-point scale. The biotic integrity of fish assemblages was then reported as: (a) very good to excellent, score of 80-100; (b) good, 60-79; (c) fair, 40-59; and, (d) poor, <40. Moyle and Marchetti (1999) developed

[8] This analysis seems to lead to a somewhat circular argument. If outflow level Q results in an expected population index of Y, outflow could be set (in theory) to Q in order to get that response on average. But if response Y does not happen, then you have to ratchet up Q to get the desired response. Thus, to maintain the population of longfin smelt at historical levels using outflow, most reservoirs feeding the estuary would have to be emptied.

an IBI for Putah Creek, a regulated low-elevation California stream, using eight fish metrics (percent native fish in samples, number of native fish species, number of age classes of native minnows and suckers, total fish species, total fish abundance, percent top carnivores, percent species tolerant of poor water quality, and percent non-native lentic species). The final IBI scores for eight stations over a three-year period ranged from 35 to 100, which tracked the patterns seen in a 10-year study of the creek fish populations by Kiernan et al. (2012). Moyle and Randall (1998) used a highly modified IBI to evaluate the "health" of 50 Sierra Nevada watersheds. For lowland rivers, May and Brown (2002) proposed six potential IBI metrics and concluded, using multivariate analyses, that the metrics were sufficiently different from one another to be used in an IBI for the Sacramento River watershed. However, no IBI was presented. Deegan et al. (1997) proposed an estuarine IBI for Atlantic coast estuaries. Whether an IBI is a suitable metric of use in the highly modified SFE is debatable but unlikely.

3.8 Models

Ideally, each of the species used as an indicator species or as part of an assemblage should have a conceptual or quantitative model associated with it, preferably one that includes metrics discussed in Section 3.6. Such models would be useful for making predictions about the effects of management actions on species or groups of species. The information required for such a model includes attributes of abundance, life history and genetic diversity, productivity, and spatial structure. However, this level of information is largely lacking with a few possible exceptions: delta smelt, Sacramento splittail, and striped bass.

Delta smelt have been the focus an extraordinary effort to understand their life history and factors affecting growth and mortality. This information has been compiled to create a detailed individual based model (Rose et al. 2013 a,b; Kimmerer and Rose 2018) and a conceptual model (IEP-MAST 2015). The models have been used to test effects of a variety of factors on delta smelt populations such as those affecting mortality, growth, and reproduction, including entrainment in the South Delta pumps and changes in their food supply. While the model continues to be potentially useful, the delta smelt has declined to such low numbers that matching the results of the model with recent data has become difficult.

Sacramento splittail were subject of a conceptual model of their life history dynamics in Moyle et al. (2004). While VSP-type parameters (for salmon) were largely unavailable, enough information was available to develop a series of hypotheses that could be tested with further life history studies and, to some extent, a simulation model of population dynamics. The study emphasized the many uncertainties in Sacramento splittail life history, but enough information was available to create a reasonable model, with caveats. In fact, the information compiled for the model plus the results of the modeling was convincing enough that the USFWS delisted the Sacramento splittail from its status as a threatened species, to a state Species of Special Concern (Moyle et al. 2015).

Striped bass have had an individual-based model developed for young-of-year in East Coast populations (Rose and Cowan 1993) but its applicability to the SFE and inflowing rivers has not been tested. Considerable data are available for developing a striped bass model, in part because three of the original long-term fish surveys in the SFE were developed for monitoring life stages of striped bass (1, 5, 4). Kimmerer et al. (2000, 2001) use these data with a simple life cycle model to deduce that the large losses of young striped bass to export pumping were offset by density dependence, and that the ~1970s decline in striped bass abundance was due to a decline in carrying capacity for young fish and the loss from the population of many of the older, larger, more fecund females. Given the potential importance of striped bass as an indicator species, the continued development and use of a life cycle model is warranted, as is continued collection of data that supports its use.

3.9 Recommendations for Setting Biological Goals

The original charge to the Panel states: “The biological goals are proposed to be quantitative targets that can be used to assess both the status and trends of representative native fish communities...” The following are the Panel’s recommendations for how to achieve the target metrics that can evaluate effects of major management-related and "natural" environmental changes on native fishes.

The general method proposed starts with the data currently being collected by the diverse (18 total) fish surveys that encompass wide areas over extended periods of time, as well as with information on the species habitat requirements. An important difference here from current methods is the Panel’s use of data from both non-listed native species and non-native species, as well as listed species. The Panel’s contention is that using the responses of a variety of fish species should expand our ability to detect change, positive or negative, and then relate those responses to effects on listed species or other native species. A particularly useful way to use the existing abundance data is to develop metrics that involve multiple species (Sections 3.4-3.6). Essentially, this approach should help to make it more likely that adaptive (experimental) management can be used to test effects of management actions.

Exploring six potential approaches (metrics) that use the survey data for Delta fishes and existing data for river fishes is recommended for determining whether or not biological goals can be reached through flow and habitat improvement projects. These approaches are: (1) the distribution and abundance of a selected fish species as indicator species; (2) distribution and abundance of multiple species, treated as assemblages of species with similar habitat requirements and/or as assemblages of species that encompass a broad range of habitat conditions; (3) the collective abundance and distribution of common non-native warm-water fish species; (4) abundance and condition (growth, etc.) in critical nursery areas for fishes (e.g., Suisun Marsh); (5) abundance and distribution of invasive plant, invertebrate, and fish species; and, (6) a river-based metric that measures response of native fishes to flow releases in tailwaters below dams. If this general approach is considered acceptable, then we recommend developing several or all of these approaches to be used simultaneously. Additional information could be collected from individual fish taken in the surveys that could provide "performance metrics" that might provide more rapid assessment of changes than the surveys themselves (3.7.1). For some well-studied species, VSP-type metrics could presumably be used as well (3.8), as shown in Chapter 4.

We also examined an approach that used responses of selected species of fish and invertebrates to changes in Delta outflow, as well as an IBI (3.7). However, both approaches had disadvantages that make them difficult to use. Finally, using some landscape metrics (e.g., extent of aquatic macrophytes in Delta) simultaneously could provide additional insights into the underlying causes of fish responses to a changing environment, especially for invasive species (3.6.5), and we recommend developing metrics that use such information to compare with fish metrics.

The first four of the six metrics above are based on tracking individual species abundance trends. This is similar to the present method used by agencies, in which changes in management are evaluated using numbers from standard surveys of listed species and fisheries species, mainly delta smelt, longfin smelt, steelhead, and the four runs of Chinook salmon. Two major problems with the present method are the increasing rarity of a number of species, such that changes in management are not likely to be detectable, and the dependence of salmonid abundance on hatchery production (Chapter 4). An improved version of this general approach would be to use *both* distribution and abundance of select group(s) of native and non-native fishes that are mutually sensitive to changes in flow and habitat. While distribution is tied to abundance and, especially, to outflow, a measure of distribution, at least for resident species in most years, could indicate how much additional habitat was restored (or lost) by management actions.

Other metrics we recommend exploring include quantifying the distribution and abundance of select, high-impact, non-native fishes (3.6.3, e.g., largemouth bass, Mississippi silverside). This is a metric that could be sensitive to failures of management to improve conditions for native fishes. A version of this negative method is to look at the response, if any, in the distribution and abundance of ecosystem-changing "invasive" non-fish species, such as overbite clam, Brazilian waterweed, and Black Sea jellyfish. A retreat of any of these species would be regarded as a positive event.

Measuring the abundance and condition (growth, etc.) of fishes inhabiting Suisun Marsh is recommended because of increasing evidence for its role as a nursery for fishes such as splittail, tule perch, striped bass, longfin smelt, and, formerly, delta smelt. Abundance of such fishes throughout the estuary may depend in part on their rearing in the Marsh. This metric would be used in conjunction with other metrics (3.6.1, 3.6.2). It would also capture some of the abundance trends in brackish-water species that are not abundant in the Delta, such as starry flounder and Pacific staghorn sculpin. These metrics should also be applied to fishes elsewhere in the Delta, for comparison.

An alternative to the above choices is to develop quantitative population models for key species such as delta smelt, splittail, and striped bass. This would require considerable investment in research to develop the parameters needed for each model or models, including those mentioned in Section 3.8. A refined model for splittail would be a good place to start because Moyle et al. (2004) indicate what additional information is needed. Likewise, Kimmerer et al. (2000, 2001) provide a simple life cycle model for striped bass that could be used as the basis for a more comprehensive model.

Presumably other non-fish metrics, such as changes in water quality, number of acres of habitat restored, and reduction in extent of invasive aquatic weeds, are also going to be needed. They would provide a more rapid assessment of progress, but such measures would also need to be tied to long-term changes in fish populations.

Overall, given the complex life history patterns and ecological relationships that Delta fishes have with their environment (i.e., there is no "one thing to rule them all") and with each other, the best approach is to use multiple approaches and metrics. This is particularly true because some metrics are not sensitive to short-term changes, but useful for measuring long-term changes and because different metrics are sensitive to changes in different habitats (e.g. open water, weedy edges, Suisun Marsh). Collectively, the fish metrics discussed here are most likely to start to tell a story after 5 years or so, if combined with metrics based on the historic data. Regardless, there is a continued need for the 18 standard surveys (Appendix Table 9.3.1) to continue, as well as for new ones to be developed that target additional fishes.

4 SALMON AND STEELHEAD

4.1 Introduction

The overall aim of this chapter is to provide recommendations for setting biological goals for populations of natural-origin Chinook salmon and steelhead in California's CV. The following question captures our interpretation of the charge to the Panel: *How should we evaluate status and trends and population-level responses of Chinook salmon and steelhead to flow and habitat restoration actions?* Answers to this question, as presented below, should help identify key metrics to be used in setting biological goals while also providing one or more approaches for informing progress and determining whether or not change is needed in management actions.

The Panel first describes key metrics for tracking viability of natural-origin populations of Chinook salmon and steelhead and their response to major management actions (Section 4.2). The need to track progress using stock-recruitment relationships that account for density dependence is emphasized, which is important even in depleted populations (Section 4.2.1). Next, a modeling framework for tracking productivity and abundance (Section 4.3.1) and for estimating population responses to management actions (Section 4.3.2) is presented. The Panel also provides an example of this approach by describing the response of juvenile Chinook salmon to environmental conditions in the Stanislaus River (Section 4.3.3, Appendix 9.5). Recommendations for tracking diversity and spatial structure are described (Section 4.4). Next, data requirements and limitations are briefly discussed (Section 4.5). The chapter concludes with recommendations for setting biological goals (Section 4.6). Answers to specific charge Questions and 19 additional questions focused on salmon are provided in Appendices 9.1 and 9.2, respectively.

4.2 Criteria for Evaluating Population Viability and Response to Actions

The VSP is typically described in terms of VSP criteria—abundance, productivity, spatial structure, and diversity (McElhany et al. 2000). Abundance and productivity are the two primary VSP parameters recommended for tracking the status and trends of salmonid populations and their response to restoration actions. Abundance is typically measured as the number of spawners, the number of fry, parr, and yearling smolts produced by those spawners, and the number of returning adult progeny. Intrinsic productivity determines the maximum rate of population growth per capita, which occurs when density is low. Intrinsic productivity depends on the product of reproductive output (number of eggs produced) and survival rates. For salmon and steelhead, it is quantified in units of smolts per spawner or adult recruits (returning salmon prior to fishing) per spawner (R/S). In the absence of harvest, a viable population is one with an average intrinsic productivity of adult recruits ≥ 1 whereas a population will decline when intrinsic productivity < 1.

Spatial structure and diversity of a population are also important VSP parameters that contribute to its stability, resilience and persistence (Hill et al. 2003, Schindler et al. 2010, Moore et al. 2014). Spatial structure describes the geographic distribution of the population or meta-population within or across watersheds, whereas diversity describes the extent of both genetic and phenotypic variation, including variation in life history types (e.g., fry, parr and yearling migrants for Chinook salmon; for simplicity, the term "smolt" in text below often refers to the sum of all out-migrating juveniles). Spatial structure and diversity typically reflect the diversity and extent of habitats that support the population (Waples et al. 2009, Rieman et al. 2015). Life history diversity is often more complex in Chinook salmon and steelhead than other salmonids and can be expressed as diversity in size and age at which juveniles emigrate from their natal watershed, residence time and habitat use in the estuary, as well as age and timing of return from the ocean. Life history diversity can also increase the capacity of a watershed or estuary to support a population to the extent that it enables the population to utilize a greater variety of habitat types (ISAB 2015). Both spatial structure and diversity contribute to viability of salmonids by spreading the risk of deleterious conditions or catastrophic events and by increasing resilience to changing environments.

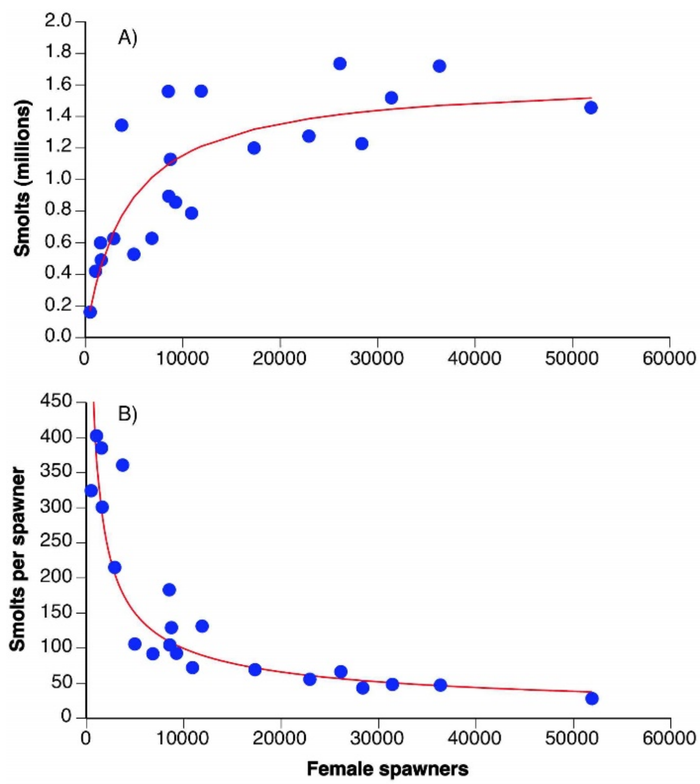
Although spatial structure and diversity are clearly important for maintaining population viability, development of biological goals in the CV should focus primarily on abundance and productivity of populations and secondarily on spatial structure and diversity. Abundance and productivity provide a more direct and immediate measure of population status and viability. They also are more direct measures for evaluating the response of salmonid populations to restoration actions in specific watersheds, including increased flow and habitat restoration, and they are more intuitive. Furthermore, time trends and annual variability in abundance and productivity reflect in part the extent of spatial and life history diversity of the population. Finally, determining the contribution of spatial, life history, and genetic diversity to productivity is challenging. For example, the relative contribution of fry, parr, and yearling smolts to adult returns of Chinook salmon and steelhead is rarely quantified because it requires specialized techniques to document the juvenile life history types represented in returning adults. In the absence of this information, the benefits of more or less diversity in outmigrants to the productivity of the population cannot be determined. Similar challenges exist with respect to monitoring and setting goals for spatial and genetic variation.

4.2.1 Density dependence

Density dependence occurs when a population's density affects its growth rate by changing one or more vital rates, including birth, death, emigration, or associated characteristics such as individual growth rate or age at maturation. Abundance and productivity of a population should be evaluated within a framework that incorporates density dependence because it can be strong, even for depleted ESA-listed species (ISAB 2015; See Box 4.1). Compensatory density dependence is critical to the resilience of populations because survival of individuals in the population is maximized when density is low. Furthermore, analysis of density dependence during specific life stages can inform restoration actions (ISAB 2015). Given these findings, the Panel recommends accounting for density dependence in the analysis rather than excluding values that may be influenced by density (i.e., the approach in SEP Group 2016).

BOX 4.1: Density Dependence Within Depleted Salmon Populations with Supplementation

Scientists often think density dependence does not constrain population growth at low population densities. This was the case in the Columbia River Basin when salmon were first listed for protection under the ESA (ISAB 2015). This belief supported an approach in the Columbia Basin to rebuild depleted salmon populations in part by supplementation with hatchery salmon. However, examination of findings and data throughout the Columbia River Basin revealed strong density dependence essentially everywhere that spawner/recruit data were collected. For example, as shown in the figure below, production of natural spring/summer Chinook salmon smolts in the Snake River Basin was strongly density dependent such that densities of more than 20,000 female spawners failed to produce more smolts (brood years 1990-2010). Furthermore, when female spawners exceeded ~6,000 fish, fewer than 145 smolts were produced per female, which is the minimum number needed for a sustainable population given the observed mean smolt to adult survival rate of 1.4% (i.e., 145 smolts/female * 0.5 females/spawner * 0.014 spawner/smolt = 1 recruit/spawner). A sustainable natural population could be achieved by reducing the number of hatchery-origin spawners (which are included in the female spawner count below), increasing smolts per spawner via flow increases or habitat restoration, and/or increasing smolt to adult survival by improving conditions during out-migration.



4.3 Tracking Salmonid Population Status and Trends and Responses to Actions

This section describes how a stock-recruit model can be used to track the status and trend of a population while accounting for density dependence. In the context of salmon and steelhead populations, a stock-recruit model describes the relationship between the number of parental spawners and the number of recruits they produce. Then, how the recruitment relationship can be used to test for the effects of management actions (e.g., flow, water temperature, habitat restoration) on productivity of the natural population while also considering effects of hatchery salmon is described. Abundance of spawners leading to maximum smolt or adult recruitment can be estimated from the relationship.

Basic salmonid fisheries data, such as displayed in a "brood table" (see example in Appendix 9.5), are needed to estimate species- and watershed-specific stock-recruit relationships. Ideally, data presented in a salmon brood table include parent spawner year, the number of spawners, and the number of progeny produced by those spawners. The number of parent spawners should be split into natural- and hatchery-origin spawners so that the proportion of hatchery origin salmon on the spawning grounds (pHOS) can be calculated (See Box 4.2). The number of progeny should include total number of outmigrating juveniles ("smolts") and/or total adults produced by the parent spawners. Total adults should be pre-fishery recruits, i.e., the progeny that are captured in fisheries (commercial, sport, tribal) plus fish that escape the fisheries and either spawn in their natal river or die prior to spawning. Salmonids mature at various ages, so age composition in the catch and escapement is needed to correctly allocate progeny back to their birth (brood) year.

4.3.1 Using stock-recruit relationships to quantify abundance and productivity

In this section, some simple mathematical models are presented to articulate how parental abundance and environmental factors control the status and trends of populations. Use of these models provides three benefits. They provide an explicit description of key aspects of population dynamics that clarifies relationships and assumptions. They can be transformed into statistical models to track the causes of changes in abundance and quantify the effects of flow and habitat enhancement on parameters that influence status and trends. Finally, these models highlight problems with experimental design and data that may challenge our ability to quantify effects of flow and habitat enhancement.

The rate at which the abundance of a population will change is determined by its intrinsic productivity. Assuming for simplicity no density dependence in survival rates, its influence on the growth of a salmon population is described by,

1) $R_{t,a=2\text{-}5} = S_t * \pi$

In the context of salmon, R is the number of returning adults prior to capture in ocean and river fisheries (recruitment), S is the number of parental spawners or "brood" spawning in year *t* that produced this recruitment, and π is the intrinsic productivity of the population. In this equation and the ones that follow, Greek letters denote variables that need to be estimated while Arabic letters denote variables for which data are available. Because Chinook salmon return as adults as early as two years (*a*=2) and typically as late as five years after hatching (*a*=5), the recruitment from any year *t* must be calculated as the sum of maturing progeny two to five years later as determined by age-specific catch plus escapement. For simplicity, equations which follow exclude *t* and *a* subscripts.

Equation 1 can be rearranged to demonstrate that the intrinsic productivity of a population is simply the ratio of total recruits to the number of spawners that produced them (i.e., π=R/S), but only if there is no density dependence. In the absence of harvest a population will increase in abundance over time if π>1 and decline in abundance if π<1. Larger π's are required to allow population growth in exploited populations. For example, if all age classes are exploited at a rate of 20% (approximate rate for winter- and spring-run CV Chinook salmon; Johnson and Lindley 2016) then productivity must be greater than 1.25 for population growth to occur. If the exploitation rate is 50% (approximate rate for fall-run Chinook salmon) then productivity must be greater than 2 for population growth to occur. Equation 1 ignores density-dependent effects on survival rate because π does not change with spawner numbers (S), but as we discuss below, it is essential and not difficult to include such effects via a stock-recruit model.

In the context of flow and habitat effects, it is logical to divide intrinsic productivity into a series of life stages that are influenced by different management decisions (e.g., tributary vs. Delta) as well as stages not affected by flow (e.g., ocean survival). For simplicity, consider the following two-stage model,

2) $R = S * \pi_{trib} * \pi_{deo}$

where π_{trib} is the intrinsic productivity from spawning to the life stage when the population outmigrates from a tributary (e.g., the ratio of smolts/spawner for juveniles that leave the tributary as smolts as determined by fecundity of females, sex ratio of spawners, and survival rates of early life and juvenile stages in the tributary), and π_{deo} is the survival rate from smolt to adult return, i.e., survival through the mainstem, Delta, lower estuary, and ocean that determine the number of adults produced per smolt. In the absence of harvest, the product of these elements must exceed 1 for the population to grow. As an example, increases in flow in a tributary may increase π_{trib} from say 50 to 100 smolts/spawner. However, this increase will not result in a positive trend for the population if π_{deo} is less than 0.01. Equation 2 can be re-arranged to define the minimum intrinsic tributary productivity required to sustain population growth given the combined Delta, estuary, and ocean survival rate as well an annual exploitation rate (U),

3) $\pi_{trib}>1/(\pi_{deo}*(1\text{-}U))$.

Incorporating density-dependent effects on survival into production relationships is critical because this process is ubiquitous among animal populations (Ricklefs 1982) and extensively documented for fish populations (Myers 2001) including Chinook salmon (Healey 1991, Rose et al. 2002). The Panel notes that there is little support for the approach to density dependence suggested in the Stanislaus recovery report (SEP Group 2016; see ISAB 2015 and BOX 4.1). It seems likely that reduced flow and habitat quality may have reduced the carrying capacity of streams and the Delta well below historical levels, so density dependence may now occur at abundances much lower than historical levels. Range contraction due to lack of access to habitat above dams and due to low returns may also result in density dependence occurring at levels much lower than historical abundance. Finally, the importance of density-dependent effects may increase in cases where abundance increases due to flow and habitat enhancement. Thus, a robust way of accounting for density dependence must be included in the assessment to test and quantify effects of flow and habitat restoration on the potential for a population to expand.

Use of stock-recruit relationships is a very common way of incorporating density-dependent effects in fish populations (Hilborn and Waters 1992, Myers 2001), and the Ricker stock-recruit model is often used to model the production relationship for Chinook salmon (Chinook Technical Committee 1999),

4) $R=S*\exp(\alpha-\beta*S)$

where α represents a density-independent rate that depends on the proportion of female spawners, their fecundity, and the survival rate from egg-adult return, and β

controls the magnitude of density-dependent effects. This equation results in a greater increase in R as S increases at small values of S, with diminishing increases in R with further increases in S due to increasing density-dependent effects (Figure 4.1). At very high levels of escapement, recruitment may decline due to over-compensatory mechanisms such as redd superimposition. As S approaches 0 (i.e., S->0) the productivity of the population increases because β*S->0, and maximum productivity (intrinsic productivity) in this form of the Ricker model is therefore calculated as e^{α} (the slope of the stock-recruit curve at the origin). If assuming that there is no density dependence in a population, then β=0 and the stock-recruit relationship is R=S*e^{α}, which is equivalent to Eqn. 1 with π from Eqn. 1 replaced with e^{α} (i.e., α represents intrinsic productivity π in log space). The 1:1 or replacement line commonly shown on stock-recruit plots represents the number of spawners required to produce sufficient recruitment to balance the population in the absence of harvest. The intersection of the replacement line and stock-recruit curve therefore represents the carrying capacity of the unexploited population. The spawner abundance that leads to maximum recruitment (the peak of the curve) is a useful metric to use as an escapement goal. Higher values of intrinsic productivity lead to an increasing space between the initial slope of the stock-recruit curve and the replacement line which results in an increase in the potential for population growth or exploitation (Figure 4.2). Conversely, the population will decline if the initial slope is less than the replacement line (i.e., < 1) and decline faster as the initial slope falls further below the replacement line.

Stock-recruit models can easily be modified to separate different life stages that may have different sensitivity to management actions. For example, a spawner-smolt recruitment model can be combined with a density-independent smolt-to-adult (SAR) survival rate to calculate a spawner-adult recruitment relationship (Figure 4.3). This particular structure follows the prevailing thinking that the majority of density dependence occurs in fresh water and that effects of population-specific density on survival in estuary and ocean environments are limited (Dorner et al. 2018, Riddell et al. 2013). In this case the replacement line for the tributary-specific spawner-smolt relationship is the inverse of the density-independent smolt-adult return survival rate. Say, for example, that tributary intrinsic productivity of a population is 150 smolts/spawner, and the smolt-adult return survival rate is 0.005. This latter value would lead to a replacement line with a slope of 200 smolts/spawner (1/0.005), indicating that the population will decline because this slope exceeds the freshwater productivity (150 smolts/spawner). In this two-stage example, the population can grow only if the slope of the tributary-specific production relationship is steeper than the replacement line. This can occur by either increasing production in the tributary (for instance by increasing tributary productivity to >200 smolts/spawner) or alternatively, by reducing the slope of the replacement line by increasing survival rates in the Delta. The greater the space between the initial slope of the production relationship and the replacement line, the greater the rate of population increase (if production line slope > replacement line) or decrease (production line < replacement line) over time.

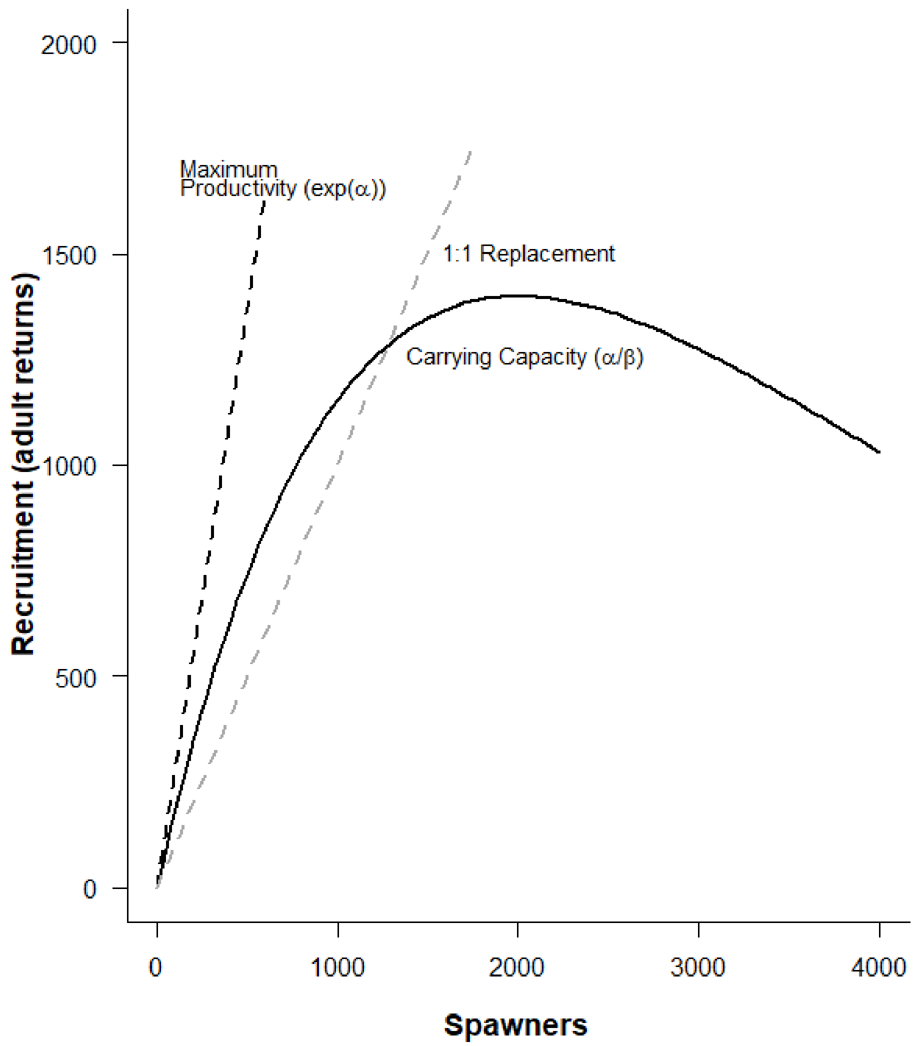


Figure 4.1. Example of a stock (escapement)-recruitment relationship (thick black curved line; equation 4). The initial slope of the curve (dashed black line) represents maximum (intrinsic) productivity and is calculated as e^a. The intersection of the stock-recruit curve and the 1:1 replacement line (dashed grey line) represents the carrying capacity of the population (escapement = recruitment) in the absence of harvest and is computed as α/β.

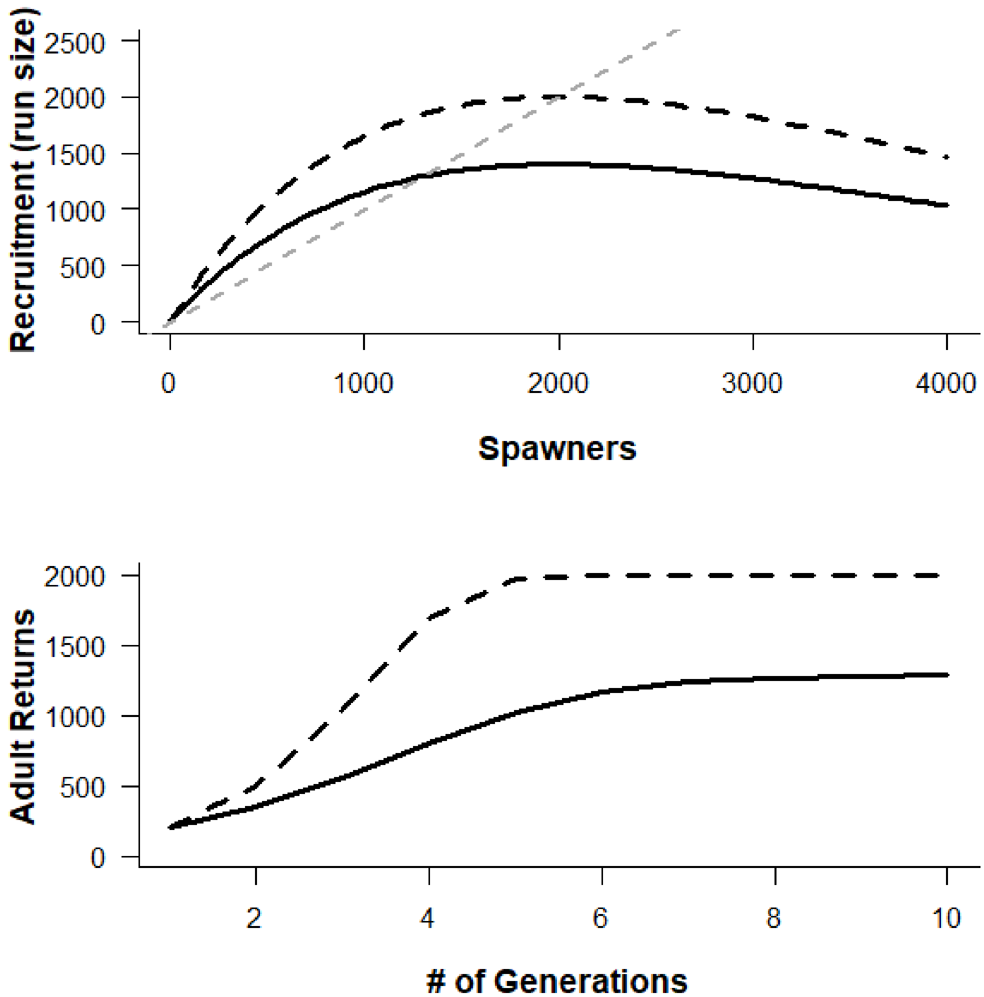


Figure 4.2. Comparison of two stock-recruit curves (top panel) where the intrinsic productivity for the base relationship (solid line) is increased by 40% (dashed line), and the resulting increase in abundance over 10 generations beginning with an escapement of 200 fish (bottom panel). This figure shows the importance of intrinsic productivity and density dependence on future abundance.

