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

**A draft report to the**
**Delta Science Program**


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# EXECUTIVE SUMMARY

The charge to this Panel from the State Water Board was to develop a methodology for formulating biological goals for narrative objectives in the Bay-Delta Plan. The focus is on methods for evaluating the responses of targeted species and ecosystems to management actions that include major manipulations of flow and other actions such as habitat restoration. The biological goals are to be formulated such that they could be assessed using data from current monitoring programs and existing data. However, realizing the inadequacy of monitoring for some geographic areas and topics, we also considered additional metrics and methodologies that could significantly improve evaluation of species and ecosystem responses to restoration actions. We divided our 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.

Chapter two 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 outflow, 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, because of the importance in the estuary of tidal oscillations and the response of salinity to variations in freshwater flow, and the much longer and richer data sets available for the estuary than for the rivers.

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 properties (e.g., nutrients, dissolved oxygen, contaminants). 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 insects. In the estuary, phytoplankton primary production and zooplankton secondary production can be estimated from existing monitoring programs. We emphasize, however, that 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 in the 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 we recommend for attention 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.

The general method proposed in Chapter three for evaluating effects of major management changes plus overall environmental conditions on other fishes in rivers and the Delta is to use abundance metrics based on data currently collected by 18 different fish surveys that encompass wide areas. An important departure from our charge is that we include responses of unlisted native fishes and non-native fishes, as well as listed fishes, to evaluate responses to proposed changes in flow regime and habitat restoration. We contend that assessing responses of all fish species should expand our ability to detect and identify the mechanisms of change, positive or negative, and then relate those responses to management of the environment to favor listed species or other species of interest.

We list eight possible methods that use river and Delta fishes for evaluating progress toward biological goals through flow and habitat improvement projects. Metrics include: (1) species abundance trends; (2) the distribution and abundance of selected fish species as indicator species; (3) the collective abundance and distribution of common non-native fish species; (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; (6) a metric that measures response of native fishes to flow releases in tailwaters below dams; (7) responses of selected species of fish and invertebrates to outflow changes; and, (8) an Index of Biotic Integrity. If this broad-based approach is reasonable, then we recommend developing several or all of these approaches to be used simultaneously.

In some cases, more detailed information can be collected from individual fishes taken in surveys that has the potential to provide further insight, and potentially more rapid assessment of changes to specific species than the abundance-based surveys

themselves. For some well-studied species (e.g., striped bass), Viable Salmonid Population (VSP)-type metrics could be used (see Chapter four for a VSP description). Finally, using landscape metrics (e.g., extent of aquatic macrophytes in Delta) could provide additional insights into fish responses to a changing environment.

Overall, given the complex life history patterns and ecological relationships that Delta fishes have with their environment, the best approach is likely to be one that uses multiple approaches and metrics. This would be particularly true if some metrics are not sensitive to short-term changes, but useful for measuring long-term changes that might allow distinction of the fishes' responses to management actions from natural environmental variability. Collectively, the fish and other metrics discussed here may start to tell the story (but not clearly) after 5-10 years at a minimum. Regardless, there will be a continued need for standardized surveys to continue and for new ones to develop that target the important fishes of today.

Chapter four 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.

Our recommended framework for evaluating abundance and productivity of salmonids accounts for density dependence in survival rates by using a stock-recruitment relationship. 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 from spawner-to-smolt or from spawner-to-adult stock-recruitment 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. An obvious goal for intrinsic productivity is that adult recruits (returning adult salmon prior to any losses from fisheries) per spawner exceeds one. Trends in intrinsic productivity can be evaluated over time using a state-space approach to determine whether conditions are improving. Additionally, the number of adult salmon returning from the parent spawning population can be examined over time to evaluate whether a natural population heavily supplemented with hatchery spawners is sustainable at that level if

hatchery fish were removed. A key potential goal for abundance is the number of spawners that leads to maximum production of juveniles or future adults.

Two of the most relevant diversity metrics include the proportion of hatchery-origin spawners in the wild (pHOS) and age composition of the returning natural-origin salmon. These metrics are also needed to reconstruct salmon recruitment to the watershed so that stock-recruitment parameters can be estimated. A reduction in pHOS is a sensible biological goal if the objective is to maximize productivity of the naturally spawning population. Biological goals for age composition and spatial structure are less straightforward, but we discuss some possibilities for these two VSP metrics.

We also recommend a covariate stock-recruitment estimation approach for quantifying population-level responses of salmon and steelhead due to changes in water flow, habitat restoration, and in pHOS. Ideally, this approach would use data collected before and after restoration actions, and possibly involve watersheds without actions that could serve as a control for environmental change. Natural interannual variation in flow may provide necessary contrast to quantify fish responses to purposeful changes in flow.

Monitoring of Chinook salmon in the Central Valley has improved in recent years, but key metrics such as those described above 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 abundance. Ultimately, data are needed to create "brood tables" for each population to support stock-recruitment analyses. Statistical studies indicate that a minimum of 15 to 20 years of monitoring data will be required to provide a reasonably reliable evaluation of flow and habitat treatment effects.

Finally, we emphasize that well-designed programs of assessment should accompany any major actions in the future, based on metrics herein and others. The cost of major actions is high, as is the uncertainty in outcomes from changes in flow or habitat, even if these are intended for increasing abundance of fish. Therefore, any major action should be embedded in an experimental design including predictive modeling, adequate monitoring and ancillary investigations, rigorous assessment of outcomes, and a designed method for feedback to change the monitoring program, the expectations for the action, and even the action itself if required.

# 1 INTRODUCTION

Defining biological goals for 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 a complex landscape like the San Francisco Estuary (SFE) and its watershed, which includes a large and heterogeneous estuary, two major rivers, and multiple tributaries. Biological goals also need to be clear, concise, quantifiable, and achievable within appropriate timeframes. Biological goals will vary by aquatic ecosystem type (tributary rivers, major rivers, and the estuary) and ultimately must be designed for specific ecosystems for which defined biological components have been identified for assessment.

This independent scientific advisory panel (Panel) was established to advise the State Water Resources Control Board (Board) on developing biological goals for measuring and assessing ecological responses to actions taken under the Board's Update to the Bay-Delta Water Quality Control Plan (Plan) (https://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/). Biological goals are intended to be linked to ongoing monitoring programs and to build upon long-term data sets. However, the Panel can also recommend areas where additional monitoring and targeted studies, coupled with analysis, synthesis, and integration, should enhance understanding of these novel and changing ecosystems, as well as interactions among them and their responses to multiple stressors and drivers of change.

The Board charged the Panel to develop recommendations for formulating quantifiable biological goals for three topics: ecosystem structure and function, other native fish species, and salmonids. The Panel was not asked to create biological goals, but to provide guidance on methods or approaches to formulate biological goals. The spatial extent 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 1.2).

Healthy, self-sustaining riverine and estuarine ecosystems provide crucial ecological and social services that are essential for human populations (Palmer et al. 2005). Billions of dollars are spent on the restoration of these ecosystems, but most restoration 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. Clear and concise biological goals are required to evaluate whether these restoration projects are effective.

The Board developed a series of charge questions for consideration by the Panel that addressed the three topics. Panel members in turn made suggestions to modify the Board's charge questions while maintaining the direction and intent of inquiry requested by the Board, thereby facilitating more direct and helpful answers to the questions. Briefly, the major themes for the charge questions under the three topic areas, and how the Panel approached them, are:

(a) Questions concerning ecosystem structure and function are directed towards how to measure the effectiveness of flow regimes, habitat restoration, and other non-flow efforts for improving ecosystem condition. More specific questions involve what ecosystem processes should be monitored. Some important differences exist between the estuary and rivers in its watershed. Some of these differences include distinctive hydrodynamics, variable salinity distributions, the role of floodplains, and the duration and commitment to monitoring and research programs. Methods and approaches to developing or enhancing 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, if there are other metrics to consider using existing monitoring data, and what additional parameters should be monitored for native fishes. We determined that just monitoring native fish species was insufficient, especially for the estuary, because: (a) some native species, such as delta smelt, are so rare that meaningful, quantitative results are unlikely; (b) monitoring data or biological knowledge of most native species is insufficient to support meaningful evaluations; (c) 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 that would indicate the condition of the estuary or river; and, (d) some abundant non-native species are associated with habitats that are unfavorable for native species, so could thus serve as negative indicators of responses. The Panel also determined that using fish assemblages that encompassed a

wide array of native and non-native species had promise for determining broad-level responses to change.

(c) Questions about Chinook salmon and steelhead populations focus on the use and relative importance of VSP criteria for evaluating status and trends and their population-level responses to major flow and habitat restoration efforts. VSP criteria include abundance, productivity, diversity and spatial structure. More specific questions involve the influence of hatchery-origin salmonids, water-year type, and data quality on the ability to track salmonid population responses to restoration actions and track progress towards biological goals. The time frame for detecting population responses also is discussed.

Assessment of monitoring methodologies 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 Bay-Delta Plan 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. For the estuary there are 18 regular fish monitoring programs, most developed for specific purposes; 10 sample open waters by various methods for pelagic juvenile salmonids, smelt, and striped bass.

Determining the full extent of the purpose, types, and details of extant ecosystem and fish monitoring throughout the entire SFE and watershed is beyond the scope of work for the Panel. Fish monitoring 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.

## 1.1 Why Has the Board Asked the Panel for This Report?

The Board is seeking recommendations from the Panel for scientifically defensible methods to formulate biological goals that can be used to assess progress toward achieving the narrative objectives in the 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 communities; and, (b) the ability of ecosystems to support native fishes. The Panel is not expected to create the biological goals themselves. Instead, the Panel has been asked to provide guidance and recommendations on the methods or approaches that should be used to formulate biological goals.

Well-crafted biological goals are intended to serve as quantitative benchmarks for assessing viability, and 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 best available scientific information, 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 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 regulations 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, the San Joaquin and Sacramento rivers, and tributaries to these rivers. There is considerable debate among stakeholders about the effectiveness of flow and non-flow management actions for improving fish populations. In practice, 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 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 2006, Cloern et al. 2016, Whipple et al. 2012). We do not describe these alterations here except to highlight key features that impinge on the relationships among flow, habitat, management, and ecological structure and function for fishes and other aquatic organisms. This is a novel ecosystem that can be improved for the benefit of focal species but not returned to its former state.

The Board's Charge to the Panel identified three geographic and somewhat distinct ecological regions for examination of flows and their effects: 1) the San Joaquin River and its major tributaries; 2) the Sacramento River, its major tributaries and tributaries that enter the eastern Delta; and, 3) the Delta region of the estuary. In this report we use "Delta" to mean the legal Delta but take an expansive view of our charge to include as much of the estuary as appropriate for evaluating potential effects of proposed actions on habitat, processes, and species.

Several features of the SFE and watershed are prominent in our discussion about the three geographic areas. First, although the rivers and estuary are connected by movements of water, substances, and organisms, the two differ enough so that they are studied by different communities of scientists. Nevertheless, both rivers and estuary are subject to anthropogenic and other changes that are permanent, while both are also subject to shorter-term, more ephemeral impacts such as variation in flow and temperature. Below we briefly discuss salient features of this system and some key differences between the rivers and the estuary (Sections 2.1-2.3).

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 salmon and other 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) and of small remnant tidal wetlands (Grimaldo et al. 2009, Howe and Simenstad 2011). Information from these studies has led to efforts to modify the land-water interface through manipulations of flooding and

restoration or reconstruction of wetlands. However, current land uses in many locations preclude expanding the extent of floodplains or the area subject to tidal flows.