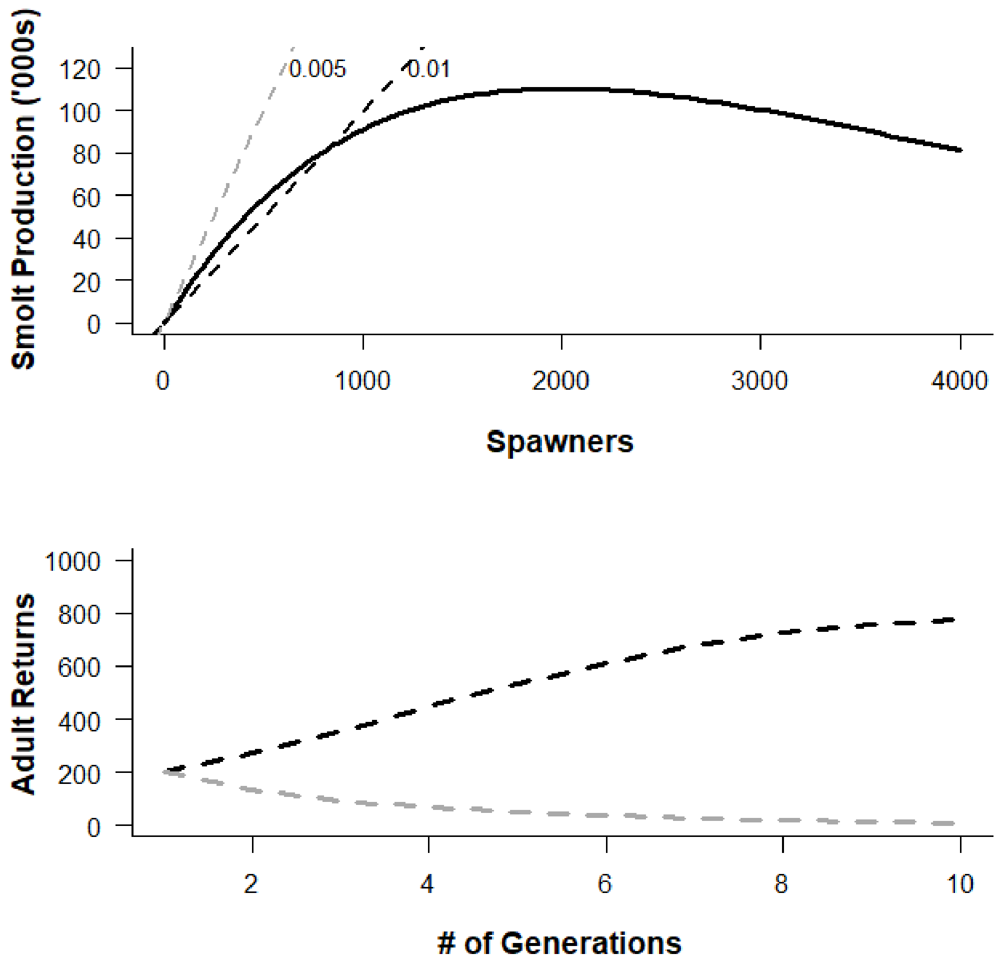


Figure 4.3. An example of a relationship between parental spawner abundance (escapement) and the resulting number of smolts produced (top), and predicted abundance trend of adult returns that depends on this recruitment curve and low and high smolt-adult survival (bottom). The freshwater stock-recruit model shown in the upper panel is based on a productivity of 150 smolts/spawner with replacement lines based on low (0.005, dashed grey line) and higher (0.01, dashed black line) density-independent smolt-adult return survival rates. Population trends in the lower panel begin with a starting population of 200 spawners. The population declines under the lower smolt-adult survival rate scenario because the product of spawner-smolt productivity and this survival rate, which represents the overall spawner-adult recruit productivity (calculated as 150 smolts/spawner*0.005 adults/smolts = 0.75 adults/spawner), is less than one. The population can increase in abundance under the 0.02 smolt-adult survival rate scenario as the spawner-recruit abundance (150*0.01=1.5) is greater than one. This relationship applies to any salmonid species where data are available.

4.3.2 Using stock-recruit relationships to quantify benefits of management actions

Incorporating effects of management actions into stock-recruit relationships is straightforward and is often done by treating these variables as additive effects on α. As an example consider,

5) $R=S*\exp(\alpha - \beta*S + \gamma*F)$.

Here, F is a covariate representing an index of flow or some other covariate at some point during the tributary incubation and/or rearing period, and □ represents the coefficient for the covariate (e.g., flow) effect. Stock-recruit parameters for a Ricker model can be estimated by standard regression techniques after transforming the model to linear form,

6) $\log(R/S) = \alpha - \beta*S + \gamma*F$

This relationship makes it obvious that $\gamma*F$ has an additive effect on α□□and that density-dependence is simply another term in the linear model. With this model form, the log of intrinsic productivity is expected to increase linearly with F (intrinsic productivity = $e^{\alpha+\gamma*F}$) and since carrying capacity is calculated as α/β, the carrying capacity is expected to increase linearly with F (i.e., capacity = $(\alpha + \gamma*F)/\beta$). It is possible to extend this model to include direct effects of flow on the density-dependent term (β) as well, but this complication is avoided for simplicity here.

This framework provides a statistical approach to quantify benefits of specific management actions. In the absence of information on juvenile production, equation 6 would be applied using tributary-specific spawner and adult return estimates. Different indices of F could be used to test alternative hypotheses about how flow (or other factors) during different times of the year affects recruitment. The value of γ would quantify the effect of changes in F, with larger positive values indicating a more beneficial effect of higher values of F.

4.3.3 Example application of the salmon stock-recruit covariate approach to spawner and juvenile data from the Stanislaus River

The Board asked the Panel if stock-recruit relationships, such as those described above, can be developed from existing monitoring programs and data collected in lower San Joaquin River tributaries. A stock-recruit covariate approach is important for setting and evaluating progress towards biological goals because it: 1) helps managers understand factors affecting productivity and abundance (goals); 2) provides information to quantify the benefits of management actions like flow increases to the goals; and 3) provides information about the effects of non-flow factors that may confound or limit the response of goals to a flow-based management action.

In Appendix 9.5, the Panel provides an example of how the stock-recruit covariate approach can be applied to existing juvenile abundance and adult spawner data for fall-run Chinook salmon in the Stanislaus River. In this example, parent spawners, flow, and water temperature were used to predict the annual number of parr and smolts passing the Oakdale rotary screw trap from 1997 to 2017. Average flows between Oct 1 - Dec 31, Oct 1 - Mar 31, Jan 1 - Mar 31, Feb 1 - Mar 31, and Apr 1 - May 31 were computed for each year to represent conditions during spawning, incubation, emergence, fry rearing and dispersal, and parr rearing and dispersal, respectively. Maximum daily water temperatures from the same flow gauge were averaged between Oct 1 and Nov 30 each year to represent conditions during the early part of the spawning and incubation period when water temperature can be elevated in some years. The data were fit to both Ricker and Beverton-Holt recruitment models. Weighted usable area (WUA)[9] estimates during spawning, incubation, and juvenile life stages were also used as alternate covariates.

This analysis demonstrates the utility of the stock-recruit covariate approach, leading to the following general observations:

- The stock-recruit covariate approach can be applied to Chinook salmon data in lower San Joaquin River tributaries, such as the Stanislaus River.
- Covariates such as flow and temperature can help explain variability in annual juvenile salmon production and may inform the utility of management actions.
- Juvenile production increased with greater flows and decreased with elevated water temperature.
- Predicted WUA of juvenile salmon habitat, which varies with flow, did not provide reliable predictions of juvenile production. Measured juvenile production decreased with increased WUA. This occurred because WUA is maximized at relatively low flows, and the spawner and juvenile data indicate that higher flows lead to higher juvenile production.

[9] WUA is defined as the total surface area having a certain combination of hydraulic conditions, multiplied by the composite probability of use by the species for that combination of conditions.

- Findings stemming from Beverton-Holt and Ricker models were similar.

- Decision-makers can use stock-recruit covariate relationships in three ways. First, they can use the models in a post-hoc analysis to estimate the contribution of a particular management action like flow on the biological goals of increasing productivity and abundance. Second, they can use the model to determine the covariate value (e.g., flow during the emergence period) needed to meet a specific biological goal (e.g., a doubling in juvenile production relative to current levels). Third, they can use the model to define a biological goal (e.g., "x" juvenile Chinook salmon from the Stanislaus River at Oakdale) based on anticipated flow or other covariate levels that can be achieved via a combination of management actions and natural variation. Furthermore, the model could be run under assumed projections of flow and temperature (e.g., most likely, optimistic, and pessimistic). This approach is appealing because the biological goal is estimated from data in the system of interest, and can incorporate model-based predictions of future flows and temperatures. This may be a better approach for setting goals compared to using less certain hypothesis about juvenile production that are determined, for example, by applying maximum survival rates estimated in other systems.

4.3.4 Extensions of the stock-recruit covariate model

A number of logical extensions to this model (equation 6) are possible. For example, if tributary-specific estimates of the number of outmigrating smolts are available in addition to estimates of total adult returns (prior to fishing), then parameters in,

7a) $\log(\text{smolts}/S) = \alpha - \beta * S + \gamma * F$

7b) $\text{adult recruits} = S * (\text{smolts}/S) * SAR$

can be estimated (SAR denotes smolt-adult survival rate). This formulation will likely lead to better estimation of γ than if estimated using only spawner and adult recruitment data because smolt data allows the procedure to remove variation caused by smolt-adult survival.

As in any statistical model, sufficient variation in independent variables (S and F) is required to estimate the coefficients that quantify their effects. Strong variation in F across some years in the time series (e.g., very wet vs. very dry years) will probably allow estimation of γ even if there are modest changes in flow due to management actions. Estimates of γ can in turn be used to predict the expected increase in productivity and carrying capacity that will result from increased flows. It may be that γ

is much greater than zero (i.e., a strong positive effect of flow), but that small increases in F within water years due to the increase in managed flows results in a relatively modest increase in predicted recruitment, as was the case in the Stanislaus River example for fall-run Chinook salmon (Section 4.3.3 and Appendix 9.5).

A highly relevant model that includes both flow and habitat effects is,

8) $\log(R/S) = \alpha - \beta*S + \gamma*F + \delta*H$

where H is an index of the increase in the number or area of restoration projects or a more complicated model-based statistic that incorporates ecosystem responses. It is not possible to accurately estimate coefficients for independent variables that vary together. In our example, contrast in F and H will likely stem from variation among years. If both F and H increase by similar amounts over the same time period, they will be strongly co-linear and coefficients γ and δ would not be identifiable. In this case only the combined effect of flow and habitat effects (e.g., F*H or F+H) can be estimated. However, owing to large expected differences in water volumes over some years, γ and δ are potentially identifiable. Given enough data and sufficient contrast in F, it may even be possible to estimate an interaction term between flow and habitat using,

9) $\log(R/S) = \alpha - \beta*S + \gamma*F + \delta*H + \kappa*F*H$

A positive value of κ would quantify the additional benefit of restored habitat at higher flows compared to lower flows, which may be a reasonable prediction in some settings (e.g., Yolo Bypass or unconstrained river reaches where floodplains and estuarine wetlands can be inundated).

Quantifying the independent variables for F and H effects could be done using a range of options. The simplest case would involve generating a purely discharge-based statistic from available data, such as the average flow during a springtime interval, or the amount of linear kilometers of stream or shoreline that have received a habitat restoration treatment. At the next level of complexity, more directed physical measurements or simple model-based output could be used. For example, to test the effect of water temperature on egg and alevin survival rate and its effect on productivity of winter-run Chinook Salmon in the mainstem Sacramento River, daily water temperatures could be used as input to existing incubation survival models (Martin et al. 2017) to predict survival rate that in turn would be used to represent F. With respect to habitat, H could be a constructed index that would depend not only on the amount of restored habitat but also on differences in observed juvenile densities in restored and un-restored shoreline areas. At the most complex level, results from ecosystem-type studies could be synthesized in a model to predict relative changes in F or H. For example, quantification of improvements in growth or survival rates of juvenile Chinook salmon using restored habitats like the Yolo Bypass could be combined with hydrologic records to calculate a time series of annual floodplain or wetland effects on salmon growth by using this metric to represent F or H.

4.3.5 Accounting for negative effects of hatchery-origin spawners on natural production

Hatchery-origin spawners often make up a large proportion of the total Chinook salmon escapement in CV tributaries, especially for fall-run Chinook salmon (Figure 4.4; Palmer-Zwahlen et al. 2018, Willmes et al. 2018). Spawner-recruit relationships should ideally account for hatchery-origin fish that spawn in the wild and recruitment (R) values must exclude hatchery-origin fish from total recruitment. As reviewed in Box 4.2, there is a considerable and growing body of evidence showing that productivity of hatchery-origin salmonids spawning in the wild is considerably lower than that of wild-origin fish. For populations where relative reproductive success is well-defined, the effect of hatchery spawners on overall productivity can be incorporated into a stock recruit model using,

10a) $R=S'*\exp(\alpha-\beta*S')$

10b) $S' = S*f(PNI)$

where S' is the adjusted number of spawners, PNI[10] is the proportionate natural influence, and f(PNI) is a function that predicts the reproductive success of the integrated population as a function of PNI (see Box 4.2). If only natural-origin spawners return to a tributary, f(PNI)=1 and therefore S'=S. At the other extreme, where all broodstock used in the hatchery was derived from hatchery production, and all spawners in the wild were also derived from the hatchery, PNI would be zero and f(PNI) would be low, but likely not zero.

The Panel recognizes that this modeling approach is an over-simplification of wild-hatchery interactions. While PNI is readily quantifiable through escapement surveys and hatchery broodstock data, f(PNI) can only be reliably estimated in systems where

[10] PNI = pNOB/(pNOB + pHOS), where pNOB = proportion of broodstock that is natural origin and pHOS is proportion of spawners in the wild that are hatchery origin.

relative reproductive success studies have been conducted in relation to PNI values (see Box 4.2).

The use of f(PNI) values stemming from other watersheds is cautioned. This could bias estimates of the number of effective spawners (S') and, therefore, intrinsic productivity, if the borrowed PNI values are not representative of the target population. For example, overestimation of the adverse effect of low PNI would lead to artificially low S' and estimates of intrinsic productivity that are too high. In other words, this bias could incorrectly lead one to believe the population was viable when in fact it was not.

In absence of f(PNI) values for specific populations, the Panel emphasizes that variable PNI and pHOS values can confound evaluation of management action effects on salmonid populations. For example, if spawning populations that receive a higher flow treatment also have variable PNI and pHOS over time and do not respond to the management actions, it will be difficult to determine whether the lack of response was due to hatchery-origin fish or because the flow action did not have a substantial effect. Low PNI, high pHOS, and high annual variability in these metrics is a major limitation in evaluating potential benefits of flow and habitat management actions.

BOX 4.2: Interactions of Hatchery and Natural-Origin Salmonids

Salmon produced in hatcheries frequently spawn in streams and interbreed with natural-origin fish. This is the result of intentional supplementation efforts to increase spawning abundances or from the inability of fisheries to harvest surplus hatchery fish. Most studies indicate hatchery salmonids spawning in the wild reduce productivity of the overall population. For example, in the Pacific Northwest, intrinsic productivity of 30 Chinook salmon, 22 coho salmon, and 18 steelhead trout populations declined with increases in the average proportion of hatchery-origin spawners (pHOS; Chilcote et al. 2011, 2013). Across watersheds, intrinsic productivity of Chinook salmon declined 75% as pHOS increased from 10% to 60%. Parentage-based tagging also reveals that hatchery salmon spawning in streams have lower reproductive success compared with natural-origin spawners (Araki et al. 2008, Ford et al. 2016). A review of fitness studies using hatchery salmon produced from wild-origin brood stock revealed a ~50% decline in fitness relative to their wild counterparts (Christie et al. 2014). Hatchery salmonids also exhibit reduced genetic diversity (Bingham et al. 2014, Christie et al. 2016).

A key question is the extent to which these adverse hatchery effects reflect genetic factors that carry over to subsequent generations (i.e., a long-term effect) or are due to behavioral differences that are not heritable and would therefore not carry over to the next generation (e.g., hatchery-origin fish selecting poor spawning habitat) or a combination of both. Few studies have teased apart genetic versus behavioral effects, but Araki et al. (2009) demonstrated a carry-over effect of hatchery reproduction to multiple generations of progeny spawning in the wild. Unplanned experiments show that salmon productivity can increase following cessation of hatchery production (Buhle et al. 2009, Jones et al. 2018). Lastly, although compensatory density dependence is a key mechanism for providing resilience to declining salmonid populations (see Box 4.1), the negative effect of hatchery supplementation on density dependence of natural origin fish is rarely considered (ISAB 2015).

The effects of a hatchery on a naturally spawning population depend on hatchery practices and differences in selective pressures in wild and hatchery environments. Programs that use hatchery-origin adults as broodstock, and where many hatchery-origin fish spawn in the wild, have the largest negative effects on the naturally-spawning populations (i.e., low PNI). Programs that use only natural-origin spawners as broodstock and minimize the number of hatchery-origin fish spawning in the wild, will have the smallest negative effects on wild populations (high PNI; e.g., Berejikian and Doornik 2018). With proper genetic monitoring, hatchery programs have the potential to maintain severely depleted populations while restoration actions are being implemented. However, given the insidious and rapid effects associated with hatchery production (Christie et al. 2016), supplementing wild populations with fish of hatchery origin should not be regarded as a permanent solution. The demonstrated substantial negative effects of hatchery-origin fish on naturally spawning populations means that the contributions of hatchery salmonids to spawning escapement and adult returns must be carefully monitored. This would enable more accurate assessment of natural-origin salmonid responses to management actions (Naish et al. 2008).

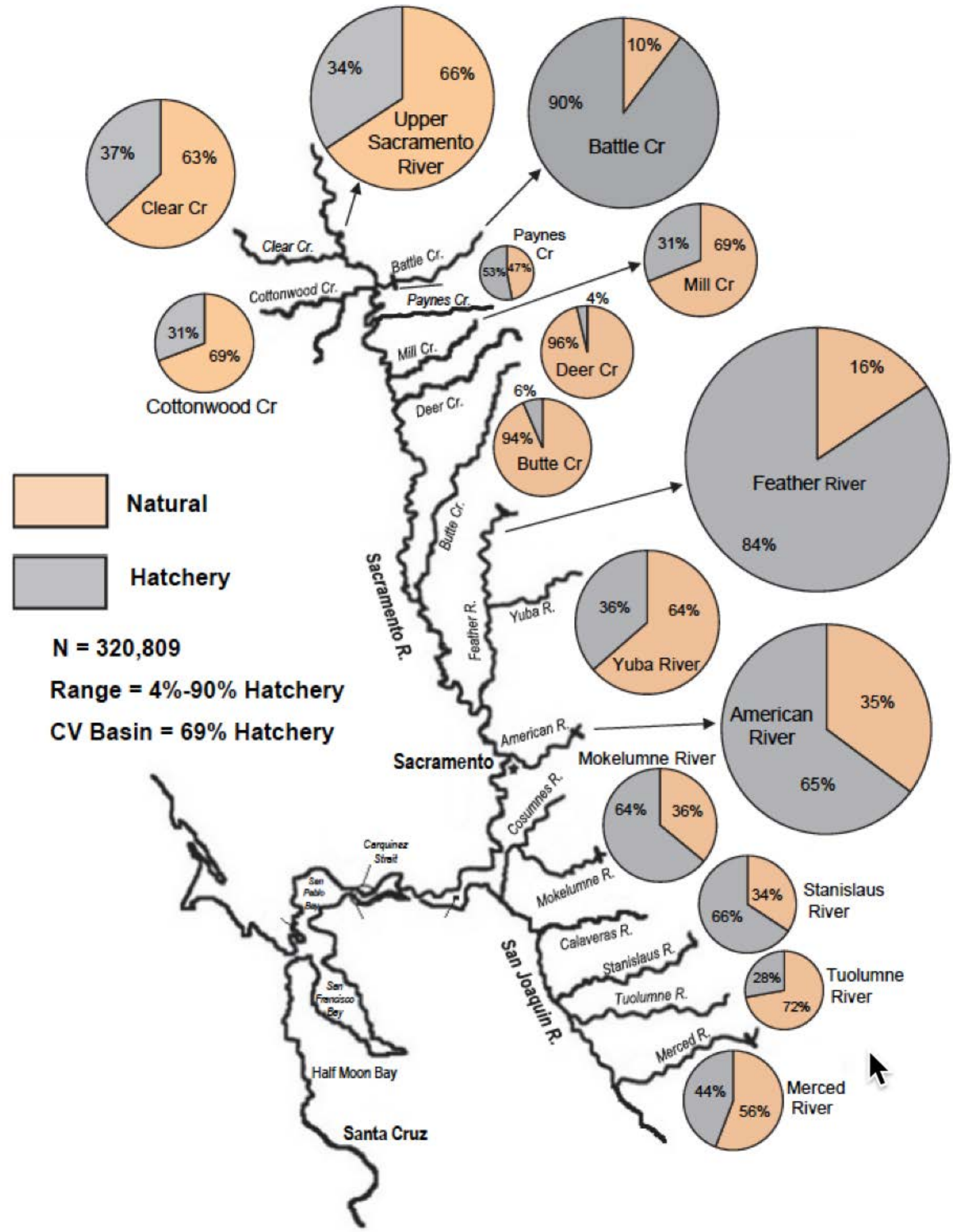


Figure 4.4. Proportion of natural and hatchery origin fall Chinook salmon on Central Valley spawning areas, 2013. Source: Palmer-Zwahlen et al. (2018).

4.4 Tracking Diversity and Spatial Structure

Salmonid diversity includes genetic composition and phenotypic expressions such as life history diversity. The proportion of hatchery spawners in a population is a key metric for tracking genetic composition of the natural spawning populations because artificial propagation tends to homogenize and alter the genetic composition of the natural population from which it is derived (Williamson and May 2005, Bingham et al. 2014). Ideally, the pHOS should be as low as possible, especially when few natural-origin fish are included in the broodstock (Lindley et al. 2007)[11]. CA HSRG (2012) recommends that pHOS be <5% when the natural population is not "integrated" with the hatchery population. For "integrated populations" where some hatchery fish are expected to spawn in rivers, CA HSRG recommends a PNI that exceeds 0.50 which can be achieved, in part, by increasing the proportion of natural-origin spawners in the broodstock. In contrast, the HSRG in the Pacific Northwest recommends a PNI of 0.67 or higher (HSRG 2014). Importantly, PNI values represent a means for tracking hatchery practices and adverse effects of hatchery salmonids on the natural population.

Age composition of adults and emigrating juveniles is another important measure of life history diversity. It is also needed to construct the brood table that is used to track productivity and abundance of progeny produced by parent spawners, as described above.

Diversity of juvenile life history types among Chinook salmon enables the species to more fully utilize available habitats, potentially increasing capacity. Diversity also reduces the risk of severe reductions in abundance due to catastrophic environmental conditions in specific habitats. Percentages of fry (~<55 mm), parr (~56-75 mm), and yearling smolts (>75 mm) captured in traps can be documented (see Miller et al. 2010) and compared over time with restoration actions (e.g., creation of shallow rearing habitats occupied by fry and parr in the Delta). Essentially all fry migrants are natural origin because hatcheries rarely release juveniles of that small size. Contribution of fry and parr to adult returns can be evaluated by assessing the frequency of these life history types in adult Chinook salmon, as described by Miller et al. (2010), who documented unexpectedly large contributions of fry migrants (20%) in adult fall Chinook salmon by analyzing otoliths.

[11] Except for populations where a conservation hatchery is intentionally maintaining the spawning population (see CA HSRG 2012).

Rainbow trout (*Oncorhynchus mykiss*) produce both resident and anadromous (steelhead) life history forms, whose frequency of occurrence depends on genetic composition and environmental factors (Courter et al. 2013, Kendall et al. 2015). Nearly all natural spawning steelhead populations in the CV are strongly influenced by large numbers of hatchery-origin steelhead, and existing data are reportedly insufficient to evaluate the viability of naturally spawning populations (Lindley et al. 2007, Johnson and Lindley 2016). Nevertheless, some evidence indicates steelhead are becoming rare in areas where they were once abundant, indicating this important component of life history diversity is being lost (McEwan 2001, Lindley et al. 2007). A shift toward resident trout has occurred below dams presumably in response to hypolimnetic releases from reservoirs that provide year-around, highly productive habitat. Although resident forms of rainbow trout may contribute to natural steelhead viability, especially at low densities, it is more common for steelhead to give rise to rainbow trout (Johnson and Lindley 2016). All the methods we describe above can be applied to steelhead if the data are available.

Quantifying spatial structure typically involves identifying populations within the overall meta-population that form an ESU or Distinct Population Segment (DPS). Lindley et al. (2007) documents the historical spatial structure of spring- and winter-run Chinook salmon and steelhead in the CV and describes the extent to which it has changed, largely in response to construction of dams that block migration to headwater habitats and caused extinction of populations. Changes in spatial structure may occur gradually over time or could change rapidly in response to catastrophic events such as a volcanic eruption (e.g., Mount St Helens in Washington state), forest fires, or construction of dams. Although National Oceanic and Atmospheric Administration (NOAA) fisheries scientists typically evaluate spatial structure across the entire ESU or DPS, it can also be described at a finer scale such as across the mainstem and tributaries within a watershed. A population with a broader spatial distribution in a watershed is less vulnerable to catastrophic events.

4.5 Estimating Status, Trends, and Management Action Effects

Quantifying status, trends, and flow and habitat enhancement effects on CV salmonid populations will depend on the availability and quality of data, sample size, and the magnitude of the restorative action in relation to natural variability. This section briefly describes data requirements, uncertainties in the data that will lead to uncertainty in VSP parameters, and summarizes key challenges in estimating these parameters.

4.5.1 Data

Spawning escapement. Escapement of spawners to each tributary is a key VSP parameter and an important metric for tracking status and trends. Escapement can be monitored at weirs or by automated counters (resistivity counters, acoustic counters) at fixed locations, or by repeat counts of spawners or redds using visual survey methods in spawning habitat. The latter methods tend to be less accurate and precise but have the advantage of providing information on spatial diversity within a watershed. Direct handling of returning spawners is needed to determine if they originated from a hatchery (see below). Bergman et al. (2012) describe the monitoring plan for Chinook salmon spawning escapement in tributaries of the CV. The objectives of this plan are to improve estimates of the number of Chinook salmon that spawn in streams, to provide estimates of accuracy and precision, and to estimate age, length, sex and hatchery/natural composition of each Chinook run when possible. This plan supports the development of brood tables that are needed to estimate stock-recruit relationships (see below and Appendix Section 9.5 for example). Enumeration of steelhead spawners is largely insufficient to evaluate trends in natural production but improvements have occurred in some watersheds (Lindley et al. 2007, Johnson and Lindley 2016).

Proportion of hatchery-origin spawners. As noted in Box 4.2, large numbers of hatchery-origin salmonids spawning in the wild can reduce intrinsic productivity of the naturally spawning population and will inhibit the ability of the natural population to survive better at low densities (a key compensatory response that can provide population resilience when abundance is low). Conversely, a conservation hatchery approach (CA HSRG 2012) may help maintain a population, at least temporarily, if productivity is so low that the natural origin component is not viable. The lack of annual pHOS estimates will limit the ability to separate hatchery and flow-habitat restoration effects on natural spawning populations. pHOS has recently been estimated in CV watersheds using coded-wire tags (CWT) and otoliths (Palmer-Zwahlen et al. 2018). For example, in the Feather River, hatchery Chinook salmon represented 55% to 90% of the fall-run spawning population (2002-2012), and pHOS was especially high in years when returns of natural-origin Chinook salmon were low (Willmes et al. 2018). In 2013, the estimated pHOS of fall-run Chinook salmon throughout surveyed watersheds in the CV was 69% (Figure 4.4; Palmer-Zwahlen et al. 2018). These field estimates of pHOS are considerably higher than those assumed in the doubling goal analysis (e.g., 40%; https://www.fws.gov/lodi/anadromous_fish_restoration/afrp_index.htm). More comprehensive estimates of pHOS for spring and winter-run populations are needed (Johnson and Lindley 2016). GrandTab is an excellent compilation of Chinook salmon spawners in the CV but it does not separate hatchery- and natural-origin salmon on the spawning grounds ("in-river") and it does not include values by age (Azat 2018). In addition to pHOS values, Proportion of Natural-Origin salmon in the hatchery Broodstock (pNOB) values in hatcheries are needed to calculate PNI values (see Section 4.3.5).

Population-specific catch estimates for natural- and hatchery-origin salmon and steelhead are needed to provide robust estimates of abundance and productivity of the population prior to harvest by commercial, Tribal, and sport fisheries. The ocean exploitation rate for winter-run Chinook averaged 16% between 2000 and 2013, and an abundance-based harvest control rule specifies maximum allowable exploitation rate of 12.9% to 19% (Johnson and Lindley 2016). The ocean exploitation rate for fall-run Chinook averaged 45% between 2011 and 2014. Ocean exploitation rates for spring-run Chinook are not available but are expected to follow trends similar to fall Chinook salmon and are likely lower, owing to their earlier return timing that reduces exposure to later fisheries (Johnson and Lindley 2016). Ocean harvest of CV steelhead is extremely rare and considered an insignificant source of mortality (Johnson and Lindley 2016).

CDFW uses CWTs to estimate hatchery-specific contributions of Chinook salmon to harvests, but we are not aware of attempts to allocate harvests of natural-origin fish in the ocean and lower mainstem to their natal streams (Barnett-Johnson et al. 2007, Kormos et al. 2012, Palmer-Zwahlen et al. 2013, 2018). Parentage-based tagging (Beacham et al. 2019), CWT data, and run reconstruction methods (e.g., USFWS 2001 (Appendix A), English et al. 2007,) could be used to identify hatchery salmon and allocate harvests of natural origin salmon (by age) to their natal streams in the CV. Run reconstructions of CV Chinook salmon to specific tributaries have been performed using a variety of assumptions; however, a key short-coming of these efforts is separation of natural-origin salmon from total salmon (Kope and Botsford 1990, Mills and Fisher 1994, USFWS 2001 (Appendix A), Mesick et al. 2009).

Age composition of returning salmonids is needed to assign harvest and escapement back to the correct brood year so that annual year class recruits can be calculated. Age composition of CWT hatchery salmon is available in recent years (Palmer-Zwahlen et al. 2018), but hatchery salmon tend to mature earlier than natural salmon and may not represent the age structure of the latter group. Thus, age composition of natural-origin salmonids should be collected every year in both fisheries and on the spawning grounds. Likewise, age composition of Chinook salmon differs in sport versus commercial fisheries and so age estimates are needed for both fisheries (Mesick et al. 2009). Zabel and Levin (2002) report that recruitment models based on averaged age composition can lead to substantial overestimation of productivity. Bergman et al. (2012) identifies a plan to estimate age, length, and sex composition of each Chinook run in each tributary. A summary of age composition for natural-origin salmon across all CV tributaries and over time is needed.

<u>Chinook run identification</u> (spring, fall, late-fall, winter-run) is needed to assign Chinook salmon to the appropriate run type (Johnson et al. 2017). For adults, Bergman et al. (2012) briefly note the methodology, which often involves professional judgment based on spawning time and the physical appearance of carcasses. Separation of spring-run and fall-run Chinook salmon in the Feather River has been difficult leading to less accurate statistics (Johnson and Lindley 2016). In some locations, such as Battle Creek, genetic analysis of tissue samples is used to estimate the proportion of each run in the escapement. Historically, run assignment for juvenile Chinook salmon has been determined using a length-by-date approach which is unreliable. For example, over 50% of individuals classified as winter-run by this method were either spring, fall, or late-full run types (Harvey et al. 2014). Ideally, single-nucleotide polymorphism (SNPs) genetic data should replace the length-by-date method for assigning run type to juveniles used to quantify freshwater production (Meek et al. 2016).

<u>Brood tables</u> should be developed for natural-origin salmon returning to each tributary where biological goals will be developed. A brood table shows the annual number of parent spawners and the number of adult progeny produced by those parents (see Appendix 9.45). Critical to the construction of brood tables in the CV is the separation of returning natural-origin salmon from hatchery salmon. Data described above are used in a run reconstruction process to create the brood table (Kope and Botsford 1990, Mills and Fisher 1994, English et al. 2007, Mesick et al. 2009).

<u>Tributary outmigrant estimates</u> are required to establish juvenile-based stock-recruitment relationships (e.g., equation 7a; Johnson et al. 2017). Rotary screw traps and mark-recapture methods are widely used to estimate the abundance of migrating populations of Chinook and steelhead fry, fingerlings, and smolts. These trapping programs can be expensive because they need to be run for many months to capture the full duration and suite of outmigrant life history types, and large sample sizes are needed to provide reasonably precise population estimates. Other challenges include distinguishing run type for outmigrating Chinook salmon, and separating hatchery- and natural-origin juvenile Chinook salmon given that approximately 25% of hatchery-origin fall-run fish are marked (but many hatchery fish are now released into the estuary rather than at the hatchery, especially during drought years). Juvenile hatchery- and natural-origin steelhead are reliably distinguished because 100% of hatchery steelhead have been clipped since 1998 (Johnson and Lindley 2016).

Developing a single index of freshwater production for Chinook salmon and steelhead to estimate a juvenile stock-recruit relationship (e.g., equation 7a) is challenging owing to the variety of migrating juvenile life history forms. For example, simply summing estimates of abundance of fry, parr, and yearling migrants as an index of freshwater recruitment assumes that each life stage contributes equally to adult recruitment, which is unlikely to be the case. Quantifying the contribution of each life history type to adult recruitment can be accomplished using otolith analysis (e.g., Miller et al. 2010) and may be possible in a few intensively studied tributaries.

Estimating survival of juveniles through the Delta is critical for evaluating whether higher flows and habitat restoration are increasing survival and whether very low survival in the Delta is limiting the response of adult recruitment to increases in flow and habitat in tributaries and the Delta (Johnson et al. 2017). Extensive acoustic tagging studies of Delta survival for larger hatchery-produced juvenile Chinook have recently been brought together in some excellent synthetic analyses (Buchanan et al. 2018, Perry et al. 2018), as have CWT-based studies that are capable of measuring survival rates for smaller hatchery-produced outmigrants (Newman 2008b, Newman and Brandes 2010). Perry et al. (2018) found that survival rates of late-fall hatchery-origin juvenile Chinook salmon in the Sacramento River were positively related to inflow, but only in reaches that transitioned from bidirectional tidal flows to unidirectional flow (as flow increased). They found that survival rates ranged from ~50 to 75% between Freeport and Chipps Island as flows increased from low values to ~30,000 cfs. These results suggest that enhanced flow and habitat in tributaries combined with higher flows through the Delta have the potential to improve overall productivity. In contrast, Buchanan et al. (2018) found that survival of fall-run hatchery-origin juveniles in the San Joaquin portion of the Delta (Mossdale to Chipps Island) ranged from 0-5% in low flows years and was only 2% in a high flow year (2011). These results suggest that: 1) increased flows alone may not increase survival rates through the southern portion of the Delta (more release experiments at higher flows and different levels of export are needed to better support this conclusion); and, 2) very modest increases in flow in the southern Delta from modest increases in tributary flow are unlikely to improve survival rates to the point where gains in production of juveniles from tributaries will lead to substantial increases in adult recruitment. In the context of our quantitative model described above, π_{deo} may be so low that the product of π_{trib} and π_{deo} will lead to an intrinsic productivity less than one, even if survival rates in the natal tributary are increased via increases in flow and habitat. Survival rates of hatchery steelhead from Mossdale to Chipps Island ranged from 25 to 75% (Buchanan 2013), which are higher than values for Chinook salmon possibly because hatchery-origin juvenile steelhead are larger than Chinook salmon. The value of applying survival rates of tagged large hatchery salmonids to smaller natural-origin salmonids is uncertain.

4.5.2 Estimation

Accuracy and precision of monitoring programs for escapement, juvenile production, and some of the other data listed above, will determine how well status and trends are estimated. Higher monitoring effort, which generally involves greater costs, usually results in more robust estimates of escapement, juvenile and adult recruitment, and other supporting data, and therefore less-biased and more precise estimates of status and trends and responses to restoration actions. Accurate estimation of hatchery-origin salmon (e.g., fall-run) in harvests and spawning escapements is critical for estimating production of natural-origin salmon, and mass marking (or tagging) hatchery salmon could improve these estimates (Cal-Nev AFS 2009, CA HSRG 2012, Mohr et al. 2017).

Sample size is an important factor to consider in estimation of fish responses to flow and habitat enhancement. Precision of the estimate is positively related to the number of years of data. Bradford et al. (2005) found that monitoring had to be conducted over a period of 4-6 generations (20-30 years for a Chinook population with a maximum age-at-return of 5 years) given a substantial increase in productivity from the flow or habitat treatment (e.g., 50% increase) and reasonably precise estimates of abundance (CV=20%). More precise monitoring can shorten this duration but only by a small amount. The need for long periods of monitoring is largely driven by the amount of interannual variation in abundance which depends on factors such as marine survival rates, which are mostly beyond management control. Sample size requirements for the stock-recruit approach advocated in this chapter would be larger than for before-after comparisons of abundance, because effects of both parental stock size and flow-habitat relationships must be estimated. Other metrics, such as acoustic tag-based estimates of survival in the Delta, provide managers with a more immediate assessment of potential flow and habitat restoration benefits. Similar efforts could be conducted in tributaries, and alternate approaches could be used to measure survival for fish that are too small to tag. These metrics are appealing, but there is considerable uncertainty in translating them to an overall effect on the population. For example, the survival rate of larger hatchery-origin fish that are tagged may not provide a good representation of survival rates for smaller natural-origin fish.

Sufficient contrast in stock size and flow/habitat effects is required to estimate effect sizes using a stock-recruit modeling approach. The magnitudes of the latter effects are a function of the volume or timing of flow releases and the size and number of habitat construction projects. There is large variation in flow among water year types. Thus, even if flow changes due to an increase in managed flows within-water year types is small, the large contrast in flows across water year types should allow estimation of a flow effect. The flow-effect parameter can then be used to quantify the benefit of the increase in managed flow. However, if critically dry and dry year water types become the norm due to effects of climate change, there may be limited variation in flow over time and hence limited ability to estimate flow effects. Similarly, habitat effects are likely to scale with the area or linear distance restored or modified relative to the area or distance over which survival is measured. Therefore assessment could be focused on areas with the largest such modifications which are likely to show the greatest contrast with unmodified areas. Any detected effects could be scaled down to locations with smaller modifications.

Evaluating effects of spawners on recruitment is at least as challenging as for flow. Spawner abundance will depend on past survival rates including those in the Delta and the ocean, as well as on exploitation rates. These factors are either largely beyond management control or may be difficult to change. As described in the modeling section of this chapter, the number of effective spawners is also strongly influenced by hatchery operations that affect pHOS and the relative reproductive success of hatchery-origin fish that spawn in the wild. Hatchery operations therefore not only affect the extent of contrast in spawner abundance over time, but also the ability to accurately estimate this abundance (given uncertainty in pHOS and the lower reproductive ability of hatchery-origin fish). The Panel believes that pHOS is one of the greatest challenges in the calculation and interpretation of VSP parameters that account for density-dependent effects via stock-recruit models.

Statistical approaches for modeling stock-recruit data are well established (Hilborn and Walters 1992). The modeling component of this chapter describes how to include management action covariates directly into a stock-recruit relationship. Status and trends in productivity can also be evaluated using state-space approaches with or without flow and habitat covariates (e.g., Fleishman et al. 2013, Staton et al. 2017). State-space models account for and clarify sources of uncertainty in the input data and parameters and are less prone to bias resulting from serial correlation in input data that can affect traditional stock-recruit analyses. Most importantly, this approach allows for annual estimation of intrinsic productivity and carrying capacity so that managers can examine temporal trends in these key VSP parameters independent of covariate effects involving management actions. This latter feature will be useful if covariate effects are small.

4.6 Recommendations for Setting Biological Goals

VSP criteria (abundance, productivity, diversity, and spatial structure) are well-established metrics for evaluating populations. The Panel recommends that VSPs form the basis of biological goals for CV Chinook salmon and steelhead populations. Abundance and productivity are the most important and intuitive metrics for setting biological goals. Diversity and spatial structure are keys to population resilience in a variable environment such as that anticipated under climate change scenarios.

The recommended framework for evaluating abundance and productivity of natural-origin populations in this report accounts for density dependence in survival rates by using a stock-recruit relationship (Sections 4.2.1, 4.3.1). This approach is needed because density dependence occurs even at low salmon densities, compensatory density dependence provides resilience and stability during periods of declining abundance, and density dependence during specific life stages can inform restoration actions (ISAB 2015). This framework, including the approach to quantify benefits of management actions, represents both an expansion and improvement of the approach identified for the Stanislaus River (SEP Group 2016). Although this earlier effort was comprehensive and covered many aspects of VSP, it did not formally consider density dependence and it did not provide a quantitative framework for evaluating the response of salmonid populations to major management actions. For comparison, Appendix 9.2 briefly describes development of biological goals for Chinook salmon in Puget Sound, Washington. These recommendations can be applied to any salmon or steelhead population if data are available.

4.6.1 Setting productivity goals

- <u>Adult recruits/spawner (R/S) > 1 (intrinsic productivity before harvest)</u>. This is an obvious initial goal because it represents the lower limit for a viable population (R/S = 1). Pre-fishery recruits (i.e., catch plus spawner escapement) are used here. The advantage to using this goal (versus post-harvest R/S) is that it excludes harvests rates and therefore provides a more accurate description of progress related to management actions in the tributaries and estuary. Reasonably accurate population-specific harvests of natural-origin salmon are needed. Incorporating covariates (flow, habitat, etc.) in the recruitment models (adult or juvenile), as described in Section 4.3.3, can improve estimates of intrinsic productivity.

- <u>Adult recruits/spawner (R/S) > 1 (intrinsic productivity post harvest)</u>. This is an obvious goal because it represents the lower limit for a viable population after fish are removed in fisheries. Only recruits to the spawning grounds are used here (catch excluded). This goal reflects the fact that harvest rates are high and variable from year to year, management typically does not have targets for natural-origin Chinook salmon spawning in specific watersheds (e.g., fall run), and allocation of harvests to specific populations can be uncertain. This metric is less suitable for estimating benefits of management actions than the pre-fishery metric because variable harvest rates contribute to variable recruitment to the spawning grounds. Nevertheless, this metric provides a somewhat simple approach for estimating viability after fish have been removed by fishermen. This R/S goal will be more difficult to achieve than the pre-fishery R/S goal because harvest rates on natural-origin salmon can be high (Section 4.5.1).

- Intrinsic productivity during the spawner-to-smolt stage sufficient to produce a viable population after considering survival during the smolt-to-adult stage (see Section 4.3.1 and Box 4.1). Tracking the spawner-to-smolt stage is advantageous because benefits of management actions will be easier and potentially quicker to detect (Section 4.3.3, Appendix 9.5). However, trends in productivity over time may be easier to track than the specific productivity needed to support a viable population because trends do not require smolt-to-adult survival estimates that are needed to estimate viability (see next).

- Intrinsic productivity increase over time. A state-space approach can be used to determine whether conditions are improving (Peterman et al. 2003, Peterman and Dorner 2012). The approach is recommended for both spawner-to-smolt and spawner-to-adult stages. The recommended covariate approach can help reduce variability that is common to recruitment relationships, thereby providing more accurate and precise estimates of productivity. Importantly this approach does not require assumptions about the desired end point stemming from management actions. Rather, it simply tests the assumption that management actions are having the assumed and desired positive effect over time.

- Annual R/S > 1. This is a simple metric for tracking viability in addition to intrinsic productivity metrics described above. This common metric shows whether a natural-origin population heavily supplemented with hatchery spawners is sustainable when hatchery fish are removed from adult returns (i.e., natural fish only). This approach does not require modeled estimates of intrinsic productivity and is easily calculated directly from a brood table (Appendix 9.4). The Panel recommends both recruits estimated prior to the fishery (catch plus escapement) and fish returning to the spawning grounds. In this metric, S includes both natural- and hatchery-origin spawners.

4.6.2 Setting abundance goals

- Number of natural-origin adults. The specific goal can be set by stakeholders after removing hatchery-origin fish from total returns. Goals should be realistic (see Box 4.3 on doubling goals). While abundance targets may be desirable to stakeholders, the Panel believes that a positive trajectory over time is an important alternative.

- Number of spawners leading to maximum production of juveniles and/or future adults, on average. These values can be estimated from stock-recruit relationships (Hilborn and Walters 1992) and can incorporate management covariates such as flow, as described in section 4.3.3 and Appendix 9.5. This approach uses empirical data to inform the development of biological goals that may be achieved via a combination of management actions and natural variation. Spawner values can include hatchery-origin fish, which will reduce overall productivity (Box 4.2).

- Number of spawners leading to equilibrium (R/S = 1) in a viable population. The number of spawners here is greater than spawners needed to maximize recruits. This approach is used in Puget Sound (Appendix 9.2). This approach can also incorporate covariates (Section 4.3.3 and Appendix 9.5).

4.6.3 Setting diversity goals

- Reduce pHOS (proportion of hatchery-origin salmonids on the spawning grounds) and increase PNI as a means to: 1) increase productivity; 2) allow the species to adapt to local conditions; and, 3) reduce genetic homogeneity associated with domesticated hatchery salmon. This recommendation may not be relevant to conservation hatcheries that are needed to recover ESA-listed species. PNI and pHOS values recommended by CA HSRG (2012) and HSRG (2014) could be used as a starting point.

4.6.4 Setting spatial structure goals

- Increase the number of habitats that support viable populations of natural-origin Chinook salmon and steelhead. Basic actions that support spatial structure goals typically involve removal of fish barriers. However, ensuring and documenting viable natural-origin populations of Chinook salmon and steelhead in existing habitats also supports this goal.

4.6.5 Setting goals related to management actions

- Track recruitment model coefficients associated with management actions based on the covariate spawner-recruit analyses described above (Section 4.3.3 and Appendix 9.5). The goal stems from the hypothesized benefit of each management action. For example, seasonal flow and habitat coefficients are hypothesized to be positive and influential, whereas temperature is typically hypothesized to have a negative coefficient. Lack of significant relationships may suggest the need for adaptive management actions, assuming data are reasonably accurate, contrast is sufficient, and the analysis is not confounded by other factors.

4.6.6 First steps for setting biological goals

Setting biological goals for natural origin salmonids and tracking progress towards those goals requires reasonably accurate data. Consistently and comprehensively estimating the contribution of hatchery-origin salmonids in the catch and spawning grounds is the greatest deterrent to reasonably accurate production estimates of natural-origin salmonids. Progress towards goals cannot be evaluated without accurate estimates of natural-origin salmonids. Some progress on this issue has been made in recent years (e.g., Willmes et al. 2018, Palmer-Zwahlen et al. 2015, 2018). However, natural-origin versus hatchery-origin abundances in the catch and spawning grounds needs to be estimated each year for each watershed and annually reported in documents, such as GrandTab (Azat 2018). Furthermore, increasing the fin-clip rate or tagging rate in hatchery fall-run Chinook salmon to 100% could improve the reliability of these estimates and potentially enhance the ability to measure the response of natural-origin salmon to management actions (Cal-Nev AFS 2009, CA HSRG 2012, Mohr et al. 2017).

Overall monitoring of Chinook salmon in the CV has improved in recent years (Bergman et al. 2012), but key metrics (Section 4.5.1) need to be consistently measured each year in every watershed of interest. Steelhead monitoring has improved in a few watersheds, but data for natural-origin steelhead are mostly inadequate and may not improve given their low abundances. Ultimately, data are needed to create "brood tables" for each population in support of support stock-recruit analyses. When data are deficient, assumptions may be used (see Appendix 9.2). However, these assumptions and lack of data will lead to greater uncertainty in estimates of the status and trends of the populations, and uncertainty in potential benefits of restoration actions.

BOX 4.3: Comment on AFRP Doubling Goals

USFWS (2001) established a goal to double natural production of Chinook salmon and steelhead (and other anadromous species) within 10 years and the goal was set in public law (www.usbr.gov/mp/cvpia/title_34/public_law_complete.html). Nevertheless, the Panel believes this goal to be unrealistic (e.g., 990,000 natural Chinook salmon, including harvested fish). Values in the baseline period likely underestimated hatchery-origin Chinook salmon in total returns, which appear to be based on professional opinion rather than actual data for hatchery-origin fish (see Mills and Fisher 1994). Recent estimates of pHOS confirm that hatchery fish on the spawning grounds are higher than those assumed in the doubling goal analysis (e.g., Willmes et al. 2018, Palmer-Zwahlen et al. 2018; Figure 4.4). The Panel is uncertain whether estimated harvests of natural-origin Chinook salmon in the doubling goal analysis were reasonably accurate, but suspect that they were too high because they probably include some hatchery fish. As described in Section 4.6, positive trends in abundance and productivity metrics may provide the best goals, rather than a goal to double abundance of the natural population.

5 CONCLUSIONS

The charge to the Panel was to develop scientifically defensible methods for formulating quantitative biological goals for narrative objectives in the Bay-Delta Plan. The Panel focused on methods for determining ecological responses to management actions including manipulations of flow and habitat restoration. The principal focus of the analysis was therefore on monitoring and evaluation of responses of salmonids and other native fishes to management actions, and identifying structural components and processes likely underlying those responses. Methods for developing biological goals largely relied upon current monitoring programs and existing data; however, monitoring is inadequate for some regions, species, and processes, so methods that would require additional monitoring and assessment were considered.

The work was divided into three chapters covering ecosystem structure and function, native fishes and fish assemblages, and Chinook salmon and steelhead.

Chapter 2 summarizes key structural and functional components of the aquatic ecosystems of tributary rivers and the estuary, emphasizing components that support native fishes by providing favorable habitat elements such as physical configuration, high water quality, adequate food, and shelter from predators. The chapter distinguishes information focused on river ecosystems from that focused on the estuary, because the estuary is strongly influenced by tidal oscillations, variations in fresh water and salinity, and it has much longer and richer data sets. Several structural ecosystem elements in both the estuary and tributary rivers provide essential underpinning to analyses of fish responses; these include physical (e.g., temperature, flow, turbidity, and conductivity or salinity) and chemical (e.g., nutrients, dissolved oxygen) properties, as well as biotic structural components such as the main primary producers (e.g., algae and plants), the microbial community, and aquatic invertebrates. Functional ecosystem components, such as energy and nutrient flows through food webs, are not consistently monitored in rivers or the estuary, though information on some components, such as primary and secondary production and ecosystem respiration, would be helpful for understanding responses of fish to management actions. Measurements in the rivers should include bioassessment tools such as those using attached (benthic) algae and aquatic invertebrates. In the estuary, phytoplankton primary production can be estimated from existing monitoring programs, and secondary production of zooplankton can be roughly estimated. However, very little monitoring and assessment of structural or functional components occurs in tidal wetlands and other peripheral ecosystems, limiting our ability to assess the responses of these systems to management actions intended to support native fishes.

Chapter 3 proposes a general approach for evaluating management actions on the abundance of fishes in the rivers and upper estuary (other than Chinook salmon and steelhead) using data collected by diverse agency surveys throughout the region. The list of native fish species provided by the charge to the Panel included several whose abundance is so low that no amount of monitoring will reveal their responses to management actions. Therefore, the Panel strongly recommends expanding this list to include other, unlisted native and some non-native fish species to evaluate more comprehensive and integrative responses to management actions. Expanding the array of fish species will enhance the ability to detect similarities and contrasts in responses of fish. Six possible methods for fish assessment that provide alternative views of the fishes' responses are discussed. A selected group of fish species should be designated as indicator species, either individually or as assemblages. Monitoring of environmental and other non-fish metrics, as noted in Chapter 2, will be necessary to help evaluate causes of short- and long-term changes in fish populations. The best approach to monitoring river and estuarine fishes is to use multiple approaches and a suite of metrics with measurements sustained over decades. Population models are available or could be developed for three species (striped bass, delta smelt, Sacramento splittail) to help managers predict effects of change; similar models should be developed for other selected species as well.

Chapter 4 provides recommendations for setting biological goals for natural-origin Chinook salmon and steelhead in the rivers and estuary while also providing a framework for quantifying responses to management actions. VSP criteria (abundance, productivity, diversity, and spatial structure) are well established and can be used to establish metrics, especially for productivity and abundance, to track progress towards biological goals. The Panel recommends a framework for evaluating salmonid abundance and productivity that incorporates density dependence in a stock-recruit relationship. However, this approach requires consistent and reasonably accurate abundance estimates of juvenile salmonids and natural-origin adult salmonids in harvest and spawning escapement. Abundance estimates of natural-origin salmonids are typically confounded by large numbers of hatchery-origin salmonids, whose abundance must be better estimated if goals for natural-origin fish are to be set and evaluated. Approaches for establishing goals for productivity and abundance are described, but the Panel also recommends tracking trends in productivity and abundance in response to management actions. In order to demonstrate the framework, an example of how the stock-recruit approach can be applied to existing juvenile and adult spawner abundance data for fall-run Chinook salmon in the Stanislaus River is provided. This example describes how empirical data can be used to set biological goals and how progress can be evaluated. Population-level measurement of responses to management actions may take 20-30 years (4-6 generations) owing to natural variation in survival rates, imprecision in monitoring, and generation time, although assessment across the affected part of the life cycle can serve as an early indicator of responses. The Panel emphasizes, however, that population viability depends on survival across the entire life history; for example, increased survival in fresh water may not readily translate to an increase in returning adults if bottlenecks remain in the estuary or in the ocean.

Ultimately, goals for populations of Chinook salmon, steelhead, other fishes, and for the ecosystem supporting them should be integrated in order to quantitatively evaluate current conditions, future trends, and the effectiveness of various management actions. Such evaluations will be most successful if the actions are implemented and managed as experiments, with clear identification of the key unknowns and a design to reduce uncertainty. There must also be a responsive feedback process for changing or refining metrics used to track goals. Success will depend on:

- Well-designed and realistic quantitative goals and objectives.
- Well-designed quantitative metrics that track progress towards goals.
- Monitoring programs that provide accurate estimates of metrics.
- Support for data management and analysis that readily adapts to new information.
- Sustained funding for long-term assessment that can distinguish environmental variability from responses to management actions.

The Panel recognizes that implementing an effective monitoring program to rigorously assess management actions is extremely difficult in this era of rapid change with increased variability in freshwater flows, rising sea level, new invaders and contaminants, and increasing water demand for human use. Nevertheless, such a program is possible, given the amount of data already available and the many people, both in agencies and outside them, who comprehend the system's complexities and are dedicated to improving the quality of California's aquatic environments.

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7 GLOSSARY

7.1 Acronyms and Abbreviations

Board, SWRCB	California State Water Resources Control Board
CDFW	California Department of Fish and Wildlife
CV	California's Central Valley
DPS	Distinct Population Segment
DWR	California Department of Water Resources
EHMP	Ecosystem Health and Monitoring Program (Moreton Bay, Australia)
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
FERC	Federal Energy Regulatory Commission
HSRG	Hatchery Scientific Reform Group (established by Congress)
IBI	Index of Biological Integrity
Panel	Authors of this report
PET	Taxa belonging to Plecoptera, Ephemeroptera and Trichoptera
Plan	Bay-Delta Water Quality Control Plan (Bay-Delta Plan)
PM	Performance Measure
pHOS	Proportion of Hatchery-Origin Salmon spawning in the wild
PNI	Proportionate Natural Influence
pNOB	Proportion of Natural-Origin salmon in the hatchery Broodstock
POD	Pelagic Organism Decline
SDM	Structured Decision-Making
SEP	Stanislaus Scientific Evaluation Process
SFE	San Francisco Estuary
TRT	Technical Review Team
UCD	University of California, Davis
USFWS	United States Fish and Wildlife Service
VSP	Viable Salmon Population
X2	Distance up estuary to salinity=2; an index of the physical response of the estuary to changes in freshwater flow (Jassby et al. 1995)

7.2 Terms

Term	Definition
Abundance (salmon)	Number of salmon at a specific life stage
Adaptive management	Experimental management, either active or passive (Walters and Holling 1997)
Assemblages	Groups of species that may occur together as based on habitat or sampling
Brood year (salmon)	The year in which parents spawn
Brood table	A table showing the number of progeny (typically adults) by age class that are produced by the parent spawning population in a brood year
Compensatory density dependence	Occurs when a populations growth rate is highest at low density and decreases as density increases
Delta	The legal Delta as defined by the California Water Code
Diversity (salmon)	Degree of genetic and phenotypic (e.g., size, time and age at juvenile migration and adult return) variation in a salmon population
Escapement (salmon)	Maturing salmon that escape fisheries and spawn in streams
Exceedance probability	Likelihood that a given level of fish abundance or recruitment will re-occur or be exceeded in a given position
Exploitation rate (U)	The proportion of salmon returning from a brood year that are harvested
Focal species	A focal species is any species chosen for special attention in a multi-species planning effort for managing important habitats (e.g. the Delta). For each effort, multiple focal species are chosen that represent an array of important habitat elements or ecosystem attributes so that measurements of their collective abundance (and/or other metrics) are also a measure of the status of the habitat or ecosystem. Developed for birds (e.g., birds of CV riparian forests) but applicable to fishes and other groups
Functional flow	Ecosystem water budgets supporting crucial processes upon which native aquatic species depend
Fundamental objective	Objective closely tied to goal
Biological goal	Defined by the Board as a quantitative benchmark for assessing progress toward narrative objectives
Hatchery-origin salmon	Salmonid progeny that were produced in hatcheries and released into the wild
Indicator species	Species that can be used to estimate responses of other species such as uncommon native species
Intrinsic productivity (salmon)	The predicted ratio of natural-origin offspring to parent spawners in the absence of density dependence
Invasive species	Introduced (non-native) species that have adverse effects on extant species, habitats, or ecosystems
Management actions	Alteration or variation in flow regimes, habitat restoration, and other non-flow changes
Means objective	Approach to achieve a fundamental objective
Meta-population (salmon)	A group of populations separated by space that have some levels of interbreeding
Proportionate Natural Influence (PNI)	PNI = pNOB/(pNOB + pPHOS), where pNOB = proportion of broodstock that is natural origin and pHOS is proportion of spawners in the wild that are hatchery origin

Natural-origin salmonids (NOR)	Salmonid progeny produced by salmon and steelhead that spawned naturally in streams, including progeny produced by hatchery fish spawning naturally
Novel ecosystem	An ecosystem greatly altered from its past state by changes in, e.g., species distributions, land uses, and geomorphology (Hobbs et al. 2013).
Pre-fishery recruits (salmon)	Numbers of returning salmon prior to harvest in fisheries, typically estimated as the sum of catch plus spawning escapement
Productivity or production (ecosystem)	(Primary) the rate at which organic carbon is formed (Secondary) the rate at which food is converted to animal mass
Productivity (salmon)	The ratio of natural-origin offspring to parent spawners
Recruits (salmon)	Number of returning adult progeny produced from a parent spawning year (brood year), typically measured as pre-fishery recruits (catch plus escapement) or at the spawning grounds
Replacement line (salmon)	In a spawner-recruit relationship, where the number of adult progeny equals the number of parent spawners such that returns per spawner (R/S) = 1
Smolt (salmon)	Juvenile salmonids that physiologically, morphologically, hormonally and actively migrating from fresh water to the sea. In this report, for simplicity, we often use "smolt" to include fry, parr, and yearling migrants.
Spatial structure (salmon)	The number and size of viable populations across the landscape
Stock-recruitment (salmon)	The relationships between numbers of spawners (stock) and the number of progeny (recruits)
Tailwater	A reach of river below a dam that regulates its flows
Unimpaired flow	Unimpaired flow is flow in rivers and streams that would have occurred in the absence of water storage and diversion projects. The unimpaired flow estimates provide a measure of total water supply available for all uses after removing the impacts of most upstream alterations. Channel improvements, levees, and flood bypasses are assumed to exist (SWRCB 2000).
Viable Salmon Population (VSP) criteria	Abundance, productivity, diversity, and spatial structure of the population or ESU, which is the meta-population

8 REFERENCES

Abal, E.G., S.E. Bunn, and W.C. Dennison (Eds.). 2005. Healthy Waterways Healthy Catchments: Making the connection in South East Queensland, Australia. Moreton Bay Waterways and Catchment Partnership, Brisbane. 240p.