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 a permanent feature 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, daily flows are sustained at artificially high levels into summer, and diurnal fluctuations are unnaturally amplified for power production. These alterations have identifiable effects on the riverine ecosystems by reducing connectivity among habitats, mismatching flow patterns needed for establishment of riparian vegetation, removing critical cues in the life histories of salmon and other native fishes, and stranding fishes.

Within the estuary, a major role of riverine flood flows is to supply the estuary with sediment. Water in the estuary has become clearer as the sediment pulse due to hydraulic mining has been winnowed (Schoellhamer 2011). This trend is expected to continue because much of the sediment mobilized in the watershed is trapped behind dams. Loss of sediment through erosion and seaward transport in the estuary causes an imbalance, such that the sediment supply today is insufficient to meet demand for maintenance and restoration of wetland habitats in most areas. Increased water clarity also degrades habitat quality for some fishes (Feyrer et al. 2007).

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 as well as fishes 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 ("X2"[1], Jassby et al. 1995) with accompanying changes in physical dynamics (Kimmerer 2002). Therefore, even large flow manipulations have minor effects on the more saline reaches of the estuary west of the Delta.

[1] (Practical Salinity Scale; location indexed as "X2", Jassby et al. 1995)

Non-native species now dominate fish, aquatic invertebrate, and aquatic plant communities in both the rivers and the upper estuary. Some introduced species (e.g. Brazilian waterweed, water hyacinth, overbite clam, Siberian prawn, Mississippi silversides) exert stronger influences on ecosystem function than other introduced species. The effects of non-native 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 non-native biota that form distinct assemblages. These assemblages, such as those associated with Brazilian waterweed and water hyacinth, support 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 to better support native species.

Discharge of contaminants and nutrients affects the aquatic ecosystems, although their effects can be difficult to quantify. Numerous chemicals in agricultural and urban use have known toxic effects on aquatic organisms (Weston et al. 2015). Nutrients are generally considered stimulants to phytoplankton production, though ammonium has been implicated in some of the declines in phytoplankton productivity in the estuary (Dugdale et al. 2007, 2017). Nutrient loading into the Delta also may help to stimulate blooms of nuisance algae (*Microcystis*) and the invasive waterweeds that choke some Delta waterways (Dahm et al. 2016).

The extreme modification of the river-estuarine system has considerable implications for our charge. For one thing, the past is only a rough guide to the future. We are stuck with novel ecosystems that are constantly changing. This greatly restricts the reliability of predictions about how the SFE and watershed will function in the future.

## 1.3 Future, Long-term Change

External change (i.e., that is due to causes other than the planned actions) may interact with the 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 poses greater difficulties both for achieving goals and for measuring progress toward them.

Although we cannot address long-term, progressive change, such as is associated with the modern climate regime, climate change will undoubtedly influence the response of the ecosystem and fish communities to purposeful changes in flow and habitat restoration. Effects of climate change vary across the landscape (Cloern et al. 2011). Increased atmospheric warming affects both the river and the estuary, although water temperature in the more seaward reaches of the estuary is influenced more by ocean conditions than local conditions. Warming is also causing a shift to earlier snowmelt runoff in the Sierra Nevada (Roos 1989, Stewart et al. 2004). This change and a likely increase in the frequency of large storms and more frequent extremes in climate (Dettinger et al. 2016) together narrow options for shifting dam operations between winter flood control and capture of runoff in spring, reducing water availability in some years. Sea level rise will inundate wetlands and, where infrastructure blocks landward migration, marshes may drown unless plant production of biomass and addition of sediment can enable them to keep up with the rising seas. Ocean acidification is unlikely to affect brackish to fresh parts of the estuary where its effect is diluted by fresh water.

Other long-term change includes 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 alien species, fish hatchery operations, failure of levees resulting in long-term or permanent reconnection of subsided lands to waterways, and changes in wastewater treatment and discharge. Large-scale planned or unplanned changes will have effects that in most cases are difficult to predict, partly because they have not been adequately studied. For example, proposals to divert fresh water from the Sacramento River (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 have not been critically analyzed. Likewise, there has been little study of ecological effects of the impending upgrade to the Sacramento Regional Sanitary District's wastewater plant.

The combined effects of climate change and local modifications result in trends that are likely to have substantial effects on the riverine and estuarine ecosystems and their 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 some salmon populations (Lindley et al.

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

2007). The Delta pelagic food web remains unproductive but may rebound following reduction of nutrient loading, and tidal wetland restoration will likely result in improvements in some ecosystem attributes. Finally, further introductions of new 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 of conservation of native species (Moyle et al. 2014).

## 1.4 Experimental Management

Ideally, variables used to track progress toward biological goals should quantify the effects of additional water allocation and habitat restoration on target species. The response of fish populations to management actions are often highly uncertain; therefore, planned management actions may not have the hoped-for effect. Our view of uncertainty in population response to changes in flow and habitat is likely shared by many 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 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 and Holling 1997), which is to reduce uncertainty over time by treating management actions as large-scale field experiments, either through explicit manipulations (active) or through 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 US 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 established (Castleberry et al. 1996), the wrong variables were selected, and uncertainty in model predictions was ignored.

Monitoring design must account for imprecision or bias in measurements that may mask responses. For example, a change in smolt production will take longer to detect if annual estimates lack precision, and for most programs at least a decade of monitoring under each treatment is required to detect a reasonably large effect (Bradford et al. 2005). Increases in river flow may lead to underestimates of smolt production if smolt detection probability is reduced at higher flows. These issues can largely be resolved by designing programs that anticipate the magnitudes of responses, developing robust monitoring programs, and accepting that outcomes of management experiments at the population level can 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. This eliminates some extraneous variability (e.g., that in subsequent survival), but requires an assumption that enhancements of 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 difficult or impossible 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 the flow increase. However, no two years are alike, so it would be impossible to attribute changes in smolt production to the difference in flow. One solution to this problem is to use a BACI (Before/After and Control/Impact) design (Underwood 1992) whereby a similar river is used as a control. However, no two rivers are alike in their temporal patterns, limiting the suitability of this method, and there are no controls for large systems such as mainstem rivers and the estuary. The difficulty of measurement is amplified if flow is increased and habitat enhanced at the same time, making it difficult to separate the two effects. On a more optimistic note, natural variation will likely provide contrasts in flow (e.g., critically dry vs. wet years) over periods long enough to make inferences about purposeful flow manipulations.

Our main objective of reviewing these challenges is to highlight that making useful inferences about the effects of an 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. In the next three chapters we consider both the appropriateness of objectives and variables to measure in the ecosystems, native fishes and salmonids of the SFE and watershed.

# 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 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, we examine some key general characteristics of the physical, chemical, and biological environments of rivers and estuaries. We begin with a broad discussion of

the principles that govern rivers (Section 2.2) and then estuaries (Section 2.3), then 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. We then discuss specific topics for developing goals for pelagic, estuarine nearshore, and riverine ecosystems (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 measured 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 our management approach 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 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% 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 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 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 of the 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 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.

The considerations presented for rivers based on our growing understanding of the structure and function of river ecosystems provide important guidance when establishing meaningful biological and abiotic goals for the highly regulated Central Valley rivers. Some key examples include capturing the concept of river structure and function, considering interactions of flow with 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 example of such an approach is the Kissimmee River Restoration Evaluation Program in south Florida (Text box A).

Central Valley 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). The flow regimes incidentally 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 the hatchery contribution (Chapter four). Riparian plants and birds as well as native aquatic invertebrates may also benefit from the 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. 2018).

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) takes 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 Is.) to 55 km (Martinez) takes 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 is a key area of 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. 2018a). 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, 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

[3] We use "restoration" here to mean construction, rehabilitation, or otherwise production of landforms that can provide habitat for aquatic organisms.

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 and Holling 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 Central Valley 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 Central Valley (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).

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, at the Delta scale it seems too small to recover the important role that the historical wetlands likely had in ecosystem structure and function at the scale of the broader estuary.

[4] http://resources.ca.gov/ecorestore/

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 and detritus that is distinct from the phytoplankton-based pelagic food web (Grimaldo et al. 2009, Howe and Simenstad 2011). Moreover, most estuarine fishes, including salmon and delta smelt, use these habitats (Sommer et al. 2001, Grimaldo et al. 2009, 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. 2018b). 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 in 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 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. From this perspective, we can separate ecosystem attributes 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 we focus on these categories, first emphasizing direct ecosystem support for fish habitat. We next consider additional ecosystem attributes necessary for the ecosystem to support fish, particularly through food supply.

### 2.5.1 Physical habitat

Throughout both the estuary and rivers, key physical characteristics include sediment transport rates and grain size, water velocity and flow rate, water temperature, and salinity (conductivity). The spatial and temporal configuration of the landscape can have a profound effect on the biotic components of the 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 and establishment or destruction of patches of terrestrial vegetation, which in turn affect flow patterns. In the SFE and watershed this historical spatial and temporal variation in physical configuration has been greatly altered. Major changes in flow and habitat are required to improve abiotic and thereby biotic habitat for fishes at a meaningful scale.

In the rivers, high flow strongly influences the spatial and temporal patterns of habitat attributes, such as by accelerating river currents, and scouring and suspending fine sediments thereby increasing turbidity, depositing sediments in low-flow areas, and

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

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, this will change but we are aware of no scientific studies predicting these changes. Numerous smaller diversions, mostly unscreened, operate throughout the system and their impacts are unknown (Moyle and Israel 2005).

Temperature is a fundamental control variable for all living systems. Several fish species of concern in this system are subject to thermal stress, notably winter 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, 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 (Chapter three).

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 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 (San Francisco Bay Area Wetlands Ecosystem Goals Project 2015). Dredged sediment is being used as fill for re-establishing wetlands, but this supply is also limited.

### 2.5.2 Chemical constituents

Low 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 algae growth and low dissolved oxygen (DO). Many species are sensitive to low DO, particularly cold-water fishes such as salmon.

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 of 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 effects in bioassays, Weston et al. 2015). A thorough treatment of these effects is well beyond our 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.

### 2.5.3 Non-native species

Some of the organismal groups in the SFEW 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; it is also true of the aquatic plants of the Delta and rivers, the zooplankton of the upper estuary, and the benthos throughout the estuary and rivers. These taxa form what appear (over our limited time scale) to be

persistent non-native assemblages. However, the principal reasons to distinguish native from established 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.

The rate of introductions appears to have abated, though risks still 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 guessed (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 should 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.

### 2.5.4 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; below we expand on this discussion for other food webs. 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 a plant- and detritus-based food web (Grimaldo et al. 2009, 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. 2018b, 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 finding of high copepod growth rates in the Yolo Bypass that were positively related to biovolume of cyanobacteria, and not to those of phytoplankton that are generally considered to be more 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 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 the lower trophic levels of the estuary do not respond strongly to freshwater flow.

The benthos is included in Figure 1 mainly to represent losses of plankton to grazing by two species of clam. We do 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. 2018a). It also should be monitored because benthic organisms, especially amphipods, are important prey of many fishes, such as tule perch, prickly sculpin, splittail, and various non-native 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 the 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).

### 2.5.5 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 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). In less-modified estuaries, shallow areas and wetlands are loci for important biogeochemical transformations. These biogeochemical processes are no doubt less important to the SFE as a whole simply on the basis of diminished area and have received relatively little monitoring and research until recently.

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 (see Other Fishes and Salmon sections). Shallow tidal areas are somewhat isolated from the larger estuarine channels, and likely respond differently to changes in freshwater flow and other large-scale management actions. 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. 2018b).