Aceituno, M.E. 1993. The relationship between instream flow and physical habitat availability for Chinook salmon in the Stanislaus River, California. Report prepared by US Fish and Wildlife Service, Sacramento CA. 92p.

Alpine, A.E., and J.E. Cloern. 1992. Trophic interactions and direct physical effects control phytoplankton biomass and production in an estuary. Limnology and Oceanography 37:946-955.

Araki, H., B.A. Berejikian, M.J. Ford, and M.S. Blouin. 2008. Fitness of hatchery-reared salmonids in the wild. Evolutionary Applications 1:342-355.

Araki H., B. Cooper, M.S. Blouin. 2009. Carry-over effect of captive breeding reduces reproductive fitness of wild-born descendants in the wild. Biology Letters 5:621-624.

Azat, J. 2018. GrandTab 2018.04.09. California Central Valley Chinook population database report. California Department of Fish and Wildlife. https://www.calfish.org/ProgramsData/Species/CDFWAnadromousResourceAssessment.aspx

Baerwald M.R., B.M. Schreier, G. Schumer, and B. May. 2012. Detection of threatened delta smelt in the gut contents of the invasive Mississippi Silverside in the San Francisco Estuary using TaqMan assays. Transactions of the American Fisheries Society 141(6):1600-1607.

Barnett-Johnson, R., C.B. Grimes, C.F. Roye, and C.J. Donohoe. 2007. Identifying the contribution of wild and hatchery Chinook salmon (*Oncorhynchus tshawytscha*) to the ocean fishery using otolith microstructure as natural tags. Canadian Journal of Fisheries and Aquatic Sciences 64(12):1683–1692.

Baumsteiger, J. and P.B. Moyle 2017. Assessing extinction. Bioscience 67:357-366. doi:10.1093/biosci/bix001

Beacham, T.D., C. Wallace, K. Jonsen et al. 2019. Comparison of coded-wire tagging with parentage-based tagging and genetic stock identification in a large-scale coho salmon fisheries application in British Columbia, Canada. Evolutionary Applications 12:230–254.

Beller, E.E., E.N. Spotswood, A.H. Robinson, M.G. Anderson, E.S. Higgs, R.J. Hobbs, K.N. Suding, E.S. Zavaleta, J.L. Grenier, and R.M. Grossinger. 2018. Building ecological resilience in highly modified landscapes. BioScience 69:80-92.

Berejikian, B.A., and D.M. Van Doornik. 2018. Increased natural reproduction and genetic diversity one generation after cessation of a steelhead trout (*Oncorhynchus mykiss*) conservation hatchery program. PLoS ONE 13(1):e0190799.

Berg, N., and A. Hall. 2017. Anthropogenic warming impacts on California snowpack during drought. Geophysical Research Letters 44(5):2511-2518.

Bergman, J., R. Nielson, and A. Low. 2012. Central Valley Chinook salmon in-river escapement monitoring plan. California Department of Fish and Game. Fisheries Branch Administrative Report Number: 2012-1.

Bernhardt, E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G.M. Kondolf, P.S. Lake, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Synthesizing U.S. river restoration efforts. Science 308:636-637.

Bingham, D.M., B.M. Kennedy, K.C. Hanson, and C.T. Smith. 2014. Loss of genetic integrity in hatchery steelhead produced by juvenile-based broodstock and wild integration: conflicts in production and conservation goals. North American Journal of Fisheries Management 34(3):609-620.

Blair, G.R., L.C. Lestelle, and L.E. Mobrand. 2009. The ecosystem diagnosis and treatment model: a tool for assessing salmonid performance potential based on habitat conditions. American Fisheries Society Symposium 71:289-309.

Bonada, N., N. Prat, V.H. Resh, and B. Statzner. 2006. Developments in aquatic insect biomonitoring: A comparative analysis of recent approaches. Annual Review of Entomology 51:495-523.

Bradford, M.J., J. Korman, and P.S. Higgins. 2005. Using confidence intervals to estimate the response of salmon populations (*Oncorhynchus* spp.) to experimental habitat alterations. Canadian Journal of Fisheries and Aquatic Sciences 62: 2716-2726.

Bradford, M.J., P.S. Higgins, J. Korman, and J. Sneep. 2011. Test of an environmental flow release in a British Columbia river: does more water mean more fish? Freshwater Biology 56:2119–2134.

Brown, L.R. 2000. Fish communities and their associations with environmental variables, lower San Joaquin River drainage, California. Environmental Biology of Fishes 57:251-269.

Brown, L.R. and T. Ford. 2002. Effects of flow on the fish communities of a regulated California river: implications for managing native fishes. River Research and Applications 18:331-342.

Brown, L.R. and P.B. Moyle. 2005. Native fishes of the Sacramento-San Joaquin drainage, California: A history of decline. American Fisheries Society Symposium 45:75-98.

Brown, L.R., W. Kimmerer, J.L. Conrad, S. Lesmeister, and A. Mueller-Solger. 2016. Food webs of the Delta, Suisun Bay, and Suisun Marsh: an update on current understanding and possibilities for management. San Francisco Estuary and Watershed Science 14(3). https://doi.org/10.15447/sfews.2016v14iss3art4

Buchanan, R. 2013. OCAP 2011 Steelhead tagging study: statistical methods and results. Report to J. Israel, U.S. Bureau of Reclamation, Bay Delta Office, Sacramento, California 95814.

Buchanan, R.A., P.L. Brandes, and J.R. Skalski. 2018. Survival of juvenile fall-run Chinook salmon through the San Joaquin River Delta, California, 2010-2015. North American Journal of Fisheries Management 38(3):663-679.

Buhle, E.R., K.K. Holsman, M.D. Scheuerell, and A. Albaugh. 2009. Using an unplanned experiment to evaluate the effects of hatcheries and environmental variation on threatened populations of wild salmon. Biological Conservation 142:2449-2455.

Bunn, S.E., and A.H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30:492-507.

CA HSRG (California Hatchery Scientific Review Group). 2012. California Hatchery Review Report. Prepared for the US Fish and Wildlife Service and Pacific States Marine Fisheries Commission. June 2012. 100 pgs.

Calbet, A., and M.R. Landry. 2004. Phytoplankton growth, microzooplankton grazing, and carbon cycling in marine systems. Limnology and Oceanography 49(1):51-57.

Calbet, A., and E. Saiz. 2005. The ciliate-copepod link in marine ecosystems. Aquatic Microbial Ecology 38(2):157-167.

Callaway, J.C., V.T. Parker, M.C. Vasey, and L.M. Schile. 2007. Emerging issues for the restoration of tidal marsh ecosystems in the context of predicted climate change. Madrono 54(3):234-248.

Cal-Nev AFS (California-Nevada Chapter of the American Fisheries Society, Executive Committee). 2009. Policy statement supporting total (100%) marking of all California hatchery produced fall run Chinook salmon. January 13, 2009. 6 p.

Castleberry, D.T., J.J. Cech, D.C. Erman, D. Hankin, M. Healey, G.M. Kondolf, M. Mangel, M. Mohr, P.B. Moyle, J. Nielsen, T.P. Speed, and J.G. Williams. 1996. Uncertainty and instream flow standards. Fisheries 20 (8): 20-21.

Cavallo, B., P. Gaskill, J. Melgo, and S.C. Zeug. 2015. Predicting juvenile Chinook salmon routing in riverine and tidal channels of a freshwater estuary. Environmental Biology of Fishes 98(6):1571-1582.

Chadwick, H.K., D.E. Stevens, L.W. Miller, and W. Van Winkle. 1977. Some factors regulating the striped bass population in the Sacramento-San Joaquin Estuary, California. Pages 18-35 *in* Proceedings of the Conference on Assessing the Effects of Power-Plant-Induced Mortality on Fish Populations, Gatlinburg, Tennessee, May 3-6, 1977. Pergamon, New York.

Chilcote, M.W., K.W. Goodson, and M.R. Falcy. 2011. Reduced recruitment performance in natural populations of anadromous salmonids associated with hatchery-reared fish. Canadian Journal of Fisheries and Aquatic Sciences 68:511-522.

Chilcote, M.W., K.W. Goodson, and M.R. Falcy. 2013. Corrigendum: Reduced recruitment performance in natural populations of anadromous salmonids associated with hatchery-reared fish. Canadian Journal of Fisheries and Aquatic Sciences 70:513-515.

Chinook Technical Committee. 1999. Maximum sustained yield or biologically based escapement goals for selected Chinook salmon stocks used by the Pacific Salmon Commission's Chinook Technical Committee for escapement assessment. Pacific Salmon Commission, Report TCCHINOOK (99)-3, Vancouver, BC.

Christie, M.R., M.J. Ford, and M.S. Blouin. 2014. On the reproductive success of early-generation hatchery fish in the wild. Evolutionary Applications 7:883-896.

Christie, M.R., M.L. Marine, S.E. Fox, R.A. French, and M.S. Blouin. 2016. A single generation of domestication heritably alters the expression of hundreds of genes. Nature Communications 7:10676 (DOI: 10.1038/ncomms10676).

Cloern, J.E. 1991. Tidal stirring and phytoplankton bloom dynamics in an estuary. Journal of Marine Research 49(1):203-221.

Cloern, J.E., N. Knowles, L.R. Brown, D. Cayan, M.D. Dettinger, T.L. Morgan, D.H. Schoellhamer, M.T. Stacey, M. van der Wegen, R.W. Wagner, and A.D. Jassby. 2011. Projected evolution of California's San Francisco Bay-Delta-River system in a century of climate change. PLoS ONE 6:e24465. https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0024465

Cloern, J.E., P.L. Barnard, E. Beller, J.C. Callaway, J.L. Grenier, E.D. Grosholz, R. Grossinger, K. Hieb, J.T. Hollibaugh, N. Knowles, M. Sutula, S. Veloz, K. Wasson, and A. Whipple. 2016. Estuaries Life on the Edge. Pages 359-387 *in* H. Mooney, and E. Zavaleta (Eds.). Ecosystems of California. University of California Press, Berkeley.

Cole, B.E., and J.E. Cloern. 1987. An empirical model for estimating phytoplankton productivity in estuaries. Marine Ecology Progress Series 36:299-305.

Conrad, J.L., A.J. Bibian, K.L. Weinersmith, D. De Carion, M.J. Young, P. Crain, E.L. Hestir, M.J. Santos, and A. Sih. 2016. Novel species interactions in a highly modified estuary: association of largemouth bass with Brazilian waterweed *Egeria densa.* Transactions of the American Fisheries Society 145(2):249-263.

Costanza, R., R. D'arge, R. De Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. Van Den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253-260.

Courter, I.I., D.B. Child, J.A. Hobbs, T.M. Garrison, J. J.G. Glessner, and S. Duery. 2013. Resident rainbow trout produce anadromous offspring in a large interior watershed. Canadian Journal of Fisheries and Aquatic Sciences 70:701–710.

Crain, P.K., K. Whitener, P.B. Moyle. 2004. Use of a restored central California floodplain by larvae of native and alien fishes. American Fisheries Society Symposium 39:125-140.

Crawford, B., and S.M. Rumsey. 2011. Guidance for monitoring recovery of Pacific Northwest salmon and steelhead listed under the federal endangered species act. National Marine Fisheries Service, NW Region.

Cummins, K.W. 1974. Structure and function of stream ecosystems. BioScience 24:631-641.

Dahm, C.N., K.W. Cummins, H.M. Valett, and R.L. Coleman. 1995. An ecosystem view of the restoration of the Kissimmee River. Restoration Ecology 3:225-238.

Dahm, C.N., A.E. Parker, A.E. Adelson, M.A. Christman, and B.A. Bergamaschi. 2016. Nutrient dynamics of the Delta: effects on primary producers. San Francisco Estuary and Watershed Science 14(4). https://doi.org/10.15447/sfews.2016v14iss4art4

Deegan, L.A., J.T. Finn, S.G. Ayvazian, C.A. Ryder-Kieffer, and J. Buonaccorsi. 1997. Development and validation of an estuarine biotic integrity index. Estuaries 20:601-617.

Dettinger, M., J. Anderson, M. Anderson, L. R. Brown, D. Cayan, and E. Maurer. 2016. Climate change and the Delta. San Francisco Estuary and Watershed Science 14(3). http://dx.doi.org/10.15447/sfews.2016v14iss2art5

Domagalski, J. 2001. Mercury and methylmercury in water and sediment of the Sacramento River Basin, California. Applied Geochemistry 16(15):1677-1691.

Dorner, B., Catalano, M.J., and R.M. Peterman. 2018. Spatial and temporal patterns of covariation in productivity of Chinook salmon populations of the northeastern Pacific Ocean. Canadian Journal of Fisheries and Aquatic Sciences 75:1082-1095.

Drinkwater, K.F., and K.T. Frank. 1994. Effects of river regulation and diversion on marine fish and invertebrates. Aquatic Conservation: Marine and Freshwater Ecosystems 4:135-151.

Dugdale, R.C., F.P. Wilkerson, V.E. Hogue, and A. Marchi. 2007. The role of ammonium and nitrate in spring bloom development in San Francisco Bay. Estuarine Coastal and Shelf Science 73:17-29.

Dugdale, R.C., F. Wilkerson, A.E. Parker, A. Marchi, and K. Taberski. 2012. River flow and ammonium discharge determine spring phytoplankton blooms in an urbanized estuary. Estuarine Coastal and Shelf Science 115:187-199.

Durand, J., W. Fleenor, R. McElreath, M. Santos, and P. Moyle. 2016. Physical controls on the distribution of the submersed aquatic weed *Egeria densa* in the Sacramento-San Joaquin Delta and implications for habitat. San Francisco Estuary and Watershed Science 14(1). https://doi.org/10.15447/sfews.2016v14iss1art4

Dybala, K.E., N. Clipperton, T. Gardali, G.H. Golet, R. Kelsey, S. Lorenzato, R. Melcer Jr, N. E. Seavy, J. G. Silveira, and G. S. Yarris. 2017. Population and habitat objectives for avian conservation in California's Central Valley riparian ecosystems. San Francisco Estuary and Watershed Science 15(1). https://doi.org/10.15447/sfews.2017v15iss1art5

English, K.K., R.E. Bailey, and D. Robichaud. 2007. Assessment of Chinook salmon returns to the Fraser River watershed using run reconstruction techniques, 1982-2004. Canadian Science Advisory Secretariat Research Document 2007/020. 76 pp. http://www.dfo-mpo.gc.ca/csas-sccs/publications/resdocs-docrech/2007/2007_020-eng.htm

Escobar-Arias, M.I., and G.B. Pasternack. 2010. A hydrogeomorphic dynamics approach to assess in-stream ecological functionality using the function flows model, part 1 – model characteristics. River Research and Applications 26:1103-1128.

Feyrer, F., M.L. Nobriga, and T.R. Sommer. 2007. Multi-decadal trends for three declining fish species: habitat patterns and mechanisms in the San Francisco Estuary, California, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 64:723-734.

Feyrer, F., K. Newman, M. Nobriga, and T. Sommer. 2011. Modeling the effects of future outflow on the abiotic habitat of an imperiled estuarine fish. Estuaries and Coasts 34:120-128.

Fleischman, S.J., M.J. Catalano, R.A. Clark, and D.R. Bernard. 2013. An age-structured state-space stock-recruit model for Pacific salmon (*Oncorhynchus* spp.). Canadian Journal of Fisheries and Aquatic Science 70:401-414.

Ford, M.J. (ed.). 2011. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. Draft U.S. Department of Commerce, NOAA Technical Memorandum NOAA-NWFSC-113. 281p.

Ford, M.J., A.R. Murdoch, M.S. Hughes, T.R. Seamons, and E.S. LaHood. 2016. Broodstock history strongly influences natural spawning success in hatchery steelhead (*Oncorhynchus mykiss*). PLoS ONE 11(10): e0164801. https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0164801

Gore, J.A., J.B. Layzer, and J. Mead. 2001. Macroinvertebrate instream flow studies after 20 years: a role in stream management and restoration. Regulated Rivers: Research and Management 17:527–542.

Grimaldo, L.F., T. Sommer, N.V. Ark, G. Jones, E. Holland, P.B. Moyle, B. Herbold, and P. Smith. 2009a. Factors affecting fish entrainment into massive water diversions in a tidal freshwater estuary: can fish losses be managed? North American Journal of Fisheries Management 29(5):1253-1270.

Grimaldo, L. F., A. R. Stewart, and W. Kimmerer. 2009b. Dietary segregation of pelagic and littoral fish assemblages in a highly modified tidal freshwater estuary. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 1:200-217.

Grimaldo, L., F. Feyrer, J. Burns, and D. Maniscalco. 2017. Sampling uncharted waters: examining rearing habitat of larval longfin smelt (*Spirinchus thaleichthys*) in the upper San Francisco Estuary. Estuaries and Coasts 40(6):1771-1784.

Hammock, B.G., J.A. Hobbs, S.B. Slater, S. Acuna, and S.J. Teh. 2015. Contaminant and food limitation stress in an endangered estuarine fish. Science of the Total Environment 532:316-326.

Hammock, B.G., R. Hartman, S.B. Slater, A. Hennessy, and S.J. Teh. 2019. Tidal wetlands associated with foraging success of delta smelt. Estuaries and Coasts. https://doi.org/10.1007/s12237-019-00521-5.

Harrell, W.C. and T.R. Sommer. 2003. Patterns of adult fish use on California's Yolo Bypass floodplain. Pages 88-93 *in* P.M Faber, editor. California riparian systems: processes and floodplain management, ecology, and restoration. Sacramento, California: Riparian Habitat Joint Venture.

Harvey, B.N., Jacobson, D.P., and M. Banks. 2014. Quantifying the uncertainty of a juvenile Chinook salmon race identification method for a mixed-race stock. North American Journal of Fisheries Management 34:1177–1186.

Hawkins, C.P., R.H. Norris, J.N. Hogue, and J.W. Feminella. 2000. Development and evaluation of predictive models for measuring the biological integrity of streams. Ecological Applications 10:1456-1477.

Healy, M.C. 1991. Life history of Chinook Salmon (*Oncorhynchus tshawytscha*). In Pacific Salmon Life Histories, C. Groot and L. Margolis [eds.]. UBC Press.

Hendrix, N. E. Jennings, A. Criss, E. Danner, V. Sridharan, C.M. Greene, H. Imaki, and S.T. Lindley. 2017. Model description for the Sacramento River winter-run Chinook salmon life cycle model. May 9, 2017. QEDA Consulting. Seattle, WA.

Herbold, B.H. 2016. Dutch Slough adaptive management and monitoring plan. Unpublished report accessed 3 April 2019 at https://water.ca.gov/-/media/DWR-Website/Web-Pages/Programs/Integrated-Regional-Water-Management/Delta-Ecosystem-Enhancement-Program/Dutch-Slough-Tidal-Restoration-Project/Files/Dutch-Slough-Adaptive-Management-and-Monitoring-Plan-2016.pdf?la=en&hash=62D384EA1C90981E94032AD675D7590CC34C16A5

Hilborn, R., and C.J. Walters. 1992. Quantitative fisheries stock assessment. Chapman and Hall, New York. 570 pp.

Hill, M.F., L.W. Botsford, and A. Hastings. 2003. The effects of spawning age distribution on salmon persistence in fluctuating environments. Journal of Animal Ecology 72:736–744.

Hobbs, J.A., W.A. Bennett, J.E. Burton, and B. Baskerville-Bridges. 2007. Modification of the biological intercept model to account for ontogenetic effects in laboratory-reared delta smelt (*Hypomesus transpacificus*). Fishery Bulletin 105:30-38.

Honey, K., R. Baxter, Z. Hymanson, T. Sommer, M. Gingras, and C.P. Cadrett. 2004. IEP Long-term Fish Monitoring Program Element Review. Interagency Ecological Program, Department of Water Resources, Sacramento Ca. 302 p.

Howard, J.K., K.A. Fesenmeyer, T.E. Grantham, J.H. Viers, P.R. Ode, P.B. Moyle, S.J. Kupferburg, J.L. Furnish, A. Rehn, J. Slusark, R.D. Mazor, N.R. Santos, R.A. Peek, and A.N. Wright. 2018. A freshwater conservation blueprint for California: Prioritizing watersheds for freshwater biodiversity. Freshwater Science 37:417-431.

Howe, E.R., and C.A. Simenstad. 2011. Isotopic determination of food web origins in restoring and ancient estuarine wetlands of the San Francisco Bay and Delta. Estuaries and Coasts 34:597-617.

HSRG (Hatchery Scientific Review Group). 2014. On the Science of Hatcheries: An updated perspective on the role of hatcheries in salmon and steelhead management in the Pacific Northwest. June 2014; revised October 2014. http://hatcheryreform.us/

Hynes, H.B.N. 1975. The stream and its valley. Verhandlungen Internationale Vereinigung für Limnologie 19:1-15.

ICF. 2016. Battle Creek winter-run Chinook salmon reintroduction plan. (ICF 00148.15.) Sacramento, CA. Prepared for California Department of Fish and Wildlife. Sacramento, CA. https://www.wildlife.ca.gov/Drought/Projects/Battle-Creek

IEP-MAST (Interagency Ecological Program, Management, Analysis, and Synthesis Team). 2015. An updated conceptual model of Delta Smelt biology: our evolving understanding of an estuarine fish. Interagency Ecological Program. Technical Report 90.

ISAB (Independent Scientific Advisory Board). 2001. Model synthesis report: an analysis of decision support tools in Columbia River Basin salmon management. ISAB 2001-1, March 2, 2001. https://www.nwcouncil.org/sites/default/files/isab2001_1.pdf

ISAB. 2015. Density dependence and its implications for fish management and restoration programs in the Columbia River Basin. ISAB 2015-1, February 25, 2015. https://www.nwcouncil.org/sites/default/files/isab2015-1.pdf

Jassby, A.D., W.J. Kimmerer, S.G. Monismith, C. Armor, J.E. Cloern, T.M. Powell, J.R. Schubel, and T.J. Vendlinski. 1995. Isohaline position as a habitat indicator for estuarine populations. Ecological Applications 5(1):272-289.

Jassby, A.D., J.E. Cloern, and B.E. Cole. 2002. Annual primary production: patterns and mechanisms of change in a nutrient-rich tidal estuary. Limnology and Oceanography 47(3):698-712.

Jassby, A. D. 2005. Phytoplankton regulation in a eutrophic tidal river (San Joaquin River, California). San Francisco Estuary and Watershed Science 3(1). https://doi.org/10.15447/sfews.2005v3iss1art5

Jassby, A., and E.E. Van Nieuwenhuyse. 2005. Low dissolved oxygen in an estuarine channel (San Joaquin River, California): mechanisms and models based on long-term time series. San Francisco Estuary and Watershed Science 3(2). https://doi.org/10.15447/sfews.2005v3iss2art2

Johnson, R.C., and S.T. Lindley. 2016. Chapter 5: Central Valley Recovery Domain. In, Viability assessment of Pacific salmon and steelhead under the Endangered Species Act: southwest. NOAA-TM-NMFS-SWFSC-564.

Johnson, R.C., S. Windell, P.L. Brandes, J.L. Conrad, J. Ferguson, P.A.L. Goertler, B.N. Harvey, J. Heublein, J.A. Israel, D.W. Kratville, J.E. Kirsch, R.W. Perry, J. Pisciotto, W.R. Poytress, K. Reece, and B.G. Swart. 2017. Science advancements key to increasing management value of life stage monitoring networks for endangered Sacramento River winter-run Chinook Salmon in California. San Francisco Estuary and Watershed Science15(3). https://escholarship.org/uc/item/6751j957

Jones, K.K., T.J. Cornwell, D.L. Bottom, S. Stein, and K.J. Anlauf-Dunn. 2018. Population viability improves following termination of coho salmon hatchery releases. North American Journal of Fisheries Management 38:39–55.

Karr, J.R. and Dudley, D.R., 1981. Ecological perspective on water quality goals. Environmental Management *5*(1):55-68.

Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale*.* Illinois Natural History Survey, Champaign, Special Publication 5.

Katz, J., C. Jeffres, L. Conrad, T. Sommer, N. Corline, J. Martinez, S. Brumbaugh, L. Takata, N. Ikemiyagi, J. Kiernan, and P. Moyle. 2013. The experimental agricultural floodplain habitat investigation at Knaggs Ranch on Yolo Bypass, 2012-2013. https://watershed.ucdavis.edu/files/biblio/Knaggs%202013%20final%20BOR%20report_0.pdf

Kendall, N.W., J.R. McMillan, M.R. Sloat, T.W. Buehrens, T.P. Quinn, G.R. Pess, K.V. Kuzishchin, M.M. McClure, and R.W. Zabel. 2015. Anadromy and residency in steelhead and rainbow trout (*Oncorhynchus mykiss*): a review of the processes and patterns. Canadian Journal of Fisheries and Aquatic Sciences 72:319-342.

Kiernan, J.D., P.B. Moyle, and P.K. Crain. 2012. Restoring native fish assemblages to a regulated California stream using the natural flow regime concept. Ecological Applications. 22:1472-1482.

Kimmerer, W.J. 2002. Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages? Marine Ecology Progress Series 243:39-55.

Kimmerer, W.J. 2006. Response of anchovies dampens effects of the invasive bivalve *Corbula amurensis* on the San Francisco Estuary foodweb. Marine Ecology Progress Series 324:207-218.

Kimmerer, W.J. 2011. Modeling delta smelt losses at the south Delta export facilities. San Francisco Estuary and Watershed Science 9(1). https://doi.org/10.15447/sfews.2011v9iss1art3

Kimmerer, W.J. 2008. Losses of Sacramento River Chinook salmon and delta smelt to entrainment in water diversions in the Sacramento-San Joaquin Delta. San Francisco Estuary and Watershed Science 6(2). https://doi.org/10.15447/sfews.2008v6iss2art2

Kimmerer, W.J., J.H. Cowan, Jr, L.W. Miller, and K.A. Rose. 2000. Analysis of an estuarine striped bass (*Morone saxatilis*) population: influence of density-dependent mortality between metamorphosis and recruitment. Canadian Journal of Fisheries and Aquatic Sciences *57*:478-486.

Kimmerer, W.J., J.H. Cowan, Jr, L.W. Miller, and K.A. Rose. 2001. Analysis of an estuarine striped bass population: effects of environmental conditions during early life. Estuaries 24:557-575.

Kimmerer, W.J., E.S. Gross, and M. MacWilliams. 2009. Is the response of estuarine nekton to freshwater flow in the San Francisco Estuary explained by variation in habitat volume? Estuaries and Coasts 32(2):375-389.

Kimmerer, W.J., A.E. Parker, U. Lidström, and E.J. Carpenter. 2012. Short-term and interannual variability in primary production in the low-salinity zone of the San Francisco Estuary. Estuaries and Coasts 35(4):913-929.

Kimmerer, W.J., M.L. MacWilliams, and E.S. Gross. 2013. Variation of fish habitat and extent of the low-salinity zone with freshwater flow in the San Francisco Estuary. San Francisco Estuary and Watershed Science 11(4). https://doi.org/10.15447/sfews.2013v11iss4art1

Kimmerer, W J., E.S. Gross, and M.L. MacWilliams. 2014. Tidal migration and retention of estuarine zooplankton investigated using a particle-tracking model. Limnology and Oceanography 59(3):901-906.

Kimmerer, W.J., and J.K. Thompson. 2014. Phytoplankton growth balanced by clam and zooplankton grazing and net transport into the low-salinity zone of the San Francisco Estuary. Estuaries and Coasts 37:1202-1218.

Kimmerer, W.J., T.R. Ignoffo, K.R. Kayfetz, and A.M. Slaughter. 2017. Effects of freshwater flow and phytoplankton biomass on growth, reproduction, and spatial subsidies of the estuarine copepod *Pseudodiaptomus forbesi*. Hydrobiologia 807(1):113-130.

Kimmerer, W.J., and K.A. Rose. 2018. Individual-based modeling of delta smelt population dynamics in the upper San Francisco Estuary III. Effects of entrainment mortality and changes in prey. Transactions of the American Fisheries Society 147: 223–243.

Kimmerer, W.J., T.R. Ignoffo, B. Bemowski, J. Moderan, A. Holmes, and B. Bergamaschi. 2018. Zooplankton dynamics in the Cache Slough complex of the upper San Francisco Estuary. San Francisco Estuary & Watershed Science 16(3). https://doi.org/10.15447/sfews.2018v16iss3art4

Kimmerer, W.J., E.S. Gross, A.M. Slaughter, and J.D. Durand. 2019. Spatial subsidies and local mortality of an estuarine copepod revealed using a box model. Estuaries and Coasts 42(1):218-236.

Kissimmee River Restoration Studies Executive Summary. 2006. Defining Success: Expectations for restoration of the Kissimmee River. South Florida Water Management District, West Palm Beach, Florida, 21p.

Kjelson, M., and P. Brandes. 1989. The use of smolt survival estimates to quantify the effects of habitat changes on salmonid stocks in the Sacramento- San Joaquin Rivers, California. Pages 100-115 *in* C. Levings, L. Holtby, and M. Henderson, editors. Proceedings of the National Workshop on Effects of Habitat Alteration on Salmonid Stocks. Canadian Special Publication of Fisheries and Aquatic Sciences 105.

Kope, R.G., and L.W. Botsford. 1990. Determination of factors affecting recruitment of Chinook salmon *Oncorhynchus tshawytscha* in central California. Fishery Bulletin 88:257-269.

Kormos, B., M.L. Palmer-Zwahlen, and A. Low. 2012. Recovery of coded-wire tags from Chinook salmon in California's Central Valley escapement and ocean harvest in 2010. In California Department of Fish and Game, Fisheries Branch Administrative Report 2012-02.

Lehman, P.W., K. Marr, G.L. Boyer, S. Acuna, and S.J. Teh. 2013. Long-term trends and causal factors associated with *Microcystis* abundance and toxicity in San Francisco Estuary and implications for climate change impacts. Hydrobiologia 718(1):141-158.

Lindley, S.T., R.S. Schick, E. Mora, P.B. Adams, J.J. Anderson, S. Greene, C. Hanson, B.P. May, D. McEwan, R.B. MacFarlane, C. Swanson, and J.G. Williams. 2007. Framework for assessing viability of threatened and endangered Chinook salmon and steelhead in the Sacramento-San Joaquin Basin. San Francisco Estuary and Watershed Science 5(1). https://doi.org/10.15447/sfews.2007v5iss1art4.

Lucas, L.V., J.K. Thompson, and L.R. Brown. 2009. Why are diverse relationships observed between phytoplankton biomass and transport time? Limnology and Oceanography 54:381-390.

Lucas, L.V., and J.K. Thompson. 2012. Changing restoration rules: Exotic bivalves interact with residence time and depth to control phytoplankton productivity. Ecosphere 3(12):Art. 17.

MacWilliams, M., A.J. Bever, and E. Foresman. 2016. 3-D Simulations of the San Francisco Estuary with subgrid bathymetry to explore long-term trends in salinity distribution and fish abundance. San Francisco Estuary and Watershed Science 14(2). https://doi.org/10.15447/sfews.2016v14iss2art3

Mahardja, B., M.J. Farruggia, B. Schreier, and T. Sommer. 2017. Evidence of a shift in the littoral fish community of the Sacramento-San Joaquin Delta. PLoS ONE 12(1): e0170683. Doi:10.1371/journal.pone.0170683.

Martin, J., M.C. Runge, J.D. Nichols, B.C. Lubow, and W.L. Kendall. 2009. Structured decision making as a conceptual framework to identify thresholds for conservation and management. Ecological Applications 19:1079-1090.

Martin, B.T., A. Pike, S.N. John, N. Hamda, J. Roberts, S.T. Lindley, and E.M. Danner. 2017. Phenomenological vs. biophysical models of thermal stress in aquatic eggs. Ecology Letters 20:50-59.

Maunder, M.N., and R.B. Deriso. 2011. A state-space multistage life cycle model to evaluate population impacts n the presence of density dependence: illustrated with application to delta smelt (*Hyposmesus transpacificus*). Canadian Journal of Fisheries and Aquatic Sciences 68:1285-1306.

May, J.T., and L.R. Brown. 2002. Fish communities of the Sacramento River Basin: implications for conservation of native fishes in the Central Valley, California. Environmental Biology of Fishes 63(4):373-388.

May, J.T., L.R. Brown, A.C. Rehn, I.R. Waite, P.R. Ode, R.D. Mazor, and K.C. Schiff. 2015. Correspondence of biological condition models of California streams at statewide and regional scales. Environmental Monitoring and Assessment 187: Article 4086. https://link.springer.com/article/10.1007/s10661-014-4086-x

Mazor, R.D., A.C. Rehn, P.R. Ode, M. Engeln, K.C. Schiff, E.D. Stein, D.J. Gillett, D.B. Herbst, and C. P. Hawkins. 2016. Bioassessment in complex environments: Designing an index for consistent meaning in different settings. Freshwater Science 35:249-271.

McDonald, L.H., A.W. Smart, and R.C. Wissmar. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. EPA 910/9-91-001.

McElhany, P., M.H. Ruckelshaus, M.J. Ford, T.C. Wainwright, and E.P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-42. 156 p.

McEwan, D.R. 2001. Central Valley steelhead. Fish Bulletin 179:1-43. Sacramento, CA. California Department of Fish and Game.

Meek, M.H., Baerwald, M.R., Stephens, M.R., Goodbla, A., Miller, M.R, Tomalty, K.M.H. and B. May. 2016. Sequencing improves our ability to study threatened migratory species: Genetic population assignment in California's Central Valley Chinook salmon. Ecology and Evolution 6:7706-7716.

Mesick, C., D. Marston, and T. Heyne. 2009. Estimating recruitment for fall-run Chinook salmon populations in the Stanislaus, Tuolumne, and Merced rivers. USFWS, Sacramento, CA.

Miller, J.A., A. Gray, and J. Merz. 2010. Quantifying the contribution of juvenile migratory phenotypes in a population of Chinook salmon *Oncorhynchus tshawytscha*. Marine Ecology Progress Series 408:227-240.

Miller, W.J. 2011. Revisiting assumptions that underlie estimates of proportional entrainment of delta smelt by state and federal water diversions from the Sacramento-San Joaquin Delta. San Francisco Estuary and Watershed Science 9(1). https://doi.org/10.15447/sfews.2011v9iss1art1

Mills, T.J., and F. Fisher. 1994. Central Valley anadromous sport fish annual run-size, harvest, and population estimates, 1967-1991. California Department of Fish and Game, Inland Fisheries Technical Report. Third draft, revised August 1994.

Mohr, M.S, W.H. Satterthwaite, D.G. Hankin, P.A. Bisson, C.B. Grimes, S. Hamelberg, D.C. Hillemeier, S.T. Lindley, A. Low, M.L. Palmer-Zwahlen, J. Smith, T.H. Williams. 2017. Evaluation of alternative marking/tagging systems for hatchery produced California fall-run Chinook salmon. NOAA-TM-NMFS-SWFSC-571.

Moore, J.W., J.D. Yeakel, D. Peard, J. Lough, and M. Beere. 2014. Life-history diversity and its importance to population stability and persistence of a migratory fish: steelhead in two large North American watersheds. Journal of Animal ecology. doi: 10.1111/1365-2656.12212

Mount, J., W. Fleenor, B. Gray, B. Herbold, and W. Kimmerer. 2014. The draft Bay-Delta conservation plan: assessment of environmental performance and governance. Hastings West-Northwest Journal of Environmental Law & Policy 20:245.

Mount, J. and E. Hanak. 2018. A chance to solve the Delta quandary. Commentary published in CalMatters, November 18, 2018. https://calmatters.org/articles/commentary/my-turn-a-chance-to-finally-solve-the-delta-quandary/?utm_source=CALmatters+Newsletter&utm_campaign=fa9d97f07a-WHATMATTERS_NEWSLETTER&utm_medium=email&utm_term=0_faa7be558d-fa9d97f07a-150236181

Moyle, P. B. 2002. Inland Fishes of California. Revised and Expanded. Berkeley: University of California Press. 502 pp.

Moyle, P.B., and P.J. Randall. 1998. Evaluating the biotic integrity of watersheds in the Sierra Nevada, California. Conservation Biology 12:1318-1326.

Moyle, P.B., and M.P. Marchetti. 1999. Applications of indices of biotic integrity to California streams and watersheds. Pages 367-380 *in* T. P. Simon and R. Hughes, editors. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, Fl.

Moyle, P.B., R.D. Baxter, T. Sommer, T.C. Foin, and S.A. Matern. 2004. Biology and population dynamics of Sacramento splittail (*Pogonichthys macrolepidotus)* in the San Francisco Estuary: a review. San Francisco Estuary and Watershed Science 2(2). https://doi.org/10.15447/sfews.2004v2iss2art3

Moyle, P.B., and J.A. Israel. 2005. Untested assumptions: effectiveness of screening diversions for conservation of fish populations. Fisheries 30(5):20-28.

Moyle, P.B., P.K. Crain, and K. Whitener. 2007. Patterns in the use of a restored California floodplain by native and alien fishes. San Francisco Estuary and Watershed Science 5: Issue 3 Article 1.

Moyle, P.B., J.V.E. Katz and R.M. Quiñones. 2011. Rapid decline of California's native inland fishes: a status assessment. Biological Conservation 144:2414-2423.

Moyle, P.B., J. Hobbs, and T. O'Rear. 2012. Fishes. Pages 161-173 *in* A. Palaima, ed. Ecology, Conservation and Restoration of Tidal Marshes: The San Francisco Estuary. Berkeley: University of California Press.

Moyle, P.B., J.D. Kiernan, P.K. Crain, and R.M. Quiñones. 2013. Climate change vulnerability of native and alien freshwater fishes of California: a systematic assessment approach. PLoS One. http://dx.plos.org/10.1371/journal.pone.0063883

Moyle, P.B., R. M. Quiñones, J.V.E. Katz, and J. Weaver. 2015. Fish Species of Special Concern in California. 3rd edition. Sacramento: California Department of Fish and Wildlife. https://www.wildlife.ca.gov/Conservation/Fishes/Special-Concern

Moyle, P., R. Lusardi, P. Samuel, and J. Katz. 2017. State of the Salmonids: Status of California's Emblematic Fishes 2017. Center for Watershed Sciences, University of California, Davis and California Trout, San Francisco, CA. 579 p. https://watershed.ucdavis.edu/files/content/news/SOS%20II_Final.pdf

Moyle, P.B., J.A. Hobbs, and J.R. Durand. 2018. Delta smelt and the politics of water in California. Fisheries 43:42-51.

Murphy, D.D., P.S. Weiland, and K.W. Cummins, K.W. 2011. A critical assessment of the use of surrogate species in conservation planning in the Sacramento-San Joaquin Delta, California (USA). Conservation Biology 25:873-878.

Myers, R.A. 2001. Stock and recruitment: generalizations about maximum reproductive rate, density dependence, and variability using meta-analytic approaches. ICES Journal of Marine Science 58:937-951.

Nagarkar, M., and K. Raulund-Rasmussen. 2016. An appraisal of adaptive management planning and implementation in ecological restoration: case studies from the San Francisco Bay Delta, USA. Ecology and Society 21(2).

Naish, K.A., J.E. Taylor, P.S. Levin, T.P. Quinn, J.R. Winton, D. Huppert, and R. Hilborn. 2008. An evaluation of the effects of conservation and fishery enhancement hatcheries on wild populations of salmon. Advances in Marine Biology 53:61-194.

Newman, K.B. 2008a. Sample design-based methodology for estimating delta smelt abundance. San Francisco Estuary and Watershed Science 6(3):[np].

Newman, K.B. 2008b. An evaluation of four Sacramento–San Joaquin River Delta juvenile salmon survival studies. Stockton (CA): U.S. Fish and Wildlife Service, Project number SCI-06-G06-299.

Newman, K.B., and P.L. Brandes. 2010. Hierarchical modeling of juvenile Chinook salmon survival as a function of Sacramento–San Joaquin Delta water exports. North American Journal of Fisheries Management 30:157–169.

Newman, K.B., L. Polansky, L. Mitchell, W. Kimmerer, P. Smith, R. Baxter, W. Bennet, M. Mander, M. Nobriga, W. Meiring, E. Laca, and F. Feyrer. 2014. A delta smelt life cycle model. Draft reported prepared by USFWS Dec 17, 2014.

Nichols, F.H. 1985. Increased benthic grazing: an alternative explanation for low phytoplankton biomass in northern San Francisco Bay during the 1976-1977 drought. Estuarine Coastal and Shelf Science 21(3):379-388.

Nichols, F., J. Cloern, S. Luoma, and D. Peterson. 1986. The modification of an estuary. Science 231:567-573.

Nicholson, E., D.B. Lindenmayer, K, Frank, and H.P. Possingham. 2013. Testing the focal species approach to making conservation decisions for species persistence. Diversity and Distributions 19:530-540.

Nixon, S.W., C.A. Oviatt, J. Frithsen, and B. Sullivan. 1986. Nutrients and the productivity of estuarine and coastal marine systems. Journal of the Limnological Society of South Africa 12:43-71.

Novick, E., R. Holleman, T. Jabusch, J. Sun, P. Trowbridge, D. Senn, M. Guerin, C. Kendall, M. Young, and S. Peek. 2015. Characterizing and quantifying nutrient sources, sinks and transformations in the Delta: synthesis, modeling, and recommendations for monitoring. San Francisco Estuary Institute, Richmond, CA.

Ode, P.R., A.C. Rehn, R.D. Mazor, K.C. Schiff, E.D. Stein, J.T. May, L.R. Brown, D.B. Herbst, D. Gillett, K. Linde, and C.P. Hawkins. 2016. Evaluating the adequacy of a reference-site pool for ecological assessments in environmentally complex regions. Freshwater Science 35:237-248.

Okansen, J., G. Blanchet, and 11 others. 2018. vegan: Community Ecology Package. R package version 2.5-3.

Opperman, J.J., P.B. Moyle, E.W. Larsen, J.L. Florsheim, and A.D. Manfree. 2017. Floodplains: Processes and Management for Ecosystem Services. University of California Press. 258 p.

Owens, S., T. Ignoffo, J. Frantzich, A. Slaughter, and W. Kimmerer. High growth rates of a dominant calanoid copepod in the northern San Francisco Estuary and their positive relationship to cyanobacteria . In prep. Journal of Plankton Research.

Paerl, H.W., N.S. Hall, B.L. Peierls, K.L. Rossignol, and A.R. Joyner. 2014. Hydrologic variability and its control of phytoplankton community structure and function in two shallow, coastal, lagoonal ecosystems: The Neuse and New River Estuaries, North Carolina, USA. Estuaries and Coasts 37(1):S31-S45.

Palmer-Zwahlen, M.L., and Kormos, B. 2013. Recovery of coded-wire tags from Chinook salmon in California's Central Valley Escapement and Ocean Harvest in 2011. In California Department of Fish and Game, Fisheries Branch Administrative Report 2013-02.

Palmer-Zwahlen, M.L., and B. Kormos. 2015. Recovery of coded-wire tags from Chinook salmon in California's Central Valley escapement, inland harvest, and ocean harvest in 2012. In California Department of Fish and Game, Fisheries Branch Administrative Report 2015-4.

Palmer-Zwahlen, M., V. Gusman, and B. Kormos. 2018. Recovery of coded-wire tags from Chinook salmon in California's Central Valley escapement, inland harvest, and ocean harvest in 2013. Pacific States Marine Fisheries Commission and California Department of Fish and Game.

Palmer, M.A., E.S. Bernhardt, J.D. Allan, P.S. Lake, G. Alexander, S. Brooks, J. Carr, S. Clayton, C.N. Dahm, J. Follstad-Shah, D.L. Galat, S.G. Loss, P. Goodwin, D.D. Hart, B. Hassett, R. Jenkinson, G.M. Kondolf, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, and E. Sudduth. 2005. Standards for ecologically successful river restoration. Journal of Applied Ecology 42:208-217.

Parken, C.K., R.E. McNicol, and J.R. Irvine. 2006. Habitat-based methods to estimate escapement goals for data limited Chinook salmon stocks in British Columbia, 2004. Canadian Science Advisory Secretariat Research Document 2006/083.

Parker, A., W. Kimmerer, and U. Lidström. 2012. Reevaluating the generality of an empirical model for light-limited primary production in the San Francisco Estuary. Estuaries and Coasts 35(4):930-942.

Perry, R.W., A.C. Pope, J.G. Romine, P.L. Brandes, J.R. Burau, A.R. Blake, A.J. Ammann, and C.J. Michel. 2018. Flow-mediated effects on travel time, routing, and survival of juvenile Chinook salmon in a spatially complex, tidally forced river delta. Canadian Journal of Fisheries and Aquatic Sciences 75:1886-1901.

Peterman, R.M., and B. Dorner. 2012. A widespread decrease in productivity of sockeye salmon populations in western North America. Canadian Journal of Fisheries and Aquatic Sciences 69(8):1255–1260.

Peterman, R.M., B.J. Pyper, and B.W. MacGregor. 2003. Use of the Kalman filter to reconstruct historical trends in productivity of Bristol Bay sockeye salmon. Canadian Journal of Fisheries and Aquatic Sciences 60(7):809–824.

Pilger, T.J., M.L. Peterson, D. Lee, A. Fuller, and D. Demko. 2019. Evaluation of long-term mark-recapture data for estimating abundance of juvenile fall-run Chinook salmon on the Stanislaus River from 1996 to 2017. San Francisco Estuary and Watershed Science 17(1). https://doi.org/10.15447/sfews.2019v17iss1art4

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The Natural Flow Regime. BioScience 47:769-784.

PSTRT (Puget Sound Technical Recovery Team). 2002. Planning ranges and preliminary guidelines for the delisting and recovery of the Puget Sound Chinook salmon evolutionarily significant unit. http://www.nwfsc.noaa.gov/trt/puget_docs/trtpopesu.pdf.

R Core Team (2014). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Railsback, S. 2016. Why it is time to put PHABSIM out to pasture. Fisheries 41(12):20–725.

Railsback, S.F., M. Gard, B.C. Harvey, J.L. White, and J.K.H. Zimmerman. 2013. Contrast of degraded and restored stream habitat using an individual-based salmon model. North American Journal of Fisheries Management 33(2):384-399.

Reaugh, M.L., M.R. Roman, and D.K. Stoecker. 2007. Changes in plankton community structure and function in response to variable freshwater flow in two tributaries of the Chesapeake Bay. Estuaries and Coasts 30(3):403-417.

Rehn, A.C. 2009. Benthic macroinvertebrates as indicators of biological condition below hydropower dams on west slope Sierra Nevada streams, California, USA. River Research and Application 25:208-228.

Reis, G.J, J.K. Howard, and J.A. Rosenfield. 2019. Clarifying effects of environmental protections on freshwater flows to—and water exports from—the San Francisco Bay Estuary. San Francisco Estuary and Watershed Science 17(1). https://doi.org/10.15447/sfews.2019v17iss1art1

Richter, B.D., J.V. Baumgartner, J. Powell, and D.P. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology 10:1163-1174.

Ricklefs, R.E. 1982. Ecology, 2nd Edition. Chiron Press, New York & Concord. 966 pp.

Riddell, B., Bradford, M., Carmichael, R., Hankin, D. Peterman, R. and A. Wertheimer. 2013. Assessment of Status and Factors for Decline of Southern BC Chinook Salmon: Independent Panel's Report. Prepared for Fisheries and Oceans Canada (Vancouver. BC) and Fraser River Aboriginal Fisheries Secretariat (Merritt, BC). 165 p.

Rieman, B.E., C.L. Smith, R.J. Naiman, G.T. Ruggerone, C.C. Wood, N. Huntly, E.N. Merrill, J.R. Alldredge, P.A. Bisson, J. Congleton, K.D. Fausch, R. Lamberson, C. Levings, W. Pearcy, D. Scarnecchia, P. Smouse. 2015. A comprehensive approach for habitat restoration in the Columbia Basin. Fisheries 40:124-135.

Roos, M. 1989. Possible climate change and its impact on water supply in California. Oceans '89: 247-249.

Rose, K.A., and J.K. Summers. 1992. Relationships among long-term fisheries abundances, hydrographic variables, and gross pollution indicators in northeastern U.S. estuaries. Fisheries Oceanography 1:281-293.

Rose, K.A., and J.H. Cowan Jr. 1993. Individual-based model of young-of-the-year striped bass population dynamics. I. Model description and baseline simulations. Transactions of the American Fisheries Society 122(3): 415-438.

Rose, K.A., J.H. Cowan, K. Winemiller, R.A. Myers, and R. Hilborn. 2002. Compensatory density dependence in fish populations: importance, controversy, understanding and prognosis. Fish and Fisheries 2:293-327.

Rose, K.A., W.J. Kimmerer, K.P. Edwards, W.A. Bennett. 2013. Individual-based modeling of delta smelt population dynamics in the upper San Francisco Estuary: I. Model description and baseline results. Transactions of the American Fisheries Society 142(5):1238-1259.

Rose, K.A., W.J. Kimmerer, K.P. Edwards, and W.A. Bennett. 2013. Individual-based modeling of delta smelt population dynamics in the upper San Francisco Estuary: II. Alternative baselines and good versus bad years. Transactions of the American Fisheries Society, 142(5):1260-1272.

San Francisco Bay Area Wetlands Ecosystem Goals Project. 2015. The Baylands and Climate Change: What We Can Do. Baylands Ecosystem Habitat Goals Science Update 2015. California State Coastal Conservancy, Oakland, CA. https://baylandsgoals.org/science-update-2016/

SBSRF (Snohomish Basin Salmon Recovery Forum). 2005. Snohomish River Basin salmon conservation plan. Snohomish County Department of Public Works, Surface Water Management Division. Everett, WA. https://snohomishcountywa.gov/1127/Snohomish-Watershed-Salmon-Recovery-Plan

Schindler, D.E., R. Hilborn, B. Chasco, C.P. Boatright, T.P. Quinn, L.A. Rogers, L.A. et al. 2010. Population diversity and the portfolio effect in an exploited species. Nature 465:609–612.

Schoellhamer, D.H. 2011. Sudden clearing of estuarine waters upon crossing the threshold from transport to supply regulation of sediment transport as an erodible sediment pool is depleted: San Francisco Bay, 1999. Estuaries and Coasts 34:885-899.

Schoellhamer, D.H., S.A. Wright, and J.Z. Drexler. 2013. Adjustment of the San Francisco estuary and watershed to decreasing sediment supply in the 20th century. Marine Geology 345:63-71.

Sengupta, A., S.K. Adams, B.P. Bledsoe, E.D. Stein, K.S. McCune, and R.D. Mazor. 2018. Tools for managing hydrologic alteration on a regional scale: Estimating changes in flow characteristics at ungauged sites. Freshwater Biology 63:769-785.

SEP Group 2016. Conservation planning foundation for restoring Chinook salmon (*Oncorhynchus tshawytscha*) and *O. mykiss* in the Stanislaus River. Report prepared by Anchor QEA, LLC, Seattle, Washington.

Shuford, W.D. and K.E. Dybala. 2017. Conservation objectives for wintering and breeding waterbirds in California's Central Valley. San Francisco Estuary and Watershed

Science 15(1). http://escholarship.org/uc/item/5tp5m718

Sobczak, W.V., J.E. Cloern, A.D. Jassby, B.E. Cole, T.S. Schraga, and A. Arnsberg. 2005. Detritus fuels ecosystem metabolism but not metazoan food webs in San Francisco estuary's freshwater Delta. Estuaries 28(1):124-137.

Sommer, T., R. Baxter, and B. Herbold. 1997. Resilience of splittail in the Sacramento-San Joaquin Estuary. Transactions of the American Fisheries Society 126:961-976.

Sommer, T., B. Harrell, M. Nobriga, R. Brown, P. Moyle, W. Kimmerer, and L. Schemel. 2001. California's Yolo Bypass: Evidence that flood control can be compatible with fisheries, wetlands, wildlife, and agriculture. Fisheries 26(8):6-16.

Sommer, T. R., L. Conrad, G. O'Leary, F. Feyrer, and W. C. Harrell. 2002. Spawning and Rearing of Splittail in a Model Floodplain Wetland. Transactions of the American Fisheries Society 131(5):966-974.

Sommer, T., C. Armor, R. Baxter, R. Breuer, L. Brown, M. Chotkowski, S. Culberson, F. Feyrer, M. Gingras, B. Herbold, W. Kimmerer, A. Mueller-Solger, M. Nobriga, and K. Souza. 2007. The collapse of pelagic fishes in the upper San Francisco Estuary. Fisheries 32(6):270-277.

Sommer, T., and F. Mejia. 2013. A place to call home: a synthesis of delta smelt habitat in the upper San Francisco Estuary. San Francisco Estuary and Watershed Science 11(2). https://doi.org/10.15447/sfews.2013v11iss2art4

SSDC (Shared Strategy Development Committee). 2007. Puget Sound salmon recovery plan. Adopted by National Marine Fisheries Service, 19 January 2007. https://www.westcoast.fisheries.noaa.gov/protected_species/salmon_steelhead/recovery_planning_and_implementation/puget_sound/puget_sound_chinook_recovery_plan.html

Staton, B.A., Catalano, M.J., and S.J. Fleischman. 2017. From sequential to integrated Bayesian analysis: Exploring the continuum with a Pacific salmon spawner-recruit model. Fisheries Research 186: 237-247.

Stewart, I.T., D.R. Cayan, and M.D. Dettinger. 2004. Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Climatic Change 62(1-3):217-232.

Stewart, I.T., D.R. Cayan, and M.D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155.

SWRCB (California State Water Resources Control Board). 2000. Revised Water Right Decision 1641. Sacramento, CA. https://water.ca.gov/-/media/DWR-Website/Web-Pages/Programs/Environmental-Services/Bio-Monitoring--Assessment/Files/State-Water-Resources-Control-Boards-Water-Right-Decision-D1641.pdf

SWRCB. 2018. Substitute environmental document in support of potential changes to the water quality control plan for the San Francisco Bay-Sacramento San Joaquin Delta

Estuary. San Joaquin River flows and Southern Delta water quality. State Clearinghouse No. 2012122071.

Ta, J., L.W.J. Anderson, M.A. Christman, S. Khanna, D. Kratville, J.D. Madsen, P.J. Moran, and J.H. Viers. 2017. Invasive aquatic vegetation management in the Sacramento–San Joaquin River Delta: status and recommendations. San Francisco Estuary & Watershed Science 15(4). https://doi.org/10.15447/sfews.2017v15iss4art5

Tank, J.L., E.J. Rosi-Marshall, N.A. Griffiths, S.A. Entrekin, and M.L. Stephen. 2010. A review of allochthonous organic matter dynamics and metabolism in streams. Journal of the North American Benthological Society 29:118-146.

Thompson, T.Q., M.R. Bellinger, S.M. O'Rourke, D.J. Prince, A.E. Stevenson, A.T. Rodrigues, M.R. Sloat, C.F. Speller, D.Y. Yang, V.L. Butler, M.A. Banks, and M.R. Miller. 2019. Anthropogenic habitat alteration leads to rapid loss of adaptive variation and restoration potential in wild salmon populations. Proceedings of the National Academy of Sciences 116:177-186.

Underwood, A.J. 1992. Beyond BACI - the detection of environmental impacts on populations in the real, but variable, world. Journal of Experimental Marine Biology and Ecology 161:145-178.

USFWS. 1996. Recovery plan for the Sacramento-San Joaquin Delta native fishes. US Fish and Wildlife Service, Portland, Oregon. 193 pp.

USFWS. 2001. Final restoration plan for the Anadromous Fish Restoration Program (AFRP): a plan to increase natural production of anadromous fish in the Central Valley California. Prepared for the Secretary of the Interior by the United States Fish and Wildlife Service with assistance from the Anadromous Fish Restoration Program Core Group under authority of the Central Valley Project Improvement Act. January 9, 2001.

Von Schiller, D., V. Acuña, I. Aristi, M. Arroita, A. Basaguren, A. Bellin, L. Boyero, A. Butturini, A. Ginebreda, E. Kalogianni, A. Larrañaga, B. Majone, A. Martínez, S. Monroy, I. Muñoz. M. Paunovíc, O. Pereda, M. Petrovic, J. Pozo, S. Rodríguez-Mozaz, D. Rivas, S. Sabater, F. Sabater, N. Skoulikidis, L. Solagaistua, L. Vardakas, and A. Elosegi. 2017. Science of the Total Environment 596-597:465-480.

Walters, C.J. 1997. Challenges in adaptive management of riparian and coastal ecosystems. Conservation Ecology 1(2).

Waples, R.S., T. Beechie, and G.R. Pess. 2009. Evolutionary history, habitat disturbance regimes, and anthropogenic changes: what do these mean for resilience of Pacific salmon populations? Ecology and Society 14(1):3.

Weston, D.P., D. Schlenk, N. Riar, M.J. Lydy, and M.L. Brooks. 2015. Effects of pyrethroid insecticides in urban runoff on Chinook salmon, steelhead trout, and their invertebrate prey. Environmental Toxicology and Chemistry 34:649-657.

Whipple, A. A., R. M. Grossinger, D. Rankin, B. Stanford, and R. Askevold. 2012. Sacramento-San Joaquin Delta historical ecology investigation: Exploring pattern and process. Richmond: San Francisco Estuary Institute-Aquatic Science Center.

Williams, J.G., P.B. Moyle, M. Kondolf, and A. Webb. 2019. Environmental Flow Assessment: Methods and Applications. Wiley-Blackwell. 240p.

Williamson, K.S. and B. May. 2005. Homogenization of fall-run Chinook salmon gene pools in the Central Valley of California, USA. North American Journal of Fisheries Management 25:993-1009.

Willmes, W., J.A. Hobbs, A.M. Sturrock, Z. Bess, L.S. Lewis, J.J.G. Glessner, R.C. Johnson, R.Kurth, and J. Kindopp. 2018. Fishery collapse, recovery, and the cryptic decline of wild salmon on a major California river. Canadian Journal of Fisheries and Aquatic Sciences 75:1836–1848.

Yarnell, S.M., G.E. Petts, J.C. Schmidt, A.A. Whipple, E.E. Beller, C.N. Dahm, P. Goodwin, and J.H. Viers. 2015. Functional flows in modified riverscapes: hydrographs, habitats, and opportunities. BioScience 65:963-972.

Yebra, L., T. Kobari, A.R. Sastri, F. Gusmao, and S. Hernandez-Leon. 2017. Advances in Biochemical Indices of Zooplankton Production. Pages 157-240 *in* B. E. Curry, editor. Advances in Marine Biology, volume 76.

York, J.L., B.A. Costas, and G.B. McManus. 2011. Microzooplankton grazing in green water—results from two contrasting estuaries. Estuaries and Coasts 34:373-385.

Young, R.G., C.D. Matthaei, and C.R. Townsend. 2008. Organic matter breakdown and ecosystem metabolism: Functional indicators for assessing river ecosystem health. Journal of the North American Benthological Society 27:605-625.

Young, M.J., K.A. Berridge, T. O'Rear, P.B. Moyle, and J.R. Durand. 2017. Habitat partitioning by native and alien fishes and decapods in novel habitats of the upper San Francisco Estuary. Biological Invasions 19:2693–2710.

Young, M.J., F.W. Feyrer, D.D. Colombano, J.L. Conrad, and A. Sih. 2018. Fish-habitat relationships along the estuarine gradient of the Sacramento-San Joaquin Delta, California: implications for habitat restoration. Estuaries and Coasts https://doi.org/10.1007/s12237-018-0417-4

Zabel, R.W., and P.S. Levin. 2002. Simple assumptions on age composition lead to erroneous conclusions on the nature of density dependence in age-structured populations. Oecologia 133:349-355.

Zedler, J. B., and S. Kercher. 2005. Wetland resources: status, trends, ecosystem services, and restorability. Annual Review of Environment and Resources 30:39-74.

Zimmerman, J.K.H., D.M. Carlisle, J.T. May, K.R. Klausmeyer, T.E. Grantham, L.R. Brown, and J.K. Howard. 2018. Patterns and magnitudes of flow alteration in California, USA. Freshwater Biology 63:859-873.

9 APPENDICES

9.1 Charge Questions and Brief Answers

The materials found in Chapters 2, 3, and 4 are designed to answer most of the Board's charge questions to the Panel. Specific questions, however, might require some additional attention. To further address these details, the Panel took the liberty to reorganize the order of the original charge questions because our report discusses ecosystem before salmon issues. The title of the chapters and some original ecosystem questions were modified to provide clarity. In the answers below, references to specific sections of the main report are provided when additional information is available.

9.1.1 Chapter 2. Ecosystem Structure and Function

The most suitable measures for monitoring of ecosystem state and its response to manipulations are also those most directly aligned to the goals of the management action. Drawing an analogy, a physician will assess human health from straightforward observations of the state of the body (e.g., weight, heart rate, blood pressure and chemistry), while the myriad processes underlying health would be investigated only if there is a problem. For the ecosystem, measures of condition and response that emphasize system state and trajectory and processes relevant to supporting native fishes by providing physical habitat, food requirements, and good water quality are preferred.

> **Charge question 1.** What approaches do the Panel recommend for establishing abiotic and biotic goals for ecosystem structure and function in the Bay-Delta system to assess the effectiveness of flow modification, habitat restoration, and other non-flow actions in meeting the narrative objectives?

The focus here is on structural and functional components of the Bay-Delta ecosystem. In general, structural elements (e.g., temperature, hydrology, salinity, turbidity, dissolved oxygen, etc.) are currently better measured than functional elements (e.g., primary production, ecosystem respiration, secondary production, decomposition rates, nutrient cycling rates, predation rates and patterns, etc.), and the upper estuary including the Delta is much better monitored than the tributary rivers. A logical approach to establishing goals for the estuary is therefore to examine the content of existing monitoring programs and design goals to assess the availability of habitat support for fishes discussed in other chapters. The Panel recognizes that there are good arguments for adding more functional measurements to assessment metrics, but the data base to which to compare is often sparse. Evaluating the effectiveness of management actions like flow modification and habitat restoration could profit by looking at existing programs where long-term evaluation success, linked to restoration projects or flow modification changes, has been achieved.