The need to include measurements in these shallow areas rests on the substantial use of these areas by smaller salmon that are more likely to be of natural origin (Miller et al. 2012, Chapter three), as well as other fishes. However, much of the food of these fishes is benthic or associated with 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. 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 such as largemouth bass 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 three).

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, a sample of shallow areas should be included in benthic and planktonic monitoring programs to improve our ability to forecast conditions in future restoration sites.

### 2.5.6 River ecosystems

The major rivers 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 three), and a growing interest in freshwater conservation (Howard et al. 2018) and defining the patterns and magnitude of flow alteration in the tributary rivers (Zimmerman et al. 2018, Sengupta et al. 2018). These rivers have varying degrees of hydrology, water quality, and biological assessment (e.g., Mazor et al. 2016, Ode et al. 2016, May et al. 2015), and each ecosystem has specific attributes that must be considered in the design of an effective evaluation program. 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. A wide range of professional expertise and local knowledge is required to achieve the best outcomes for the rivers of the Central Valley.

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 are sampled that link to the main goals 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, Kissimmee River Restoration Program, Grand Canyon Monitoring and Research Center), we recommend 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 three and four, we focus here on the other five categories.

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 temperatures and flows during spawning, egg incubation and rearing, the turbidity and level of fine particles in spawning areas, and the frequency of no water on redds.

There are many 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 region where higher temperatures are becoming increasingly commonplace. Temperature mapping that includes three-dimensional coverage including shallow sediments is of growing importance. Dissolved oxygen is critical to measure as temperature increases and river metabolism changes. 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 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 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, algal, macroinvertebrate, and fish communities), but river function is increasingly being used to assess effects of environmental stressors (von Schiller et al. 2017). Examples of functional measures include ecosystem metabolism, organic matter decomposition, secondary production, predation rates, nutrient cycling, 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 can be used to determine long-term growth rates and carbon sources and food web linkages.

Algal bioindicators can be useful components for setting biological goals in Central Valley 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 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., wastewater or agriculture).

Aquatic invertebrates also have a long tradition of use in bioassessment of rivers because they are common, widespread, and easily sampled, and often 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 Central Valley 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 the fish community. 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, we recommend 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 to be Measured

### 2.6.1 Delta metrics

Aquatic ecosystems have many attributes that should be measured to understand how well they are functioning from a human perspective. Here we present a long list of possible metrics; in the ideal world, most would be measured frequently, at multiple locations, which would presumably be cost prohibitive. Realistically, a subset should be chosen for monitoring. The first suite of metrics focuses on the Delta. Not all of these metrics are recommended for every situation. This depends on what the 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
- Salinity (estuary) or specific conductivity (rivers)

**Metrics of chemical condition:** some of these are now feasible with continuous monitors, 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. Which metrics to measure needs to be determined by the main concerns within the 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

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## 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 programs provides valuable information to the public, to decision makers, and to politicians on this major public investment in river restoration.

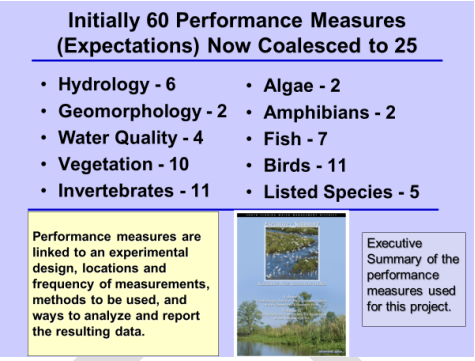
Developing the expectations (analogous to biological goals but including physical and chemical goals) followed a well-defined process (Kissimmee River Restoration Studies 2006 and Dahm et al. 1995). The overall goal of the restoration projects is to create ecological integrity as defined by endpoints 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 between 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 amphibians and reptiles expectations 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 particular 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.




## 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 into a monitoring and evaluation program that included both Moreton Bay and the 15 major catchments flowing into Moreton Bay in 2002. Moreton Bay is about 1523 $km^2$ 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.

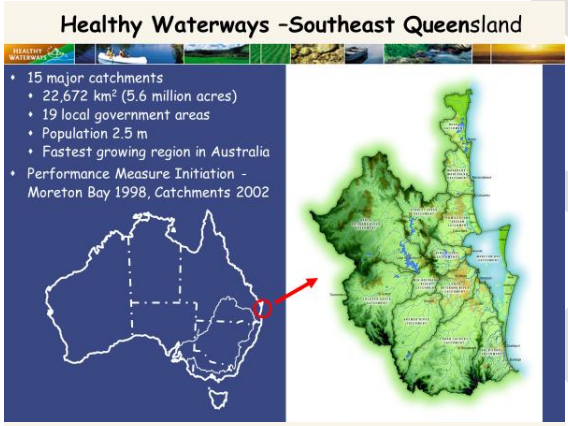
EHMP is an important tool for both managers and the public. Through the use of biological, physical, and chemical indicators, EHMP provides an objective assessment of the condition of the ecosystems being sampled. It has been implemented to independently evaluate how effective various environmental protection 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?

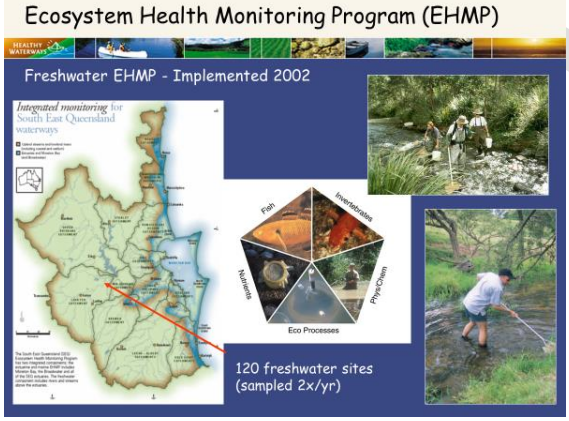
The freshwater component of EHMP was developed over a two-year period using 1) conceptual models to identify potential indicators, 2) river and stream classification to different river and stream types, 3) pilot studies to develop and assess indicators, and 4) a major field trial in multiple catchments. 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.