Charge question 2. Would some of the processes listed below be appropriate for monitoring the progress of biological goals pertaining to ecosystem structure and function? Which among these or other attributes of ecosystem structure and function would be the most informative for monitoring progress towards achieving biological goals?

- Nutrient concentrations and cycling pathways
- Indicators of primary and secondary productivity
- Spatial-temporal dynamics (e.g., habitat connectivity, species distributions through time)
- Community dynamics (e.g., food webs, community composition, cross-habitat interactions, new introductions)

For river ecosystems, a recent synthesis paper by von Schiller et al. (2017) looks at approaches, criteria for use, and sensitivity to environmental stressors of various ecosystem processes. This synthetic review paper is very helpful in discussing which ecosystem processes in rivers are most responsive to biotic and abiotic stressors. Recommended ecosystem processes to consider in functional assessments include organic matter decomposition, nutrient cycling, metabolism, pollutant dynamics, and community dynamics. While the four categories listed in the charge question are very broad, certain processes within each category might have utility to ascertaining ecosystem response to specific stressors. As a monitoring program is developed or expanded, the utility of specific metrics drawn from these four categories should be discussed and evaluated for inclusion based on their merits.

For the estuary, estimates of productivity by phytoplankton (primary) and by zooplankton (not strictly secondary) can be made from existing data and used for goals indicating support of the ecosystem for fishes of concern. The other suggested attributes are of interest less for setting goals than for research and for exploring reasons for changes in productivity.

> **Charge question 3.** Among existing monitoring programs, which measures of ecosystem structure and function are likely to be sensitive to proposed changes (e.g., flow changes)? What locations (e.g., rivers, Delta, lower estuary) would be most suitable as the focus of analysis?

A difficulty in answering this question is the Panel's limited familiarity with all existing monitoring programs, especially for the tributary rivers. River monitoring programs elsewhere, where flow modification and habitat restoration are at the forefront of management efforts, have included a mix of structural and functional components. For example, the Healthy Waterways Initiative chose metrics that were in five categories (physical and chemical indicators, algal bioassays, macroinvertebrates, fish, and ecosystem processes) and the Kissimmee River Restoration Evaluation Program metrics were in eight categories (hydrology and geomorphology, water quality, algae, aquatic plants, invertebrates, herpetofauna, fish, and birds). Which structural and functional components of the ecosystems to measure, as the location moved from tributary rivers to main stem rivers to the Delta to the Bay, changes as the physical configuration, flow dynamics, salinity, and community structure changes. Moreover, mechanisms for effects of changes in flow shift from direct in the rivers (e.g., transport, turbulence, habitat heterogeneity) to indirect in the estuary especially in brackish to saline waters (e.g., stratification, density-driven circulation, retention of organisms). Therefore, it is difficult to answer this question without specifics as to the planned changes and their location.

> **Charge question 4.** What additional monitoring would provide clarity to the responses of the ecosystem to proposed changes? This would require that the measurements be feasible, practicable (e.g., not too expensive), and sensitive to these changes.

Additional monitoring would need to be linked to existing monitoring and to gaps in areas (topical, geographic, or temporal) likely to be affected by changes. Without information on these areas it would be premature to suggest specific additional monitoring. In addition, monitoring design should consider attributes of ecosystems, other fishes, and salmon. For example, because of the emphasis and interest in other fishes and salmon, the ecosystem structure and function components should be factors critical and of concern for the fish assemblages. Structural elements such as temperature, dissolved oxygen, turbidity, salinity (conductivity), and toxic constituents to fish should be monitored. Other structural components such as pathogens, algal community structure, aquatic invertebrate structure, and fish community structure are common monitoring constituents in well-regarded monitoring and evaluation programs (see Box 2.2 on the Healthy Waterways Initiative Program in Australia), and might be of use here. What is feasible, practicable, and sensitive depends upon the ecosystem under consideration and the budget available for monitoring and evaluation. The Panel notes that decisions on what to monitor are strongly linked to budget, and clarity on the budget brings the decision-making on what to monitor very much into focus.

> **Charge question 5.** What measurable structural and functional attributes of riverine and estuarine ecosystems provide the greatest clarity about the responses of these ecosystems to modifications like restoration projects or more functional flows?

Again, the measurable structural and functional attributes of rivers that respond to functional flows and restoration projects must consider: 1) the characteristics of the ecosystem to be modified; and, 2) the body of literature on successful restoration efforts worldwide where flow and habitat have been modified for restoration goals. These successful riverine projects that are both long-term and set clear goals are limited, but they do provide state of practice guidelines. In the estuary, the greatest clarity comes from observing attributes most closely linked to the management activity, such as colonization of restored habitats by wanted and unwanted taxa, use of those habitats by fishes of interest, and interaction of those habitats with the surrounding waters. The effects of flow manipulations depends on their magnitude, duration, timing, and locus, but again one should observe features of the ecosystem that are expected to respond sharply, such as salinity gradients and corresponding patterns of species abundance and movement.

> **Charge question 6.** How might flow or non-flow actions interact across the landscape, or with other long-term changes in the rivers and estuary, including local anthropogenic change and change imposed by climate? To what extent will interactions amplify or obscure ecosystem responses to the actions?

This is a very difficult, complex question with open-ended components. Climate change alone opens so many pathways and confounding interactions that most certainly will bring major impacts on this fluvial system. Flow and non-flow interactions are intertwined in so many ways that defining a flow or non-flow action that does not interact is nearly impossible. Feedbacks involving flow conditions, climate change, human activity, and landscape ecology present a nearly infinite number of potential interactions.

The Panel offer a few apparent or emerging examples where climate change and local anthropogenic change interact in ways that affect native fishes in the major tributary rivers and the estuary. One example is the interactions among drought, heat waves, and river temperature management. Warming superimposed with episodic heat waves at times of low flow is a growing concern that will get worse. Another example is the impact of catastrophic forest fire and forest die-back on turbidity and water quality with direct impacts on fish and indirect impacts on food webs. Another interesting example of interactions between a non-flow action with a flow action is the improvement in floodplain habitat and configuration, such that increased river flows will produce more regular floodplain inundation. Two-dimensional flow-inundation mapping can help characterize the extent and duration of flooding and model flow dynamics on the flooded landscape to assist in designing restoration and flow management activities. Another emerging example is the size of future mountain snow packs, the timing of snowmelt runoff, and the decreased water yields due to enhanced sublimation and evapotranspiration. In the estuary, effects of sea-level rise have many challenging facets including infrastructure protection, the need for increased freshwater outflows for salinity management in the Delta, and flow-based management activities for estuarine fishes.

9.1.2 Chapter 3. Native Fishes and Fish Assemblages

The following are questions posed in the charges to the Panel that we address in the native fishes and fish assemblages chapter (Chapter 3). Brief answers to each question are also provided, to direct readers to appropriate sections of this report.

> **Charge Question 1.** What, if any, of the VSP parameters could be applied to the recovery of other native fish species under the existing monitoring programs?

The typical use of VSP parameters does not appear to be appropriate for fishes other than salmon and steelhead. The information required to assess attributes of abundance, life history and genetic diversity, productivity, and spatial structure is largely lacking with a few possible exceptions (e.g., delta smelt).

> **Charge Question 2.** If VSP parameters are not applicable to other native fishes, what, if any, suitable alternative parameters can the Panel recommend to assess the trend and status of other native fishes.

Parameters most available for non-salmonid fishes are related to their distribution and abundance in the diverse surveys of Delta and riverine fish (Section 3.2).

> **Charge Question 3.** What, if any, population and/or community metrics would the Panel recommend to assess the condition of other native fishes over time, given that

many species-specific biological parameters may not be monitored under the existing network of monitoring programs?

Most of the chapter is devoted to answering this question with specific metrics proposed in Section 3.3.

Charge Question 4. Would the Panel recommend grouping the other native fishes according to ecological similarities such as life-history, functional groups or habitat associations?

Such a recommendation is discussed in Section 3.4.

Charge Question 5. Can the panel provide advice on the umbrella species concept and whether this conservation strategy can be applied to the native fishes of the Bay-Delta system.

The umbrella species concept represents a good possible approach, which is encapsulated in some respect by the organization of fish assemblages, but the Panel determined that the broad term "indicator species" was more applicable. Umbrella species typically require large areas of habitat to persist, so their conservation results in non-target species being conserved at the same time, in a subset of habitats. As described by the Panel, indicator species are those species that are especially sensitive to changing conditions and are abundant enough so they can represent responses of other, perhaps less abundant species that similarly occupy the same general habitat. In this case, the main concern is for Delta indicator species, e.g., those fishes that will have detectable responses to changing conditions in the Delta and inflowing rivers that affect listed or special concern species, especially in fresh water. Exploring the use of focal species, a concept developed by groups working on bird conservation, is also recommended. Focal species are species chosen for monitoring as a group, because the species have diverse habitat requirements, ideally encompassing all of the characteristics of the broad habitat (e.g. riparian forest, Delta) that is being protected. (Section 3.6, Multi-species Metrics).

Charge Question 6. What additional vital rates or population parameters should be monitored to improve measurements of the status of native fishes for the purpose of developing and assessing biological goals?

Some additional possibilities are provided in Section 3.7.

Charge Question 7. What criteria or trends would the Panel recommend including as biological goals to determine if the recovery of native fishes is succeeding in the Bay-Delta and its tributaries?

This is the main focus of Sections 3.5-3.7.

Charge Question 8. What additional habitats or locations should be included beyond the existing monitoring network?

The existing monitoring network covers the Delta relatively well but it is clear that more monitoring approaches and spatial distributions of resident, non-pelagic fishes in the Delta are needed. Systematic monitoring of fish populations of all species (not just salmonids) is needed in tributary rivers below dams (e.g., Tuolumne, Feather, Merced, Mokelumne), or if such monitoring already exists, make the data readily accessible. It is worth noting that many of the existing monitoring programs have stations that are west of the Delta, in Suisun, San Pablo and San Francisco Bays, where a number of Delta fishes spend part of their life cycle.

9.1.3 Chapter 4. Salmon and Steelhead

The following are questions summarized from the charge. Brief answers are provided to direct readers to appropriate parts of the salmon and steelhead chapter.

> **Charge Question 1.** Is the approach taken by the SEP (SEP Group 2016) to estimate VSP parameters—productivity, life history diversity, genetic diversity, spatial structure, and hatchery vs. wild metrics—a suitable method for assessing progress toward achieving the narrative objectives to protect native salmonids in the Sacramento and San Joaquin River watersheds?-What are the strengths and weaknesses of the SEP approach? Does the Panel recommend that separate biological goals be created for each VSP parameter, or that one unifying goal be established based on the cumulative information provided by all the parameters?

VSP criteria are well-established tools for evaluating viability of salmonid populations, as discussed in Section 4.2. However, density dependence is important even in depleted populations, so evaluating VSP within a density dependence framework (see Section 4.2, 4.3, 4.4) rather than excluding values when density is high, as suggested by SEP Group (2016), is recommended. Separate biological goals should be created for each VSP parameter so that progress towards goals associated with each parameter can be independently evaluated (also see Section 1.6).

The SEP Group (2016) report is thorough, well written, and contains valuable information but is somewhat cumbersome given its length (471 pages), including a 60-page summary. Comprehensive comments on the long SEP report are beyond the scope of this effort. The Panel offers the following comments while emphasizing that it is much easier to identify a few critical comments than to discuss many positive comments:

- The Panel believes the doubling goal (USFWS 2001) is unlikely to be achieved for the following reasons: 1) The estimated abundance of natural origin Chinook salmon during the baseline period (1967-1991) is likely too high. Many hatchery-origin salmon are likely included in estimates of returning natural-origin salmon because methods used to separate returns of hatchery- and natural-origin salmon seem to be based on professional opinion (Mills and Fisher 1994). They do not seem to be consistent with recent estimates based on tagging. 2) Density dependence in relation to available habitat conditions may constrain smolt production. 3) Low survival in the estuary and ocean limits adult abundance. And, 4) Projected climate change and human population growth will likely constrain potential gains in salmon survival and abundance. The objective to increase productivity sufficiently to enable the natural-origin Chinook salmon populations to double within 10 years is highly unlikely. As discussed in Chapter 4, time needed to make significant progress towards salmon recovery takes much longer than typically assumed.

- Overall, the Panel finds that the SEP report identifies too many VSP objectives for juvenile salmonids, especially for life history diversity. How will these objectives be used within an adaptive management framework and which objectives will carry the most weight? What management actions are expected to influence these metrics? Keeping track of progress toward all of these objectives will require considerable effort. The monitoring program should be designed such that it can be funded and implemented indefinitely. In contrast to the all-encompassing approach described by the SEP report, the Panel approach to VSP monitoring and objectives is intentionally simple and focuses on key fundamental objectives and means objectives related to population viability.
- The Panel agrees that developing goals for freshwater life stages is very important (e.g., spawner-to-smolt survival; see Section 4.3.3 and Appendix 9.5; Johnson et al. 2017) and understands the logic to focus on life stages over which the SEP effort has most control. Nevertheless, as generally recognized by SEP, goals associated with the entire life cycle of salmonids are absolutely critical when evaluating population viability. Additionally, the general public is most interested in the number of returning adult salmonids. A major challenge for recovery of salmonid populations is to coordinate efforts that influence salmonid abundance throughout their life cycle, including efforts in the estuary and through fisheries management. Nevertheless, as discussed in our main text, evaluating productivity during the spawner to smolt stage may provide the quickest and least confounded approach for documenting the response of salmonids to management actions in the tributaries. However, these management actions could be meaningless if survival in the estuary and ocean approaches zero and few natural origin adults return to spawn.
- One of the genetic diversity goals in the SEP report for fall-run Chinook salmon is having hatchery fish representing less than 20% of total spawners (pHOS). As discussed in Chapter 4, this high pHOS level will impede recovery of genetic diversity and population productivity. A weir or other capture/diversion techniques could be used to: 1) recycle hatchery fish back to sport fisheries; 2) allow "natural" salmon to spawn upstream; and 3) select "natural" salmon for broodstock when numbers are sufficient. However, the present 25% marking of fall-run Chinook salmon limits this approach. The Panel encourages hatcheries to mark all hatchery Chinook salmon as a means to enable selective removal when possible and to potentially improve abundance estimates for natural-origin Chinook salmon, especially when sampling rates are small (Cal-Nev AFS 2009, CA HSRG 2012, Mohr et al. 2017).

- The Panel questions whether recovery of spring-run Chinook salmon below Goodwin Dam is a reasonable goal given that the population spawned primarily above this area, avoiding introgression with fall run Chinook salmon. Recent genomic studies show that the spring run "premature return" life history is governed by a single gene complex, which is inherited such that introgression with abundant hatchery- and natural-origin fall run Chinook salmon would adversely affect this gene complex (Thompson et al. 2019). However, the presence of apparent '"hybrid" spring/fall-run salmon below Oroville Dam on the Feather River suggests that selection for the spring-run phenotype, regardless of genotype, might be possible.
- The word "stressors" is used throughout the report as equivalent to "limiting factors" in other reports. Usually, stressors refer to factors that affect individual fish whereas limiting factors are those that affect entire populations.

  Charge Question 2. If the SEP's VSP parameters provide a suitable approach for the development of biological goals, what specific metrics should be monitored for each VSP parameter? Below is a non-exhaustive list of some metrics for each VSP parameter. Are these metrics appropriate, and more importantly, can the Panel provide additional metrics that are not included? Which of the VSP parameters would provide the most useful information on whether the narrative objective for salmonids is successful?

All of the metrics listed could be monitored as a means to fully evaluate population dynamics of salmon and steelhead populations, but several metrics as the focus for establishing biological goals and for evaluating progress towards those goals are recommended (Section 4.6). Although both diversity and spatial structure are important and should be monitored, the Panel recommends that key biological goals stem from abundance and intrinsic productivity metrics because they also reflect diversity and spatial structure (Section 4.2). Biological goals should be established for the proportion of hatchery-origin spawners in the population (pHOS), because this metric has critical implications for evaluating abundance and productivity of natural populations (Sections 4.4, 4.6).

> **Charge Question 3.** Several VSP parameters may vary spatially and temporally with environmental factors, including California's highly variable hydrology. How should analysis of such unpredictable conditions be integrated into the development of biological goals and the evaluation of whether progress is being made in meeting the narrative objectives? For example, should water-year type be a significant factor to consider when assessing and evaluating the progress of biological goals, or should the assessment of biological goals remain consistent over time regardless of the water-year type? If water year type should be a significant consideration when assessing and evaluating the progress of biological goals, how should biological goals appropriately consider the water year type?

The Panel recommends an approach that specifically tests, preferably through active adaptive management experimentation, for the impact of management actions on population productivity and abundance (section 4.3). This approach can account for variation in flow in relation to specific actions and water-year type. The latter variation will likely provide more contrast that will help to determine the population-level response to flow.

> **Charge Question 4.** Under the existing network of monitoring programs, what site-specific and system-wide monitoring methods does the Panel recommend using to assess achievement of the biological goals, including each VSP parameter? For example, could rotary screw traps be used to effectively monitor site-specific salmonid productivity? Could adult escapement rates be used to monitor system-wide population abundances?

The Panel did not critique the large number of potential monitoring techniques that are available but did discuss the types of data needed, the level of effort, data quality and some monitoring approaches (Section 4.5.1). Screw traps can be very effective for estimating juvenile production when deployed in suitable conditions and coupled with mark-recapture analyses to determine capture efficiency, although large smolts such as steelhead can avoid traps. The Panel strongly recommends accurate estimates of the spawning population, along with pHOS, because spawning escapement is the basis for constructing the "brood tables" discussed in Section 4.3. Furthermore, if the goal is to evaluate returns of natural-origin adult salmon and life cycle productivity, it is critical to estimate the total abundance of hatchery-origin salmon so that numbers of natural-origin salmon can be estimated. This comment applies to both fishery harvests (which can remove a significant percentage of returning fish) and spawning escapement counts. Adult age data are necessary to assign adults back to the parent-spawning year (brood year). As described in Section 4.5, run reconstruction is needed to create brood tables

(Appendix 9.4) for each population so that productivity and abundance can be estimated.

> **Charge Question 5.** Recovery of native salmon populations may take many years. Can the Panel provide guidance on how to account for that factor in the development of biological goals? Specifically, what VSP trends or other methods would indicate a high probability of population extinction, stability, or recovery? How many years will be required to get meaningful population metrics for assessing progress toward achieving the narrative objectives?

Recovery time will depend on environmental conditions in freshwater and marine habitats, the size and effectiveness of restoration actions, and the ability to adequately monitor viability metrics, especially productivity and abundance (Section 4.5.2). However, given the complex life history structure of Chinook salmon and natural variability, the proposed stock-recruit analysis will likely require 20-30 years of data to provide moderately reliable inferences about the effects of management actions on abundance and productivity (Bradford et al. 2005). Analyses involving the spawner-to-smolt stage require less time because they do not involve variability associated with survival in the estuary, ocean, and fisheries, and confounding adult values associated with large numbers of returning hatchery salmon. Nevertheless, it is critical to consider the entire life cycle when evaluating population extinction risk, stability, and recovery. The framework described by Lindley et al. (2007) for evaluating extinction risk is recommended.

9.2 Answers to Additional Salmon Questions by the Board

During its review of the draft Biological Goals report, the Board asked the Panel to address the following questions in the final report.

1. How were biological goals developed for Chinook salmon in the Puget Sound?

The Shared Strategy for Puget Sound, National Marine Fisheries Service, and co-managers developed recovery goals for natural-origin Chinook salmon in the Puget Sound ESU and planning targets for each watershed. Methods to develop planning targets in each watershed were not described in detail (e.g., PSTRT 2002, SSDC 2007, Ford 2011). Their approach for developing targets used an Ecosystem Diagnosis and Treatment (EDT) analysis of NOAA's "properly functional conditions" (PFC) in a Beverton-Holt stock-recruit framework (Blair et al. 2009) to estimate three salmon metrics for watershed conditions considered to be "properly functioning": 1) equilibrium abundance when spawner abundance is relatively high (R/S = 1); 2) spawner abundance that supports maximum sustained yield (MSY) when the population is productive; and, 3) productivity (R/S) at MSY. In one watershed, the targeted salmon range was considered to be 75-80% of historical abundance, and the watershed plan was to work toward the target over the next 50 years (SBSRF 2005). The goals for diversity and spatial structure are not quantitative (SSDC 2007). The diversity goal is to restore the historical pattern of life history diversity. The spatial structure goal is to protect existing and potential future habitat used by salmon throughout their life history.

The Puget Sound approach differs from the Panel's recommended approach in that their planning targets for future abundances of Chinook salmon stem from professional opinion assumptions within the PFC and EDT framework, such as future habitat conditions and the degree to which salmon productivity and abundance respond to those future conditions (a major uncertainty). Planning targets based on EDT should be considered hypotheses with considerable uncertainty (ISAB 2001). The Puget Sound approach does not allow for direct quantitative evaluation of management action effects, as does the Panel approach. Rather, progress can be predicted from the PFC/EDT model and the underlying assumptions related to how fish respond to changes in habitat. Additionally, the Puget Sound approach does not directly account for changes in ocean conditions and climate. Puget Sound co-managers recognize the need to enumerate hatchery salmon on the spawning grounds and in harvests and improvements have recently been made in monitoring. The planners recognize that recovery will take multiple decades.

Abundance

2. Can the Anadromous Fish Restoration Program (AFRP) doubling targets for fall-run Chinook salmon (Final Restoration Plan for the AFRP (USFWS 2001), Appendix B) or other adult abundance goals for each tributary be used to calculate targets for outmigrant survival?

Yes, they can. A logical alternative is to use estimates of survival between juvenile outmigration until adult return (prior to fishery) and divide the pre-fishery abundance doubling goal by this survival rate, as follows:

Smolt Requirement = (Pre-Fishery Adult Requirement)/(Smolt-adult Return Survival)

Estimation of this survival rate is likely only possible at present for juveniles released from the hatchery that are marked with coded wire tags. The hatchery-based survival rate would likely be lower than that for naturally-produced smolts (assuming hatchery fish released in the natal river rather than lower estuary), leading to a possible upward bias in the juvenile tributary requirements. This exercise would be useful, especially if conducted in tributaries with reliable estimates of juvenile production. The Panel cautions, however, that this approach should also account for numbers and survival of fry migrants. The smolt requirement could then be compared to observed production of natural-origin juveniles to determine whether the requirement can be realistically achieved. This approach was mentioned in Box 4.1 and Section 4.3.1.

3. Is it important to establish abundance targets specific to life stages for biological goals? For example, identifying abundance targets for number and size of redds, juvenile outmigration, and returning adults?

As described in Chapter 4, several targets for the juvenile outmigration stage and the returning adult stage are recommended. Evaluation of juvenile production in relation to goals and in response to management actions in the tributaries will be easier to detect compared with the adult stage, which includes additional variability ("noise") associated with smolt-to-adult survival. Nevertheless, evaluation of adult returns is essential for determining the viability of the population. As noted in Question 2 and in Chapter 4, smolt abundance targets to achieve adult abundance targets can be calculated if smolt-to-adult survival is known. Goals for salmon redd counts could be developed and used instead of the number of spawners in the stock-recruit approach. Redd counts are often translated to female spawner or total spawner counts.

Abundance targets based on redd number and size, as described in the question, presumably refers to the capacity of the watershed to support spawners if spawning areas were known. This approach is not recommended, because the fish are much better than fish biologists at detecting suitable spawning habitat. This is why the Panel recommended empirically-derived metrics from the spawner-recruit relationship.

4. What methods are available to establish numeric abundance goals for other salmonids and native species?

For abundance, numbers of spawning salmon that lead to maximum recruitment of natural-origin juveniles or adults, or spawners that lead to population equilibrium in a viable population are recommended. The Panel also recommends tracking abundance over time to document the level of fish responses (i.e., salmonids, native fishes, and select non-native fishes) to management actions since the basic assumption of most management actions is to increase abundance. Here, the basic goal is a positive population growth trajectory. The panel prefers this simple approach because it does not require highly uncertain assumptions about the level of fish response to management actions—only that the response is positive.

Alternatively, the approach used in Puget Sound (see above) could be used to develop hypotheses about future abundance based on assumed changes in PFC and assumed functional relationships between fish and habitat.

Productivity

5. Can stock-recruitment relationships be developed at this time based on existing monitoring programs and data on the Lower San Joaquin River tributaries?

Yes, please see Appendix 9.5, which uses juvenile and parent spawner data from the Stanislaus River. Additional effort is likely needed to estimate abundance of natural-origin salmon in annual adult runs to the tributaries and natural-origin returns by brood year (both pre-fishery recruits and recruits entering the tributaries).

6. The Ricker stock-recruit (S-R) model has certain advantages for statistical modeling, but also contains over-compensatory density-dependence that covaries with maximum carrying capacity and intrinsic productivity. Can the Panel provide any recent references that evaluate the appropriateness of these assumptions for Chinook salmon? Please comment on appropriate statistical methods to use with a purely compensatory S-R model such as the Beverton-Holt.

There are many examples where Ricker stock-recruit models have been fit to data from Chinook salmon populations. A survey of the literature is beyond the scope of the review, but from our experience there are more cases where a Ricker model has been fit to Chinook salmon data than a Beverton-Holt model or other forms (e.g., Shephard model). However, it is likely that the data are not precise enough in the vast majority of cases to determine which model form is most consistent with the data. Thus, other considerations, like parameter identifiability, lower bias in the productivity estimate, and ease of computation, are the best reasons to select a Ricker model. For the purposes of this review, the critical question is the influence of model form (Ricker or Beverton-Holt) on the covariate effect size. The use of Ricker versus Beverton-Holt models is examined in Stanislaus River example in Appendix 9.5. It isn't likely that selection of model form will have an important effect on inferences with respect to evaluating flow, water temperature, or habitat restoration effects.

7. What are the assumptions that allow the development of S-R methods when data are deficient? Can these assumptions be used to create brood tables that will allow a reasonable estimation of the status and trends of the population on each tributary?

Assumptions may be used to fill-in missing values or to create a brood table for a salmon population. However, assumptions will lead to greater uncertainty in population abundance and productivity.

If age of returning Chinook salmon is missing for a population, average age from representative populations could be assumed (note: hatchery salmon tend to mature at younger ages than natural-origin salmon; additional age monitoring is needed). Years with missing age proportions could be estimated from adjacent years when data are available. Within a population, age-specific returns (abundance) could be estimated from siblings that returned in the previous year using simple regression. pHOS can be estimated as described in Question 18. Population-specific harvests of natural-origin Chinook salmon is typically assumed to be proportional to their relative abundance in spawning escapement throughout the basin when adult return timing is the same. For example, if 10% of the total spawning escapement of natural-origin salmon occurs in tributary X, then typically it can be assumed that 10% of the harvest is from tributary X.

8. Are the population-specific catch estimates such as commercial and recreational ocean exploitation rates (referred to in Section 4.5.2 Data Estimation) used to derive the "U" exploitation variable in equation 3?

Yes. The spatial scale of a productivity analysis like equation 3 requires estimates of catch (exploitation rate) at the same spatial scale. In the absence of this information, it is somewhat common to assume all stocks or tributaries are exposed to the same exploitation rate in "downstream" fisheries. Sampling of DNA or coded wire tags in the fisheries can be used to define stock and tributary-specific exploitation rates. This is done in almost all other major Chinook-bearing river systems in North America.

Data and Estimation

9. Are estimates of returning adult fall-run Chinook salmon for each tributary provided by California Department of Fish and Wildlife in the Grandtab database (http://www.cbr.washington.edu/sacramento/data/query_adult_grandtab.html) sufficient for use as spawning escapement?

The GrandTab database is an excellent compilation of spawner counts in CV tributaries. It is beyond the scope of this effort to evaluate the adequacy of spawner count methodology in each watershed. However, the understanding is that the counts typically represent total counts or nearly so (natural-and hatchery-origin fish in the river) and that fish have been enumerated fairly consistently over time (Bergman et al. 2012).

Improvements to the GrandTab database could be made if spawner counts were shown for each age category, and if natural-origin fish were estimated in addition to the total counts that are presently shown (i.e., hatchery and natural-origin fish that spawn in the river). Appendix 9.4 provides an example “brood table” that should be developed for CV Chinook salmon. The brood table includes pre-fishery recruits as well as spawners in the tributaries.

10. What are the methods for estimating population-specific catch for fall-run Chinook salmon?

Hatchery-specific populations of Chinook salmon in harvests can be estimated from expansion of coded-wire-tags of those fish and fin-clipped fish recovered in the fisheries (see Section 4.5.1; Barnett-Johnson et al. 2007, Kormos et al. 2012, Palmer-Zwahlen et al. 2013, 2018). Population-specific estimation of natural-origin salmon harvested in fisheries requires a run reconstruction approach given that these fish are not marked and not likely identifiable using current genetic sampling in the CV (e.g., USFWS 2001 (Appendix A), English et al. 2007, Mesick et al. 2009). Key to run reconstruction is to estimate abundances of natural-origin Chinook salmon on the spawning grounds of each population (i.e., large tributaries), then apply these proportions to the harvests of natural origin Chinook salmon (after subtracting hatchery fish from total harvests). Fish spawning in unmonitored tributaries should be accounted for, otherwise population-specific harvests will be overestimated. Differences between commercial and sport fisheries should be considered (see Mesick et al. 2009). The reliability of natural-origin abundances in fisheries and on the spawning grounds relies upon the ability to accurately estimate hatchery-origin salmon (Cal-Nev AFS 2009, CA HSRG 2012, Mohr et al. 2017).

11. Can reference streams be used as a short-term solution for tributaries that may not have extensive monitoring or little to no available data?

Yes. For example, simple regression can be used to predict an abundance target in a less monitored watershed from basin size. Parken et al. (2006) describe this approach, which was utilized by Ford (2011) in Puget Sound. However, there can be considerable uncertainty in such predictions. See Section 4.6 for recommendations on abundance targets such as spawner abundance leading to maximum recruitment or population replacement (R/S = 1) and smolt production leading to viable populations.

12. The report states that a minimum of 5 to 10 years (or 15 to 20 years for salmonids) of monitoring data will be required to provide a reasonably reliable evaluation of flow and habitat treatment effects. However, natural resources management strategies would need to see any meaningful progress in a much shorter timeline. What would be the reasonable time periods for such intermediate assessments?

The amount of time required to estimate the effect of a management action on a particular metric (e.g., egg to fry survival rate) depends on: 1) the magnitude of the management action in relation to the targeted life stage (bigger effect takes less time to observe); 2) the amount of measurement error (less error = less time); and, 3) the amount of unexplained variation (process error) in the metric (less error = less time). Unfortunately, the understandable need for decision-makers to rapidly assess costly management actions has no influence on these scientific and statistical realities. The only way to speed up the process is to model the effects of the management action and to focus on life stages that are most influenced by the action. But a reliable inference from such an exercise requires a reliable model. As discussed in the report, current models are not reliable enough except in a few cases (e.g., water temperature effects on incubation survival, migration survival of salmonids through Sacramento and Delta via acoustic tagging model). Appendix 9.5 demonstrates how the stock-recruit covariate analysis can be applied to data in the Stanislaus River. Analyses such as these can inform managers about potential effects of future management actions.