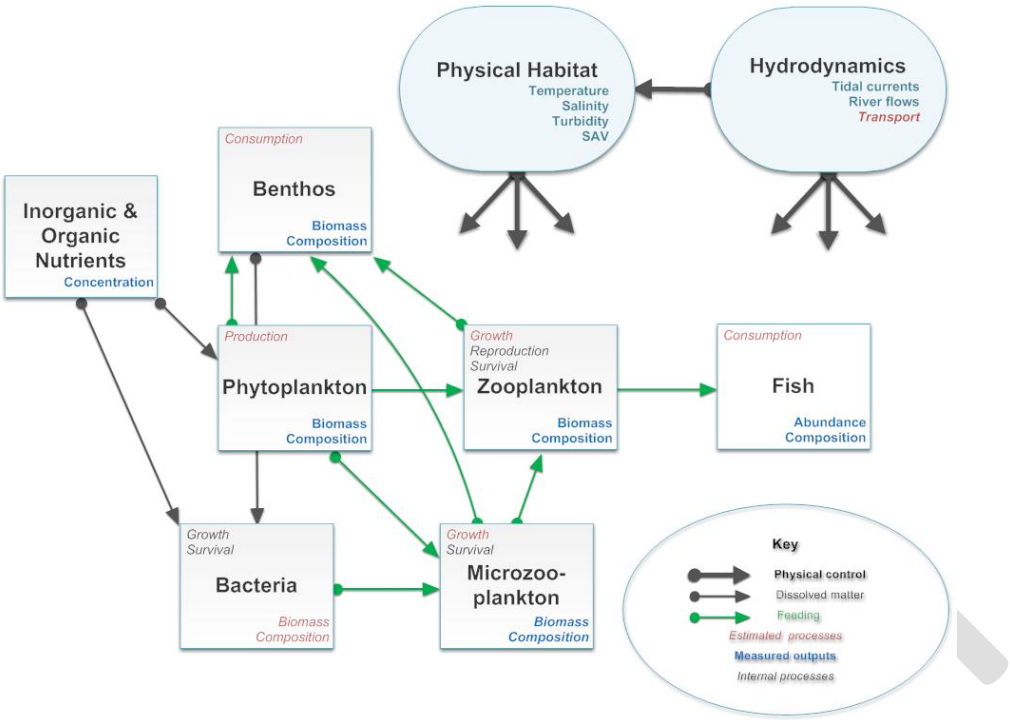



Figure 2.1. Simplified flow diagram of the pelagic food web of the upper Estuary. Boxes are major functional groups. Within each box in the lower right are measurable outputs (generally biomass and composition) and in the upper left are internal processes. Blue lettering indicates outputs that could 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 SAV. All of these factors influence the pelagic food web in various ways.

# 3 NATIVE FISHES AND FISH ASSEMBLAGES

## 3.1 Introduction

Controlled "environmental" flow releases are often made from dams into rivers of the Central Valley and then into the Delta[6]. These flows, however, are usually intended to improve conditions for species listed as threatened or endangered under the state and federal Endangered Species Acts (henceforth "listed species") or for unlisted runs of Chinook salmon. However, modifying flows as the main strategy 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 Tables 1 and 2). It is clear that a broader approach is needed to understand and enhance the effectiveness of new environmental flow releases or other management measures taken to improve the lot of declining native fishes. In this chapter we show why a broader approach should involve additional species, selected as indicator species, and fish assemblages (co-occurring groups of species with similar environmental requirements). In this chapter, we suggest several alternative metrics that can be used to more effectively monitor fishes to show responses to changes in flow and improvement in habitat. These metrics are designed to provide a "quantitative assessment of progress towards meeting the narrative objectives." These are labeled as "biological goals" in our charge with a focus on native fishes.

We think 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. Therefore, it is important to use as many species as possible, including non-native species, in order to have metrics that are sensitive to a broad range of conditions and are actually measurable. The approach of using both native and non-native species should provide a more comprehensive view of how change, whether through deliberate manipulation of the environment or not, is likely to affect fishes in the estuary and its inflowing rivers. It should also provide insights into how native and non-native fishes interact in the changing environment.

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

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

- Delta smelt: listed as threatened (later) under the plan; they are near extirpation today (Moyle et al. 2018).
- Longfin smelt: listed by California as threatened and near 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 Central Valley fall-run Evolutionary Significant Unit (ESU); the wild population has been replaced by a hatchery-driven population.
- Sacramento perch: extirpated from Delta, but present in ponds.

Unfortunately, this multi-species approach failed because most of the effort was placed on delta smelt recovery and the other 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 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 federal and state Endangered Species Acts (ESAs).

In this chapter, we explore some alternative approaches to the current strategy of management by listed species, one crisis at a time. We examine 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. Salmon and steelhead/rainbow trout are included as part of the approach but not emphasized, for reasons discussed in the next section. Our general approach should be compatible with the ecosystem-based approaches described in the previous chapter.

The basic rationale for this chapter is based on one of the charges given to our 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 the Appendix (Section 7.1).

The direct and indirect responses to the charge questions are organized through the following major chapter sections: (a) evaluation of current surveys for fishes, (b) evaluation of species most useful for future monitoring, (c) determination of species assemblages with potential for monitoring, (d) discussion of approaches for developing goals, (e) possible and recommended metrics, and (f) conclusions, a determination of what metrics might work best for monitoring of effects of new actions.

## 3.2 Surveys for Fishes

The diversity of existing fish surveys provides a means to identify possible indicator species and habitat assemblages and possible metrics based on these surveys, as requested in the questions above. Here, we pose answers to questions regarding the value of the existing surveys for monitoring: What are the major surveys that are currently used to assess status and trends in fishes in the Delta? What species are sampled by each survey and how well are they sampled? Are additional surveys needed?

### A. What are the surveys?

Surveys that encompass the Delta, Suisun Marsh, and, more broadly, the SFE, are listed below and described in Appendix Table 1. Most are also described in detail in Honey et al. (2004).

1. California Department of Fish and Wildlife (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. United States Fish and Wildlife Service (USFWS) beach seine
9. USFWS Chipps Island trawl

10. USFWS Mossdale trawl
11. USFWS Sacramento midwater trawl
12. USFWS Sacramento Kodiak trawl
13. University of California, Davis (UCD) Electrofishing surveys
14. UCD Suisun Marsh beach seining
15. UCD Suisun Marsh otter trawl
16. Department of Water Resources (DWR) Yolo Bypass fyke net
17. DWR Yolo Bypass beach seine
18. Delta pumping plant salvage

The USFWS Sacramento midwater and Kodiak trawl surveys (11, 12) are usually treated as one survey but the techniques are sufficiently different so that we treat them separately. The UCD electrofishing survey (13) is based on data 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 (15) consists of monthly sampling at 1-3 locations in Suisun Marsh, since 1980; it is rarely reported as a separate survey. The DWR Yolo Bypass fyke net survey 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 quantifies the relative abundance of 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.

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### B. 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); non-native fishes typically dominate captures in these surveys. However, many of these surveys were originally designed to target one or a few 'important' species, such as striped bass. As a result, improved sampling is needed for most Delta resident, non-migratory fishes; the principal source of information on resident fishes (survey 13) is actually a hodge-podge of short-term electrofishing sampling programs. 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, Cosumnes River floodplain, and lower Cache Creek.

**Figure 3.1 (next page).** 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.

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| | 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 | 0 |
| **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 |

**Rankings**
0 = No Catch
1 = Low Catch
2 = Moderate Catch
3 = High Catch

* indicates native species

### C. Explanation of Ranking System for Heat Map (Figure 1)

Abundance rankings 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. Examples of rankings in Figures 2-4 below are from CDFW Fall Midwater Trawl catch data.

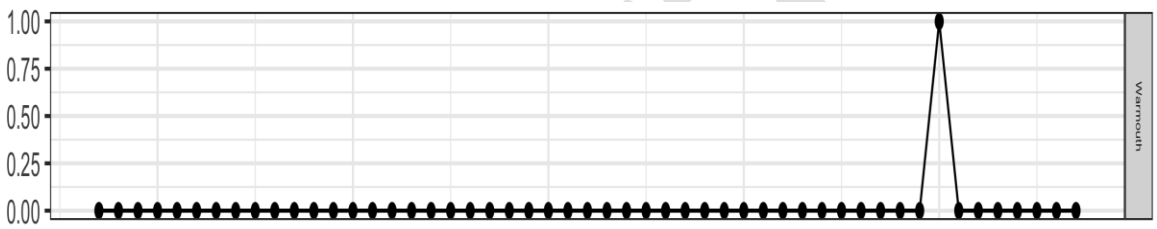



Figure 2. Example of rank "1" species. The species (warmouth) was captured only once over the course of the survey period. Note the scale on the Y-axis. Survey is not useful for determining abundance, trends, or dynamics of warmouth in the San Francisco Estuary.

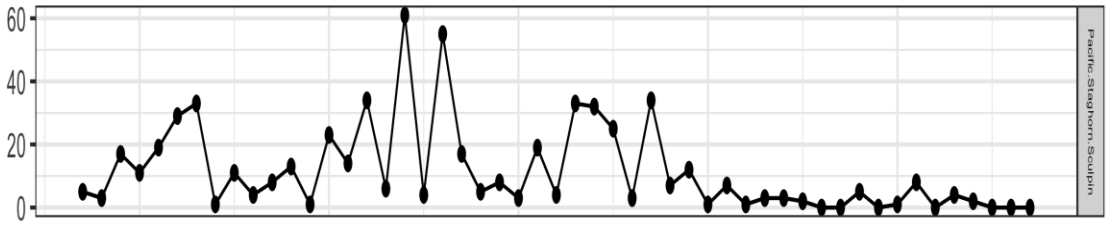



Figure 3. Example of rank "2" species. The species (Pacific staghorn sculpin) was captured most or all years of the survey period, but at low abundance. Note the scale on the Y-axis. Total catch appears to show a downward trend in catch over the survey period, and yearly catch exhibits considerable annual variation. 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.

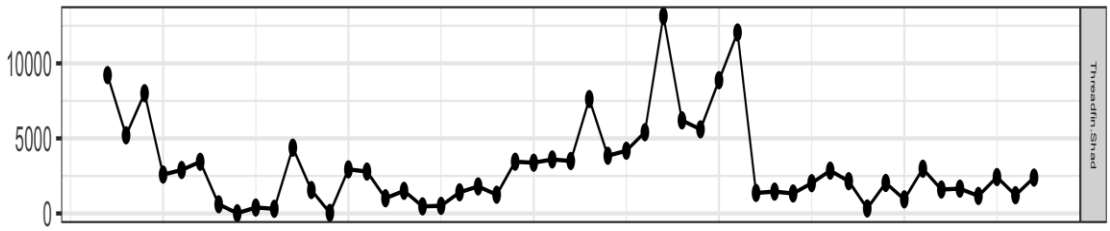



Figure 4. Example of rank "3" species. The species (threadfin shad) was captured in all survey years in moderate to high numbers. Strong trends and variability can be observed, with clear inflection points of decline/increase. Note the scale on the Y-axis. This survey is robust for analysis of threadfin shad abundance and dynamics within the SFE.

## 3.3 What Species are Useful for Monitoring?

### A. Delta species

A list of 36 common Delta species is presented in Appendix Tables 3.1 and 3.2 (Section 7.2), which show various measures of status and species characteristics, respectively. The questions become: Which species are likely to be useful as indicator species? Which species are likely to form assemblages of species that would react to change in similar ways?