Natural Production and Percent Hatchery Origin Spawners (pHOS)

13. Is pHOS derived from the number of hatchery origin fish that spawn in the river or is it extrapolated from hatchery return rates?

pHOS is the proportion of total spawners in a river that are hatchery-origin fish. It is not extrapolated from returns to the hatchery. pHOS is typically estimated from external fin clips on hatchery fish (approximately 25% of hatchery fall-run Chinook salmon are fin-clipped, leading to potentially inaccurate estimates if few fish are sampled [Cal-Nev AFS 2009]). It could also be estimated from coded-wire-tag extrapolations, otolith analysis of spawners (thermal marks in the hatchery, or potentially from analysis of otolith chemistry), or parentage-based tagging (Mohr et al. 2017, Beacham et al. 2019).

14. Should an impairment variable (f(PNI), equation 10b) of hatchery-origin spawners be included in stock-recruit models for each tributary or only those with hatcheries?

The answer to this question is discussed in Section 4.3.5. pHOS should be estimated in tributaries with and without hatcheries. Hatchery salmon stray to watersheds beyond the natal hatchery watershed, especially when hatchery fish are transported and released in the estuary. Transportation of juvenile salmon can inhibit imprinting, which is necessary to guide salmon back to their natal watershed (or hatchery).

The Panel cautions that use of an assumed hatchery impairment function (e.g., f(PNI)) could bias estimates of the number of effective spawners (S') and therefore intrinsic productivity if the borrowed PNI values are not representative of the target population. For example, overestimation of the adverse effect of low PNI would lead to artificially low S' and estimates of intrinsic productivity that are too high. In other words, this bias could incorrectly lead one to believe the population was viable when in fact it was not. In absence of f(PNI) values for specific populations in the CV, we emphasize that pHOS and PNI be estimated but that modeling use total spawner counts rather than assumed effective spawners.

15. Given the challenges and importance of quantifying the impacts of hatchery-origin spawners on natural production, is there a management approach that is feasible, accurate, and sustainable for estimating a value for the impairment variable while conducting experiments to improve the precision of this variable?

As noted in the answer to Question 14, an incorrect assumption regarding the effect of pHOS or PNI could significantly bias coefficients produced by the stock-recruit analysis. For example, an f(PNI) value that is biased high (strong negative effect) would lead to low effective spawners (S'), which would lead to productivity of the natural population that is too high. Given this complication, this approach to account for a hatchery effect should not be used unless it accurately reflects the target population. High pHOS or low PNI will reduce productivity of the naturally spawning population (Box 4.2), which includes both hatchery and natural-origin spawners. Year-to-year variation in pHOS or PNI will contribute to variable productivity, which will make it more difficult to estimate the potential benefit of management actions. Ideally, pHOS should be minimized and PNI maximized in watersheds. Regardless, hatchery-origin fish must be estimated in both spawning escapement and harvests if status and trends of natural-origin populations is to be documented. See Section 4.3.5.

16. Would the use of a PNI (proportionate natural influence) estimate in the Central Valley apply only to winter-run Chinook salmon tributaries as it is the only run that is occasionally supported with a hatchery broodstock program?

No, a PNI value should be estimated for every population in a watershed that has hatchery production because many hatchery fish spawn in the rivers. It is worthwhile to know the PNI value as an index of the degree to which hatchery fish influence the natural-origin population. PNI values are simple to calculate using pHOS and pNOB. PNI = pNOB/(pNOB + pHOS), where pNOB = proportion of broodstock that is natural origin and pHOS is proportion of spawners in the wild that are hatchery origin.

Spatial Structure and Assessment

17. Would a simple spatial structure biological goal be meeting abundance and productivity targets for each tributary?

Yes. These targets should include population viability in which intrinsic productivity exceeds 1, when estimated using pre-fishery natural-origin recruits and/or recruits that survive to the spawning grounds after harvests. See Section 4.6.

Brood Tables

18. Can estimates of percent hatchery origin spawners (pHOS) cited in the literature be used to generate brood tables and derive production targets while tracking of pHOS improves?

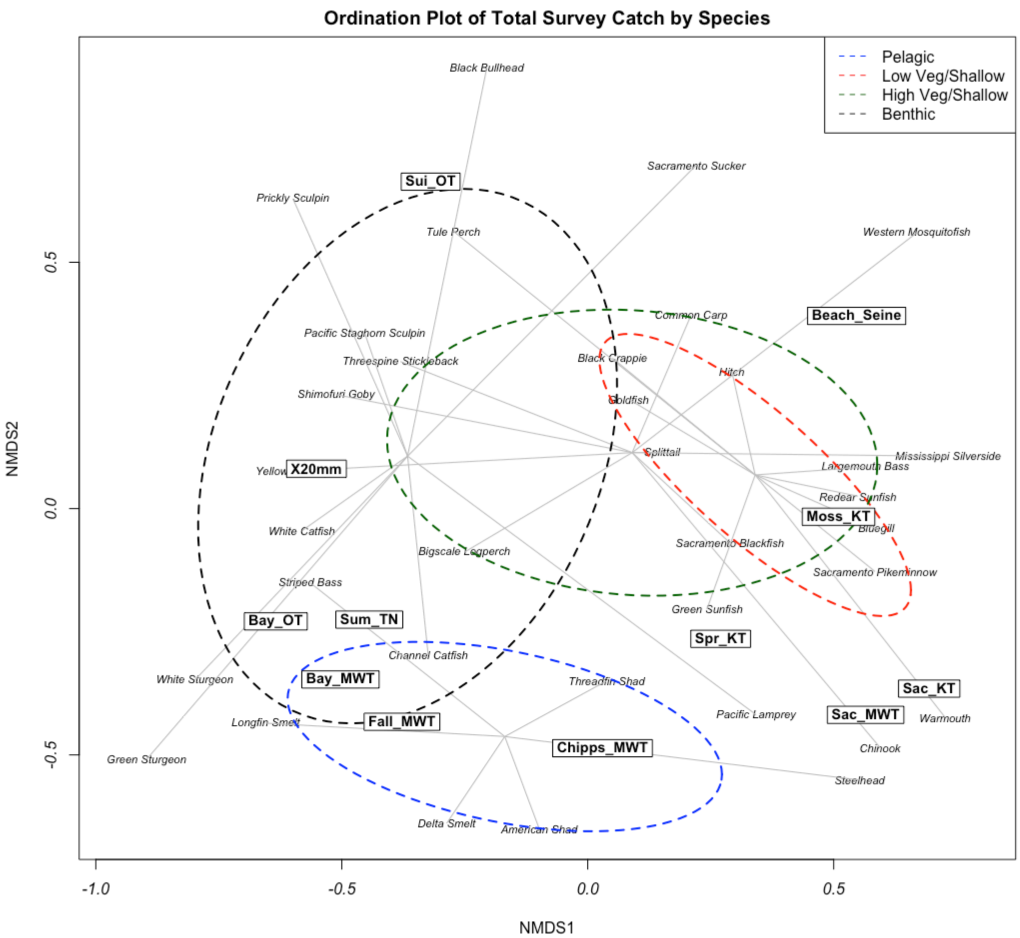
Yes. pHOS data has been estimated in some CV tributaries in recent years, as briefly described in Section 4.5.1. These values could be used to approximate pHOS in earlier years when pHOS was not estimated from tag or mark data. This approach should consider the level of hatchery production and natural production over time. As noted in Section 4.5.1, the AFRP doubling goals attempt to account for hatchery fish on the spawning grounds, but we suspect the values were too low, leading to overestimates of natural-origin production during the baseline period, and hence an overestimate of the target. Escapement counts of Chinook salmon reported in Pacific Fishery Management Council reports and in GrandTab (i.e., in rivers) do not separate hatchery and natural-origin spawners.

Importantly, estimates of hatchery-origin salmon are needed for both harvests and spawning escapement so that adult returns of natural-origin salmon can be estimated. Estimating pHOS in early years from recent year data will lead to greater uncertainty in production values for those earlier years. Therefore, some caution is warranted when using earlier data.

19. Could development of a brood table for one tributary be used to inform brood table development and management decisions on another tributary?

A complete brood table in one tributary might be used to estimate some missing values in another brood table if there is positive correlation in the values. However, it is unlikely that a brood table for one population could be used to make management decisions in another tributary unless there is information explicitly indicting that the two populations are correlated.

9.3 Native Fishes and Fish Assemblages



Appendix Figure 9.3.1. Nonmetric multidimensional scaling (NMDS) plot of total species catch across CDFW Bay Study Otter Trawl (Bay_OT), CDFW Bay Study Midwater Trawl (Bay_MWT), CDFW Fall Midwater Trawl (Fall_MWT), CDFW Summer Townet Survey (Sum_TN), CDFW 20mm Trawl (X20mm), CDFW Spring Kodiak Trawl (Spr_KT), USFWS Beach Seine Survey (Beach_Seine), USFWS Sacramento Midwater Trawl (Sac_MWT), USFWS Sacramento Kodiak Trawl (Sac_KT), USFWS Mossdale Kodiak Trawl (Moss_KT), USFWS Chipps Island Midwater Trawl (Chipps_MWT) and the UC Davis Suisun Marsh Otter Trawl (Sui_OT). Species grouped based on general habitat associations (Pelagic, Low Veg/Shallow, High Veg/Shallow, Benthic) and ordination calculated using aggregate survey catch with Bray-Curtis dissimilarity distance in R package "vegan" (Oksanen et al. 2018; R core team 2014). Analysis by Dylan Stompe, UCD.

Appendix Table 9.3.1. Long Term Surveys Involving Delta Sampling

Survey	Agency	Years of Sampling	Timeframe	Method	Location(s)	Number of Stations	Sampling Intensity	Survey Purpose	Area Sampled
Egg and Larval Survey	CDFW	1967-1977, 1984-1986, 1988-1993 and 1994	February - July	Egg and Larval Trawl	Suisun Bay, Lower Sacramento River, Middle Sacramento River, Lower San Joaquin River	106	Every 2-4 days during sampling period	Striped Bass egg and larval abundance	Oblique - Bottom, Mid, Surface
San Francisco Bay Study	CDFW	1980 - Present	Year Round	Midwater Trawl	South Bay, Central Bay, San Pablo Bay, Suisun Bay, West Delta, Lower Sacramento River, Lower San Joaquin River	35 historic, 17 added between 1988-1994	All stations sampled once per month; 12 complete surveys/year.	To determine effects of freshwater outflow on abundance and distribution of fish and mobile crustaceans	Mid - Oblique retrieval
San Francisco Bay Study	CDFW	1980 - Present	Year Round	Otter Trawl	South Bay, Central Bay, San Pablo Bay, Suisun Bay, West Delta, Lower Sacramento River, Lower San Joaquin River	35 historic, 17 added between 1988-1994	All stations sampled once per month; 12 complete surveys/year.	To determine effects of freshwater outflow on abundance and distribution of fish and mobile crustaceans	Bottom
Fall Midwater Trawl	CDFW	1967 - Present (less 1974 and 1979)	September - December	Midwater Trawl	San Pablo Bay, Napa River, Suisun Bay, Delta, Lower Sacramento River, Lower San Joaquin, Deepwater Ship Channel	100 historic, 22 added between 1990-2010	All stations sampled once per month; generally 9 days to sample all stations. Four complete surveys/year.	Age-0 Striped Bass, Delta Smelt, American Shad, Longfin Smelt, Splittail, and Threadfin Shad Abundance	Mid
Summer Townet	CDFW	1959 - Present	June - August	Tow Net	Napa River, Suisun Bay and Sloughs, Delta, Lower Sacramento River	32 historic, 8 added in 2011 for Delta Smelt	All stations sampled 2-5 times/yr historically, standardized to 6/yr in 2003	Age-0 Striped Bass and Delta Smelt Abundance	Bottom

Survey	Agency	Years of Sampling	Timeframe	Method	Location(s)	Number of Stations	Sampling Intensity	Survey Purpose	Area Sampled
20mm Survey	CDFW	1995 - Present	April - July	Egg and Larval Trawl	San Pablo Bay, Napa River, Suisun Bay, Delta, Lower San Joaquin, Deepwater Ship Channel	54	8-10 complete surveys/year, conducted fortnightly	Postlarval-juvenile Delta Smelt distribution and abundance	Oblique - Bottom, Mid, Surface
Spring Kodiak Trawl	CDFW	2002 - Present	January - May	Kodiak Trawl	Napa River, Suisun Bay, Delta, Lower Sacramento River, Lower San Joaquin River	40	All stations sampled once per month; generally 4-5 days to sample all stations. Five complete surveys/year.	Abundance and distribution of spawning Delta Smelt	Surface
Beach Seine Survey	USFWS	1976 - Present	Year Round. Three sites on Sacramento River only sampled October - January	50ft Beach Seine	Central Bay, San Pablo Bay, Delta, Lower Sacramento River, Middle Sacramento River, Lower San Joaquin River	58	0.5-3 days per week depending on station. Majority of sites sampled once per week	Juvenile Salmon and other resident fishes monitoring	Beach
Chipps Island Trawl	USFWS	1976 - Present	Year Round	Midwater Trawl	Suisun Bay	1	Three times per week	Juvenile Salmon abundance monitoring	Surface
Mossdale Trawl	USFWS	1994 - Present	Year Round	Kodiak Trawl	Lower San Joaquin River	1	Three times per week	Juvenile Salmon abundance monitoring	Surface
Sacramento Trawl	USFWS	1988 - Present	April - September	Midwater Trawl	Lower Sacramento River	1	2-3 times per week	Juvenile Salmon abundance monitoring	Surface
Sacramento Trawl	USFWS	1994 - Present	October - March	Kodiak Trawl	Lower Sacramento River	1	Three times per week	Juvenile Salmon abundance monitoring	Surface
Resident Fish Surveys	CDFW, UCD	Sporadic - *See Footnote	Varied	Boat E-Fishing	Legal Delta	Varied	Varied	Resident fish abundance monitoring	Shallow/Edge Habitat
Suisun Marsh Fish Study	UC Davis	1979 - Present	Year Round	Beach Seine	Suisun Marsh	3	All stations sampled once per month	Resident fish abundance monitoring	Beach

Suisun Marsh Fish Study	UC Davis	1979 - Present	Year Round	Otter Trawl	Suisun Marsh	21	All stations sampled once per month	Resident fish abundance monitoring	Bottom
Yolo Bypass Fyke Net	DWR	1998 - Present	October - June	Fyke Trap	Yolo Bypass Toe Drain	1	Trap fished continuously	Adult fish abundance monitoring	Bank
Yolo Bypass Beach Seine	DWR	1998 - Present	Year Round	Beach Seine	Yolo Bypass - Perennial Pond, Toe Drain, Floodplain	14	Biweekly	Juvenile fish abundance monitoring	Beach

*Surveys historically maintained over 1-2 year periods beginning in 1980s(?). Systematic compilation not needed.

Appendix Table 9.3.2. Status of Common Delta fishes. Columns explained at bottom of table.

Species	Native	Status (1-5)	Pop. Trends (1-4)	Climate Change (-2 - +2)	Indicator Value (1-5)	South Delta
Pacific lamprey, *Entosphenus tridentatus*	Yes	SSC (3.3)	2	-2	3	RA (MI)
White sturgeon, *Acipenser transmontanus*	Yes	SSC (2.6)	2	-1	5	RA
Green sturgeon, *Acipenser medirostris*	Yes	Listed (1.6)	1 or 2	0	4	RA
American shad, *Alosa sapidissimus*	No 50+	None	2	-1	4	MI
Threadfin shad, *Dorosoma cepidianus*	No 50+	None	2-3	0	1	AB
Common Carp, *Cyprinus carpio*	No 50+	None	3	+2	1	AB
Goldfish, *Carassius auratus*	No 50+	None	3	+2	1	CO
Sacramento Hitch, *Lavinia exilicauda*	Yes	SSC (3.1)	1	0	2?	RA
Sacramento blackfish, *Orthodon microlepidotus*	Yes	None (4.4)	1	+1	1	LO?
Sacramento pikeminnow, *Ptychocheilus grandis*	Yes	None (4.7)	3	0	2	LO
Splittail, *Pogonichthys macrolepidotus*	Yes	SSC (3.1)	3	-1	5	MI
Sacramento Sucker, *Catostomus occidentalis*	Yes	None (5.0)	3	0	2	LO
Black bullhead, *Ameiurus melas*	No 50+	None	4	+2	1	AB
Channel catfish, *Ictalurus punctatus*	No 50+	None	3	0	1	CO
White catfish, *Ameiurus catus*	No 50+	None	3	+1	3	AB
Delta smelt, *Hypomesus transpacificus*	Yes	Listed (1.4)	1	-2	5	RA (MI)
Longfin smelt, *Spirinchus thaleichthys*	Yes	Listed (2.0)	1	-2	5	RA (MI)
Steelhead/rainbow trout, *Oncorhynchus mykiss*	Yes	Listed	1	-2?	2	MI

Chinook salmon (fall run), *Oncorhynchus tshawytscha*	Yes	SSC (3.4)	2	-1	4	MI
Mississippi silverside, *Menidia audens*	No	None	4	+1	3	AB
Western Mosquitofish, *Gambusia affinis*	No 50+	None	3	+2	1	AB
Threespine stickleback, *Gasterosteus aculeatus*	Yes	None (4.1)	3	-1	3	LO?
Prickly sculpin, *Cottus asper*	Yes	None (4.7)	3	+1	3	CO
Staghorn sculpin, *Leptocottus armatus*	Yes	None (no score)	3	+2	3	RA
Striped bass, *Morone saxatilis*	No 50+	None	2	0	5	CO
Black crappie, *Pomoxis nigromaculatus*	No 50+	None	3	0	1	AB
Bluegill, *Lepomis macrochirus*	No 50+	None	3	+1	1	AB
Redear sunfish, *Lepomis microlophus*	No 50+	None	3	+1	1	AB
Green sunfish, *Lepomis cyanellus*	No 50+	None	3	+2	1	CO
Largemouth bass, *Micropterus salmoides*	No 50+	None	4?	+1	1	AB
Spotted bass, *Micropterus punctulatus*	No 50+	None	3	+1	1	CO
Warmouth, *Lepomis gulosus*	No 50+	None	3	+1	1	CO
Bigscale logperch, *Percina macrolepidotus*	No 50+	None	3	0	1	AB
Tule perch, *Hysterocarpus traski*	Yes	None (4.0)	2	-1	3	CO?
Shimofuri goby, *Tridentiger bifasciatus*	No 50+	None	4	+1	3	AB
Yellowfin goby, *Acanthogobius flavimanus*	No 50+	None	3	0	3	RA

Explanation of columns
1. Names

2. Native vs non-native?
   a. Yes, native
   b. No, non-native for <50 years
   c. No, Non-native for more than 50 years (naturalized)

3. Official status: State, federal T&E, state Special Concern
The number in parentheses is status score assigned by Moyle et al. 2015 (updated from Moyle et al. 2011), based on six metrics. Scores range from 0.0 (extinct) to 5.0 (abundant & widespread).

4. Current population trends (based on Moyle et al. 2011, 2015, 2017)
   1. Rare, decreasing
   2. Abundant/common, decreasing
   3. Abundant/common, not increasing or decreasing
   4. Abundant/common, increasing

5. Sensitivity to climate change in next 100 years or less (from Moyle et al. 2013).
   -2. Extinction or major decline
   -1. Decline
   0. No change
   +1 Increase
   +2 Major expansion of abundance

6. Potential value as indicator species for estuarine "health" of Delta, where a healthy Delta is one that functions as an estuary with strong gradients in salinity and temperature, seasonable variability in outflows, etc. that create an environmental favorable to euryhaline fishes (our assessment).
   1. Very low, if resident, more abundant outside SFE than in, indicative of fresh, warm water or riverine conditions
   2. Low, if resident, more abundant outside SFE than in, indicative of fresh, cool water conditions
   3. Moderate, euryhaline species but as abundant outside SFE as in.
   4. High, one or more life history stages depends on Delta but populations heavily influenced by outside conditions (flow releases. Ocean conditions, etc.)
   5. Very High. Species depends on functioning estuary (salinity, temperature gradients etc.) for persistence in California

7. Status in south and central Delta

RA. Rare/absent in most years

LO. Present in low numbers

CO. Common

AB. Abundant

MI. Migratory (passing through or drawn in by pumping plants)

Appendix Table 9.3.3. Habitat and life history characteristics of common Delta fishes. Scientific names can be found Table 1. Explanation of columns is provided below table.

Species	Salinity	Temp	Adult size	Life span	Spawn	Migration	Habitat	Know-ledge Delta
Pacific lamprey	EAN	Cool	M	3	River	AN	Ben (l) Pel (ad)	Low
White sturgeon	EAN	Cool	L	5	River	AN	Benthic	Mod
Green sturgeon	EAN	Cold	L	5	River	AN	Benthic	Mod
American shad	EAN	Cool	M	3	River	AN	Pelagic	Mod
Threadfin shad	EFW	Warm	S	1	Delta	none	Pelagic	Mod
Carp	EFW	Eury	L	4	Floodpl ain	Within Delta,	Benthic	Low
Goldfish	SFW	Warm	M	3	Delta FW	None?	Veg	Low
Hitch	SFW	Cool	M	3	Delta	Up-river	Pelagic	Low
Sacramento blackfish	SSW	Warm	M	3	Delta	None?	Pelagic	Low
Sacramento pikeminnow	SFW	Cool	L	4	River	Up-river	Pel (ad) Edg (j)	Mod
Splittail	EFW	Cool	M	3	Floodpl ain	Up-river	Benthic	High
Sacramento sucker	EFW	Eury	L	4	River	Up-river	Benthic	Mod
Black bullhead	SFW	Warm	M	3	Delta	none	Ben +veg	Low
Channel catfish	EFW	Eury	L	4	Delta	none	Benthic	Low
White catfish	EFW	Eury	M	3	Delta	None?	Ben+veg	Mod
Delta smelt	EFW	Cool	S	1	Delta	Within Delta	Pelagic	High
Longfin smelt	EAN	Cool	S	2	Delta	AN	Pelagic	Mod
Chinook salmon	EAN	Cold	L	2	River	AN	Pelagic Edg(J)	High
Steelhead	EAN	Cold	L	3	River	AN	Pelagic	High
Mississippi silverside	EFW	Eury	S	1	Delta	none	Edge	Mod

Mosquitofish	EFW	Warm	S	1	Delta	none	Edge Veg	High
Threespine stickleback	EFW/MA	Cool	S	1	Delta	none	Edge Veg	Low
Prickly sculpin	EFW	Eury	S	2	Delta	None?	Benthic	Mod
Staghorn sculpin	EMA	Cool	M	2	SF Bay	To SF Bay	Benthic	Low
Striped bass	EAN	Eury	L	4	River	River	Pelagic	High
Black crappie	SFW	Warm	M	2	Delta	None	Veg, Edge	Low
Bluegill	SFW	Warm	M	2	Delta	None	Veg	Low
Redear sunfish	SFW	Warm	M	2	Delta	None	Veg	Low
Green sunfish	SFW	Warm	M	2	Delta	None	Edge Veg	Low
Largemouth bass	SFW	Warm	L	3	Delta	None	Veg	Mod
Spotted bass	SFW	Warm	L	3	Delta?	None	Veg	Low
Warmouth	SFW	Eury	M	3	Delta	None	Edge	Low
Bigscale logperch	SFW	Warm	S	3	Delta	None	Benthic, Veg	Low
Tule perch	EFW	Cool	M	3	Delta	None	Edge, Veg	High
Shimofuri goby	EFW	Eury	S	1	Delta	None	Benthic	High
Yellowfin goby	EMA	Cool	M	2	SF Bay	To SF Bay	Benthic, Edg	Mod

<u>Explanation of columns</u>

1. Names
2. Salinity tolerance (From Moyle et al. 2012)

    Euryhaline Anadromous (EAN), stenohaline freshwater (SFW), euryhaline freshwater (EFW), stenohaline marine (SMA), euryhaline marine (EMA)

3. Preferred/critical temperatures for adult life stages (generalized, lots of overlap)
    1. Cold (18-20C). Salmonids
    2. Cool (20-25C] Most native fishes
    3. Warm (25-35C)
    4. Eurythermal (do not fit categories well)
4. Adult size (maturity)
    1. small (<10 cm)
    2. medium (10-50 cm)

3. large (50+ cm)

5. Life span (potential in wild)
    1. 1-2 years
    2. 2-5 years
    3. 5-10 years
    4. 10-50 years
    5. 50+ years

6. Spawn: principal spawning location
    1. DR Delta resident (in situ spawning)
    2. DM Delta migrant (seasonal migrations)
    3. RI Sacramento River and tributaries
    4. SFB San Francisco Bay

7. Migration for spawning?
    1. No
    2. Within Delta
    3. Up Sacramento River and tributaries
    4. To SF Bay (post juvenile rearing)
    5. Anadromous

8. Broad habitat in Delta (where most likely to be encountered)

    Pel = pelagic (open water column)

    Ben = benthic (associated with channel bottom)

    Veg = associated with aquatic macrophytes

    Edg = edge habitat, often rip-rapped

9. Knowledge of life history in Delta (based on Moyle 2002+ recent papers)

    Low –life history studies lacking

    Moderate – some studies available, basic life history understood

    High – Well studied species, with published life history studies

9.4 Salmon and Steelhead (example brood table)

Example of a brood table constructed from adult Kvichak River sockeye salmon in Bristol Bay Alaska, 1963-2017. Age 1.2 refers to 1 winter in fresh water and two winters at sea before returning to spawn. Returning progeny includes catch plus escapement from the fishery (counted from towers), i.e., pre-fishery recruits. Minor age groups not shown, but they are included in total recruits (progeny). No hatchery salmon in Bristol Bay. Data source: G. Buck, ADFG, pers. comm.

		Number of returning progeny by age							
Brood Year	Parent Spawners	Age 1.2	Age 1.3	Age 1.4	Age 2.1	Age 2.2	Age 2.3	Total progeny	Return per spawner
1963	338,760	31,231	80,023	0	113	925,528	336,813	1,388,216	4.1
1964	957,120	2,288,423	288,371	3,941	111,882	2,613,881	446,099	5,763,515	6.0
1965	24,325,926	10,321,864	299,793	0	485,629	33,528,681	1,159,447	45,820,689	1.9
1966	3,755,185	524,322	885,878	0	16,342	4,568,771	517,436	6,522,062	1.7
1967	3,216,208	342,819	321,956	65	2,299	991,239	109,957	1,784,048	0.6
1968	2,557,440	300,687	38,561	4,514	0	104,011	176,512	635,324	0.2
1969	8,394,204	149,287	321,597	0	6,124	4,476,992	546,728	5,513,626	0.7
1970	13,935,306	45,471	40,280	0	28,962	15,247,973	0	15,363,872	1.1
1971	2,387,392	321,013	653,575	0	45,182	919,844	95,529	2,036,285	0.9
1972	1,009,962	1,971,913	142,305	10,886	0	886,252	232,955	3,248,671	3.2
1973	226,554	531,509	1,048,800	0	2,634	252,857	351,748	2,203,241	9.7
1974	4,433,844	6,047,999	1,845,015	0	319,232	16,927,229	616,516	25,784,407	5.8
1975	13,140,450	5,468,931	1,305,049	0	314,282	29,958,382	382,348	37,439,011	2.8
1976	1,965,282	5,940,228	675,758	3,355	37,502	3,806,686	230,826	10,716,323	5.5
1977	1,341,144	2,024,758	718,010	0	2,130	174,744	121,674	3,089,502	2.3
1978	4,149,288	1,613,301	1,063,136	0	15,273	1,372,783	987,655	5,055,228	1.2
1979	11,218,434	18,868,217	2,470,065	0	69,687	17,920,528	3,664,906	43,049,770	3.8
1980	22,505,268	2,539,067	1,385,037	2,963	13,603	8,291,131	364,137	12,597,313	0.6
1981	1,754,358	745,205	188,998	0	0	962,185	147,140	2,048,789	1.2
1982	1,134,840	492,725	385,823	4,844	531	514,201	111,122	1,509,246	1.3
1983	3,569,982	9,267,005	2,995,170	4,092	2,976	1,111,077	386,132	13,775,451	3.9
1984	10,490,670	2,578,693	1,438,443	0	46,572	17,559,242	1,663,051	23,287,185	2.2
1985	7,211,046	1,051,305	959,016	3,134	36,966	14,851,621	1,382,907	18,314,833	2.5
1986	1,179,322	652,917	868,159	22,540	0	1,539,424	1,007,436	4,114,460	3.5
1987	6,065,880	4,715,392	2,193,831	3,168	32,612	4,276,086	329,082	11,648,130	1.9
1988	4,065,216	3,035,792	1,958,434	1,920	17,858	3,698,337	453,907	9,205,714	2.3
1989	8,317,500	1,860,644	1,072,383	0	146,493	18,335,389	3,276,621	24,800,933	3.0
1990	6,970,020	1,635,680	890,767	0	82,363	22,046,414	1,626,784	26,298,686	3.8
1991	4,222,788	2,192,435	1,181,693	12,027	1,940	1,008,516	236,952	4,637,250	1.1
1992	4,725,864	651,583	300,635	0	2,138	751,845	162,224	1,875,603	0.4
1993	4,025,166	1,087,088	873,116	4,918	678	683,919	477,949	3,130,470	0.8
1994	8,355,936	2,023,631	1,062,072	94	47,953	3,920,261	247,105	7,303,050	0.9
1995	10,038,720	7,737,952	2,098,056	7,703	0	677,133	96,802	10,636,782	1.1
1996	1,450,578	547,556	1,651,818	9,155	0	24,302	27,656	2,260,607	1.6
1997	1,503,732	159,365	140,516	576	15	342,017	173,309	816,242	0.5
1998	2,296,074	375,942	422,187	18,296	638	343,819	93,558	1,254,499	0.5
1999	6,196,914	1,010,493	278,782	12,362	51,079	5,815,772	208,249	7,378,782	1.2
2000	1,827,780	1,884,652	1,264,567	1,992	0	742,843	367,259	4,261,658	2.3
2001	1,095,348	633,259	2,051,098	12,834	1,450	819,689	901,131	4,421,265	4.0
2002	703,884	2,456,932	1,265,247	623	2,386	142,426	10,246	3,881,251	5.5
2003	1,686,804	3,595,854	1,186,181	9,277	0	31,390	129,764	4,966,281	2.9
2004	5,500,134	4,797,865	2,931,164	0	0	2,634,426	554,819	10,918,274	2.0
2005	2,320,332	1,254,243	2,033,447	8,367	0	4,527,248	1,754,061	9,582,839	4.1
2006	3,068,226	3,663,815	2,701,997	1,878	0	1,197,115	746,641	8,319,191	2.7
2007	2,810,208	1,542,520	1,852,364	5,955	24,305	6,972,782	2,379,818	12,795,126	4.6
2008	2,757,912	2,679,158	1,930,847	30,142	2,520	1,247,528	679,005	6,577,118	2.4
2009	2,266,140	740,947	1,001,605	5,284	11,611	9,725,832	1,396,254	12,889,440	5.7
2010	4,207,410	6,053,034	5,545,200	0	87,386	13,231,078	679,369	NA	NA
2011	2,264,352	2,846,209	1,768,634	0	43,953	2,289,956	525,629	NA	NA
2012	4,164,444	7,924,673	2,820,675	NA	3,639	423,296	NA	NA	NA
2013	2,088,576	4,001,282	NA	NA	2,200	NA	NA	NA	NA
2014	4,458,540	NA	NA	NA	NA	NA	NA	NA	NA
2015	7,341,612	NA	NA	NA	NA	NA	NA	NA	NA
2016	4,462,728	NA	NA	NA	NA	NA	NA	NA	NA
2017	3,163,404	NA	NA	NA	NA	NA	NA	NA	NA

9.5 Example Application of the Salmon Stock-Recruit Covariate Approach to Spawner and Juvenile Data from the Stanislaus River

To provide an illustrative example of how the stock-recruit framework described in Chapter 4 can be used to help formulate biological goals for Chinook salmon and steelhead, the approach is applied to Chinook salmon data from the Stanislaus River. In Chapter 4, the Panel recommended that the primary biological goal for Chinook salmon and steelhead should be to increase their abundance. Abundance will be determined by productivity, which in turn depends on previous spawner abundances, flow, and other environmental factors. The stock-recruit framework is a method to quantify the influence of various factors (covariates) that drive productivity, which in turn determines abundance. Covariates included in the model can be managed or partly-manageable quantities like flow, reservoir storage, and temperature, but can also include quantities that may be very difficult or impossible to manage in some circumstances, like large woody debris, channel complexity, or predator abundance.

Annual estimates of the abundance of juvenile (fry and parr+smolt stages) fall-run Chinook salmon in the Stanislaus River migrating past a rotary screw trap (RST) near Oakdale, CA are available for brood years 1997 to 2017 (Pilger et al. 2019). The number of total spawners and female spawners upstream of the Oakdale trap are available for this period, based on a combination of carcass surveys (CDFW 1997-2002) and weir counts (FISHBIO 2003-2014)[12].

The following Ricker model was fit to these data,

$$R_t = S_t * \exp(\alpha - \beta * S_t + \gamma * F_t)$$

where R_t are annual estimates of the total number of parr and smolts from brood year *t* migrating past the RST, and S_t are annual estimates of female spawners. F_t are year-specific covariate values that index flow, water temperature, or a derived variable (like weighted useable area) at some point during the spawning, incubation, and juvenile tributary-rearing and -dispersal periods in each year, and γ represents the coefficient for the covariate effect. Covariate values each year were converted to z-scores so that the average of F values across years was zero. Thus, α represents the log of productivity at the average covariate value (because γ*F=0), and $\gamma * F_t$ is the additive effect of F on log productivity for each year. β represents the magnitude of density-dependent effects.

[12] Juvenile, escapement, discharge, and water temperature data were provided by Andrea Fuller, FISHBIO.

The analysis was repeated using the following version of the Beverton-Holt stock-recruit function,

$R_t = \alpha * S/(1 + (\alpha/\beta)*S) * \exp(\gamma * F_t)$

where α is maximum productivity and β is carrying capacity. Note the $\exp(\gamma * F_t)$ term is a multiplier with a value that will always be greater than zero. This multiplier will result in an increase in the base curve if its value is greater than 1, and a decrease in the curve if the value is less than 1.

Posterior distributions of model parameters (Greek letters in equations) for both Ricker and Beverton-Holt versions were estimated in a Bayesian framework using the WinBUGS software. Bayesian estimation or non-linear search is required to estimate parameters for the Beverton-Holt model and provides a better way to describe uncertainty in predictions for both models. For brevity, the results from the Ricker model are the focus, but they are similar to those from the Beverton-Holt model.

A range of flow covariate statistics based on discharges below Goodwin Dam (USGS gauge 11302000 near Knights Ferry, CA) were computed to evaluate alternate hypotheses about how flow could affect survival rates for different life history stages. Average flows between Oct 1 - Dec 31, Oct 1 – Mar 31, Jan 1 - Mar 31, Feb 1 - Mar 31, and Apr 1 – May 31 were computed for each year to represent conditions during spawning, incubation, emergence, fry rearing and dispersal, and parr rearing and dispersal, respectively. Maximum daily water temperatures from the same gauge were averaged between Oct 1 and Nov 30 each year to represent conditions during the early part of the spawning and incubation period when water temperature can be elevated in some years. These date ranges were largely based on the periodicity chart in Aceituno (1993) with slight modifications based on input from a biologist that works on the Stanislaus River (A. Fuller, FISHBIO, pers. comm.).

Covariate models using flow during the incubation, fry, and parr periods had equivalent fits and predictive reliability (Table 9.5.1a). They explained between 47% to 49% of the interannual variation in the log of juvenile (parr and smolt) production and had very similar Deviance Information Score (DIC) values. DIC represents the reliability of each model (its out-of-sample predictive power). This metric is the Bayesian equivalent of the familiar Akaike Information Criteria (AIC) and identifies the model that strikes the best balance between fit and complexity (number of parameters). The mean of the covariate effect (γ) was positive (more water = more juvenile production), and there was a high probability that the effect was positive (> 96%). Doubling the mean flows resulted in a 1.2- to 1.6-fold increase in juvenile production (2x column in Table 9.5.1a). Results for the emergence period (Jan 1 – Mar 31) flow model are shown in Figure 9.5.1. Predictions of a positive flow effect for this model were largely determined by higher than expected juvenile production in brood years 1997, 1998, and 2005, when Jan-Mar flows affecting those broods were well above the average (see lower-left panel). The female spawner-juvenile production relationships at the across-year minimum and maximum flow levels during the emergence period were substantially different (Figure 9.5.1 top-right panel) because the range between minimum and maximum flows was very large (lower-left panel).

The salmon emergence flow stock-recruit model shown here (Fig. 9.5.1) provides an example of how covariate stock-recruit models can be used to determine the contribution of a particular management action (e.g., flow), or natural variation in the covariate value, to changes in abundance. For example, this model can be used to predict a juvenile production relationship based on a doubling of flow (relative to the mean) during the salmon emergence period. This new curve predicts a 1.6-fold increase in juvenile production due to the doubling in flow (green line in Fig. 9.5.1). Decision-makers can use these types of relationships in three ways. First, in cases where no baseline data are available and a flow increase or other management action is implemented, they can use them in a post-hoc analysis to estimate the extent to which the action increased biological goals of productivity and abundance. Second, in cases where the data already exist (like the Stanislaus River), they can use the model to determine the covariate value (e.g., flow during the emergence period) needed to meet a specific biological goal (e.g., a doubling in juvenile production relative to current levels). Third, they can use the model to define a biological goal (e.g., "x" juvenile Chinook salmon from the Stanislaus River at Oakdale) based on anticipated flow or other covariate levels that can be achieved via a combination of management actions and natural variation. This model could be run under most likely, optimistic, and pessimistic projections of flow and temperature. This latter option is appealing because the biological goal is estimated from data in the system of interest and can incorporate model-based predictions of future flows and temperatures. This may be a better approach for setting goals compared to using less certain hypothesis about juvenile production that are determined, for example, by applying maximum survival rates estimated in other systems.

The stock-recruit framework can also be used to quantify how covariates that can be difficult to manage, such as water temperature in drought years, will influence productivity and the ultimate biological goal of increasing abundance. In the Stanislaus River, average water temperature during the early part of the spawning and incubation period (Oct 1 – Nov 30) explained 72% of the across-year variation in the log of estimated larger juvenile (parr and smolt) abundance, which was considerably higher than in the model that did not include a covariate (39%; Table 9.5.1a). The mean of estimates of the covariate effect (γ) from the posterior distribution was negative, logically predicting that higher water temperatures during the early part of spawning and incubation result in lower juvenile production the following spring. The probability that the covariate effect was greater than zero (i.e., a beneficial effect of higher temperatures) was zero. The water temperature covariate model had a DIC value that was ~13 units lower than the next best model, which indicates it is more reliable than ones based on the flow covariates we examined. The fit and predictions from the early spawning and incubation water temperature model are shown in Figure 9.5.2. The upper-left panel shows that it correctly predicts low juvenile production for 2014-2016 brood years that may have been caused by elevated water temperatures at the end of an extended drought when cold water supplies in New Melones Reservoir were exhausted (note red lines pointing downwards towards the data points).

The preliminary result that the water temperature model was more reliable than flow models for predicting juvenile Chinook salmon abundance in the Stanislaus River in no way implies that increasing flow will not be a helpful action to meet biological goals. The example simply indicates that it may be difficult to increase juvenile production through higher flows in years when water temperature during the spawning and early incubation period is elevated. Thus, understanding the effects of confounding factors like water temperature is fundamental to properly interpreting temporal changes in juvenile abundance and evaluating flow-based management actions. For example, if flows had been purposefully increased in 2014-2016 and monitoring showed lower juvenile production in those years, decision-makers might incorrectly conclude that increasing flows will not help meet juvenile or spawner abundance goals in the long-term. But, by understanding the limiting effect of water temperature in these drought years through the stock-recruit covariate model, decision-makers might maintain the higher flow management action under the assumption that the unusually high water temperatures observed between 2014 and 2016 are unlikely to be the norm in the long-term.

These example conclusions hold when a Beverton-Holt model is used instead of a Ricker model (Table 9.5.1b). The Beverton-Holt model predicts a bigger positive effect of spawning stock size on resulting juvenile production (i.e., less density-dependence) and a slightly larger water temperature covariate effect size (Figure 9.5.3) compared to the Ricker model. This occurs because the mean stock-recruit curve continues to rise with increasing numbers of female spawners, and two of the higher water temperature data points occurred in years with larger numbers of spawners. Thus, the model needs to estimate a more negative (bigger) water temperature effect to fit these data points. However, there is considerable overlap in the credible intervals for the covariate effects based on Ricker and Beverton-Holt models (Table 9.5.2), leading to the conclusion that the difference in the covariate effect size among model forms is not substantial. A similar pattern occurs for flow covariates, which have slightly larger coefficients under the Beverton-Holt model but considerable overlap with predictions from the

Ricker model. Parameters from Beverton-Holt models are typically harder to estimate than for Ricker models, largely because a range of α values can fit the data equally well. This was the case in its application to the Stanislaus River data, requiring use of a minimally-informative prior[13] on the productivity term to produce reliable results. The sensitivity of covariate effect size to this prior assumption needs to be investigated, but this is beyond the scope of this exercise. No such sensitivity analysis is required for the Ricker model, and its parameters can be estimated using a much simpler linear regression approach.

As mentioned in Chapter 4, more complex models can be used to define alternative covariates to evaluate in the stock-recruit framework to gain a better understanding of factors influencing the biological goals of increasing abundance and productivity. As an example, annual covariates based on weighted useable area-flow relationships from the Stanislaus River were computed. Aceituno (1993) predicted weighted useable area (WUA) for spawning, incubation, fry, and juvenile rearing stages as a function of flow using the PHABSIM model (Figure 9.5.4). He predicted that WUA reaches maximum values at 25 cfs for fry, at about 200 cfs for incubation and parr, and at about 300 cfs for spawning. These relationships were used to calculate daily WUA values from mean daily discharge, and then averaged these WUA predictions over the spawning, incubation, fry, and parr periods defined above. The Ricker model was then fit using these annual WUA covariates statistics.

The spawning WUA covariate added little explanatory power to the model as it had the same fit statistic (r^2) as the model without a covariate and had a mean effect size near zero (Table 9.5.2). Interestingly, the mean effect sizes for all other WUA models were negative and had a very low probability of being greater than zero. This indicates that higher values of WUA result in lower juvenile production. This occurs because higher values of WUA for these life stages occur at lower flows (Figure 9.5.4), but the juvenile data indicates that higher flows lead to higher juvenile production (note positive γ values for flow-based covariates in Table 9.5.2).

Preliminary results presented in this appendix highlight how the stock-recruit covariate framework can be used to understand the influence of managed- and naturally-varying covariates on salmonid productivity and ultimately abundance. This information can inform development of biological goals. This preliminary analysis could be extended by: 1) considering alternate time windows to compute covariate values that reflect updated information on life

[13] In Bayesian statistics, a prior probability distribution (often called the prior) for an estimated parameter (like □ in this example), expresses one's beliefs about its values before the data from the current application (e.g., Stanislaus spawner-juvenile production data) is taken into account. Uninformative priors express vague knowledge about parameter values. A minimally informative prior is the least informative prior needed to allow reliable estimation of the parameter when the model is applied to the data.

history timing or management windows of interest (e.g., Feb-June); 2) including multiple but uncorrelated covariates in the same model (e.g. water temperature during fall and flow the following spring); 3) using existing flow-habitat relationships to compute additional covariate statistics (e.g., flow-inundation relationships as defined in Chapter 19 of SWRCB 2018); and 4) using existing flow routing and temperature models to make predictions of juvenile production responses to alternate flow and temperature management regimes.

Table 9.5.1. Preliminary results of applying the Ricker (a) and Beverton-Holt (b) stock-recruit covariate models to data from the Stanislaus River. The tables show the estimated covariate (γ) effect (mean and lower (LCL) and upper (UCL) 95% credible intervals), the fit of the model (Pearson r^2 statistic), and the difference between each models Deviance Information Score (DIC) relative to the most reliable model that has the lowest DIC score (ΔDIC). "p>0" is the probability that the covariate effect (γ) is greater than zero as determined by the posterior samples of γ. "2x" is the relative increase in juvenile production based on a doubling of mean flow covariate values (e.g., a value of 1.5 represents a 1.5-fold increase (50% increase) in juvenile production, 1.0 indicates no change).

a) Ricker Model

	γ (covariate effect)						
Covariate	**mean**	**LCL**	**UCL**	**p>0**	**2x**	**r^2**	**ΔDIC**
No covariate	NA	NA	NA	NA	NA	0.39	14.7
Spawning (Oct_Dec)	0.213	-0.161	0.583	87.9	1.5	0.43	15.3
Incubation (Oct_Mar)	0.324	-0.029	0.671	96.7	1.6	0.49	12.8
Emergence (Jan_Mar)	0.315	-0.04	0.66	96.4	1.4	0.48	13.0
Fry (Feb_Mar)	0.306	-0.051	0.652	95.8	1,4	0.47	13.3
Parr (Apr_May)	0.103	-0.303	0.486	70.7	1.2	0.40	16.5
Early incubation water temp.	-0.662	-0.966	-0.367	0	NA	0.72	0.0

b) Beverton-Holt Model

Covariate	**mean**	**LCL**	**UCL**	**p>0**	**2x**	**r^2**	**ΔDIC**
No covariate	NA	NA	NA	NA	NA	0.25	20.4
Spawning (Oct_Dec)	0.299	-0.107	0.727	93.1	1.8	0.31	20.4
Incubation (Oct_Mar)	0.384	-0.01	0.784	97.1	1.7	0.39	18.7
Emergence (Jan_Mar)	0.36	-0.055	0.768	96.3	1.4	0.39	19.2
Fry (Feb_Mar)	0.355	-0.049	0.77	95.9	1.4	0.38	19.4
Parr (Apr_May)	0.002	-0.442	0.453	50.4	1.0	0.25	22.6
Early incubation water temp.	-0.74	-0.996	-0.482	0	NA	0.71	0.0

Table 9.5.2. Preliminary comparison of the Ricker stock-recruit model based on flow covariates and weighted useable area (WUA) covariates predicted by PHABSIM for the Stanislaus River. See caption for Table 9.5.1 for additional details.

		γ (covariate effect)						
Covariate	**Type**	**mean**	**LCL**	**UCL**	**p>0**	**2x**	**r^2**	**ΔDIC**
No covariate		NA	NA	NA	NA	NA	0.39	14.7
Spawning	Flow	0.213	-0.161	0.583	87.9	1.4	0.43	15.3
	WUA	0.081	-0.307	0.461	66.7	1.1	0.39	16.7
Incubation	Flow	0.324	-0.029	0.671	96.7	1.6	0.49	12.8
	WUA	-0.313	-0.678	0.032	3.9	0.3	0.48	13.2
Fry	Flow	0.306	-0.051	0.652	95.8	1.4	0.47	13.3
	WUA	-0.393	-0.731	-0.068	1.1	0.1	0.54	10.6
Parr	Flow	0.103	-0.303	0.486	70.7	1.2	0.40	16.5
	WUA	-0.356	-0.703	-0.021	2.0	0.2	0.54	11.9

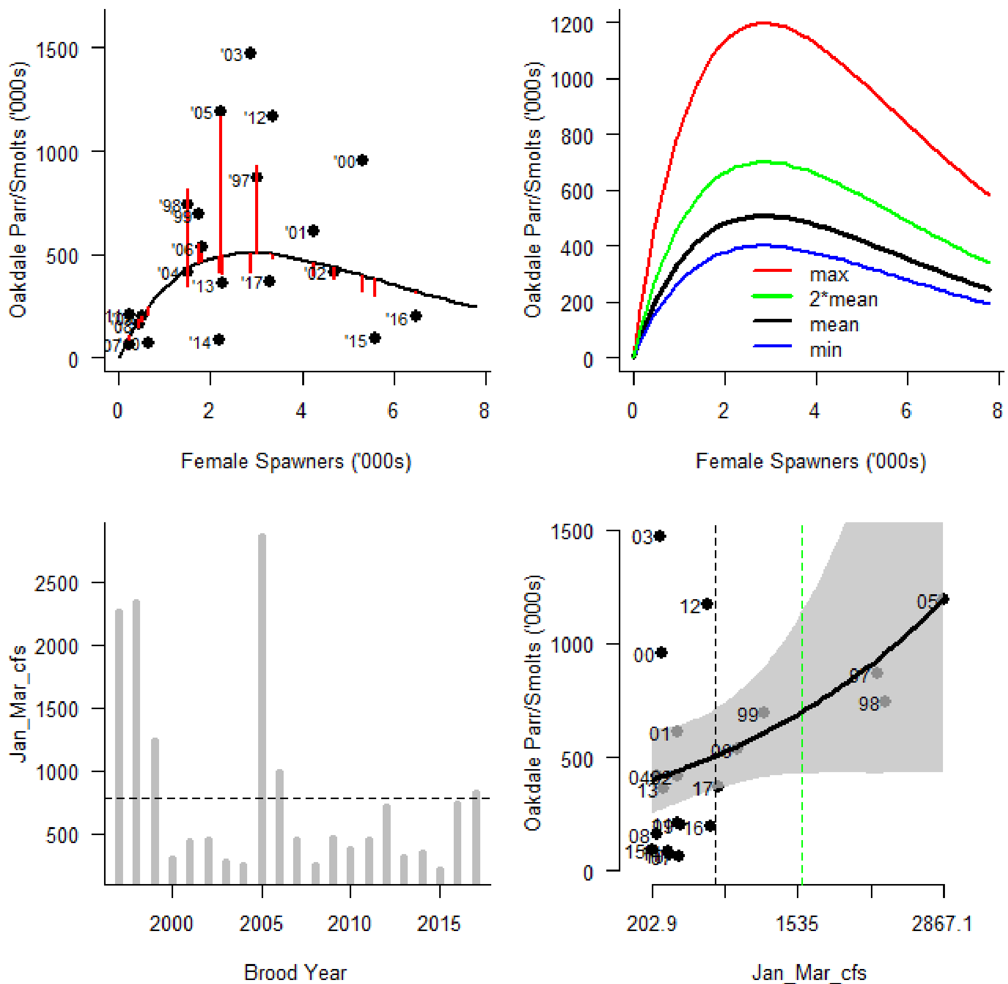


Figure 9.5.1. Fit of the Ricker covariate model to juvenile (parr + smolt) fall-run Chinook Salmon abundance estimates from the rotary screw trap at Oakdale as a function of the number of female spawners and flows during the emergence period (Jan 1-Mar 1). The top-left panel shows the female spawner and juvenile estimates in each year (points with labels identifying brood year). The black line represents the Ricker stock-recruit relationship at the average of flows during the emergence period across years. The vertical red lines show predictions based on year-specific flows. The lower-left panel shows the covariate values (average Jan-Mar flows) by brood year with the horizontal line showing the across year average. The upper right panel shows predictions from the model based on the across-year maximum, average, and minimum flows, as well as based on a 2-fold increase in flow relative to the mean. The bottom-right panel shows the predicted relationship between flow and juvenile production at the average female spawner abundance (grey band represents the 95% credible interval). The vertical dashed black and green lines identify the average flow across years and a two-fold increase in the average, respectively.

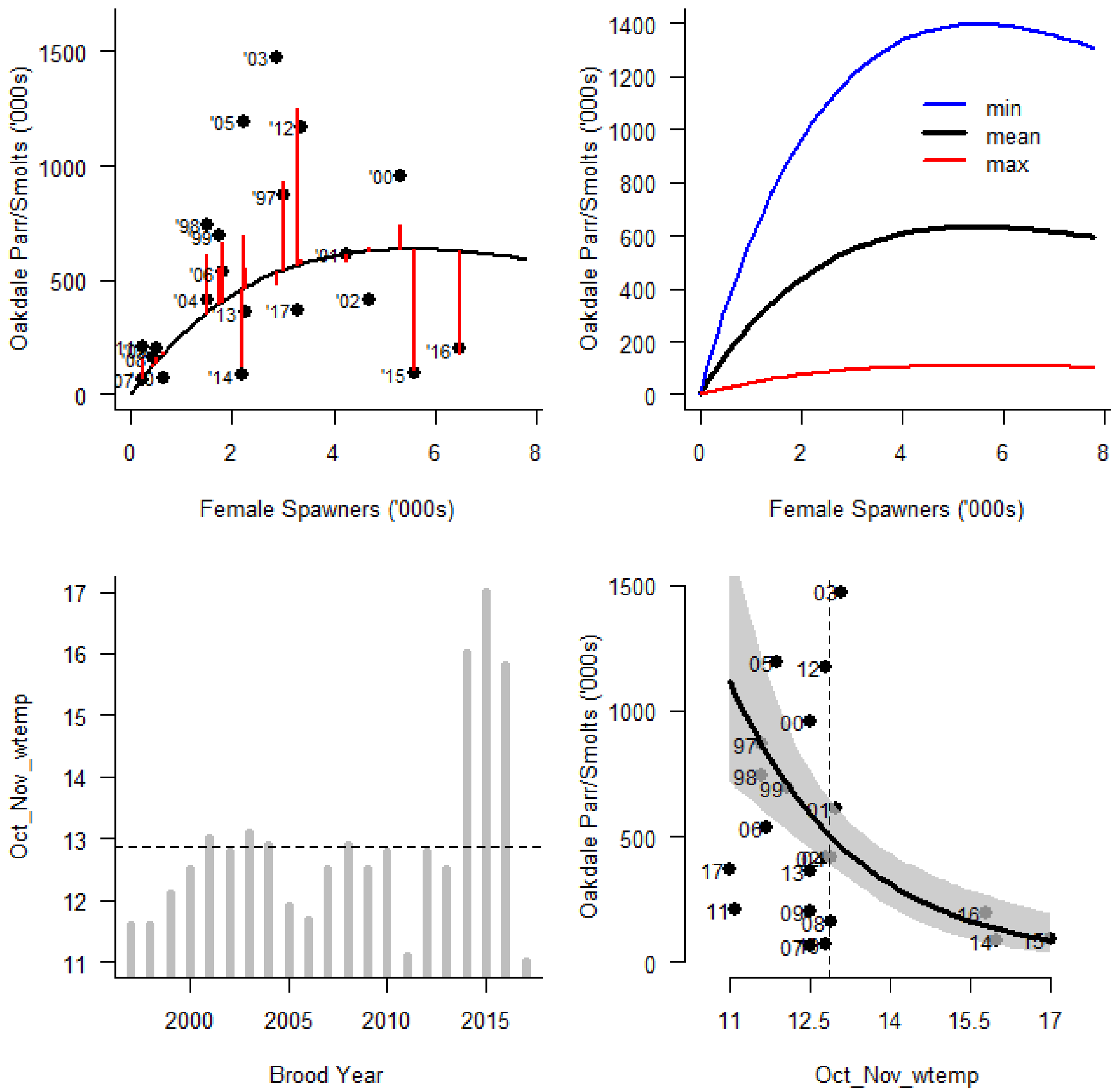


Figure 9.5.2. Fit of the Ricker covariate model to juvenile (parr + smolt) fall-run Chinook salmon abundance estimates from the rotary screw trap at Oakdale as a function of the number of female spawners and average water temperature during the early part of the incubation period (Oct 1 – Nov 30). See caption for Figure 9.5.1 for additional details.

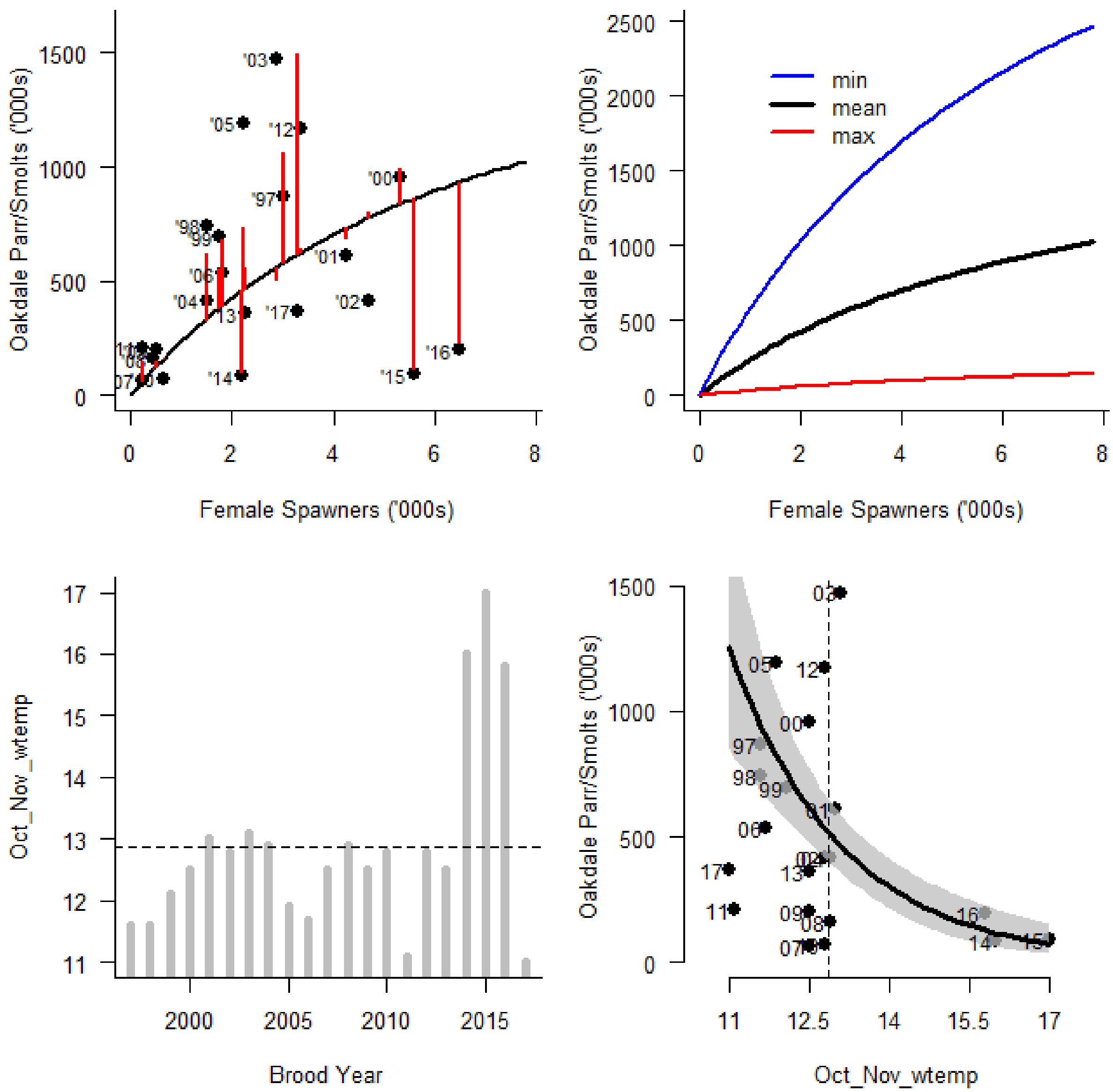


Figure 9.5.3. Fit of the Beverton-Holt covariate model to juvenile (parr + smolt) fall-run Chinook Salmon abundance estimates from the rotary screw trap at Oakdale as a function of the number of female spawners and average water temperature during the early part of the incubation period (Oct 1 – Nov 30). See caption for Figure 9.5.1 for additional details.

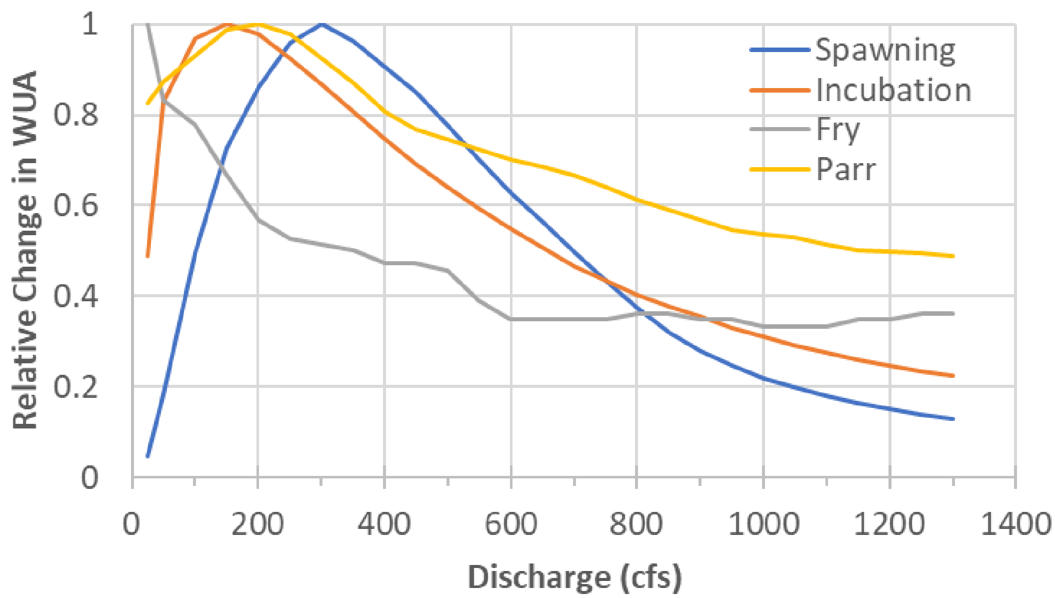


Figure 9.5.4. Standardized predictions of weighted useable area (WUA) as a function of discharge (cubic feet per second at Goodwin Dam) for different stages of fall run Chinook salmon in the Stanislaus River based on results from Aceituno (1993). WUA predictions were standardized for each life stage so maximum values were one. This allowed a comparison of WUA-flow curves on the same plot.