### B. Characteristics: Delta indicator species.

Most conservation-oriented monitoring in the rivers and Delta has been oriented toward the four runs of Chinook salmon, steelhead, delta smelt and longfin smelt. This is because they support fisheries, are listed under the ESAs, or both. However, the monitoring has largely tracked population declines and/or collapse. Here we examine the role that other species might play as indicator species. By *indicator species* we mean those species that are sensitive to changing conditions and can represent responses of other species, especially native species. The two smelt species ideally would serve as indicator species but they are too rare to have populations with detectable responses to changing conditions, especially in the Delta. Their main value for monitoring is presence/absence to determine whether or not they are extinct in the estuary, following criteria of Baumsteiger and Moyle (2017).

Delta indicator species should have the following characteristics: (a) have year-around presence in the legal 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 life history, physiological characteristics, and distribution that are similar enough to those of native species, so that, if conditions favor the indicator species, the native species will likely 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 what we are calling indicator species. Our analysis in the next section (C) shows that by choosing indicator species carefully, our last characteristic can be met by a handful of species.

### C. Candidate Indicator Species.

Candidate indicator species, chosen from the list in Appendix Table 3.1 because of their potential to meet the above six criteria as indicator species, are shown in Table 3.1.

Table 3.1. Candidate indicator species in the Delta; letters stand for characteristics discussed in text. An 'x' indicates the species has the characteristic. A '?' indicates it may have the characteristic but there is insufficient information. Blank indicates 'no.' (**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 |

Candidate 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). They would thus seem the best candidates for indicator species. 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 be an indicator species.

## 3.4 Species Assemblages

### A. How were assemblages determined?

Potential fish assemblages (with **native** fishes in bold below) that could be used for monitoring were initially constructed by grouping fishes with common habitat requirements and with co-occurrence in the various sampling programs (see Appendix Table 2 for species habitat requirements). Groupings were refined by a simple correlation analysis among all species in all sampling programs in which they were abundant (D. Stompe, unpublished). In addition, a multivariate analysis indicates that the fishes fall into broad 'natural' groupings that coincide with the assemblages, more or less (Appendix Figure 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 rarely 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 (Figure 3.1). Included species presumably have enough data on their abundance so trend analyses are possible. Chinook salmon and steelhead juveniles are treated separately in Chapter four of this report and are caught mainly in sampling programs designed to catch them. They are short-term residents of pelagic and inshore assemblages so could be added to those assemblages if desired.

### B. Possible Delta fish assemblages

1. *Pelagic species assemblage* (number of surveys in which abundant). Members are typically juveniles or small adults caught together in mid-water trawls in open channels of the Delta. Threadfin shad may be too wide spread to be useful here.

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

2. *Inshore, shallow-water assemblage*. Most fishes are juveniles and seasonal in occurrence; the habitat tends to be breaks in aquatic plant beds, where seines can easily be worked; Mississippi silverside is usually most abundant species; most other species are erratic but regular in occurrence. Splittail and young-of-year Chinook salmon are seasonal users. In the list below "*" identifies freshwater species and italicized text identifies likely resident species.

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

3. *Benthic channel assemblage.* 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 species to this assemblage.

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

4. *Inshore Vegetation assemblage.* 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.

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

**C. Native tailwater fishes assemblage (see text below for description)**

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

The native tailwater fishes assemblage represents a significant departure from assemblages in the Delta. It is the native fish assemblage found in tailwaters below dams but not in the Delta. Many of these fishes are not even captured in the regular sampling programs (heat map data) that focus on the Delta. The assemblage contains some combination of 18 native species depending on the river. The native species often constitute 75-90% of the species and biomass, with the percent decreasing with distance from the cold-water source, reduced velocity and reduced gradient. A typical assemblage would likely include 10-12 native species. In the above list of potential

constituent species, asterisks indicates species that are listed or are species of special concern. The two sturgeon species are 'big river' fish so occur mainly in the Sacramento and Feather rivers, but individuals are seen in the lower reaches of most tributary streams on occasion. 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, and smallmouth bass (Brown 2000). For the Sacramento basin, May and Brown (2002) found four fish community metrics tied to environmental/water quality: (1) percentage of native fish, (2) percentage of intolerant fish, (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.

## 3.5 Discussion: What Does This All Mean for Establishing Metrics?

The following are observations derived from our search for useful metrics to evaluate the effects of altered flows, habitat, and management on fishes of the Delta region and Central Valley.

1. Non-native fishes dominate most general surveys in the freshwater Delta; they are most abundant whether measured by numbers or biomass. Native fishes are 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 to reliably quantify trends in abundance. 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. 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 they did not catch a large number of identifiable species. However, given that they sample early life history stages, their utility as indicators of change should be explored.

5. Juvenile Chinook salmon and steelhead were caught in surveys designed specifically to catch them (e.g., surveys 9, 10). However, the best survey that catches both salmon (including smaller natural origin salmon) and other fishes is the USFWS seining survey (3) and the DWR beach seine survey (17). Perhaps what is the most distinctive feature of Chinook salmon is that most smolts (i.e., larger juveniles) are caught in midwater trawls. Most of the effort to capture juvenile salmon and steelhead is through screw traps set in the main rivers (Chapter four).

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 were not readily available). 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 (perhaps 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 pumping plants in the South Delta that supply water for the Central Valley Project and State Water Project. Their 'sampling' can be described as 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. Trend analysis is possible for many but not all species using existing data if the species has a heat map score of 3 in at least one survey (Figure 3.1).

9. Two non-native species that appear to be good candidates as Delta indicator species are striped bass and American shad. Both are captured in a wide variety of surveys, are still abundant, and have multiple life history stages in the rivers and Delta. Other widespread and abundant non-native species for consideration are threadfin shad and white catfish.

10. Because of their abundance, non-native species are likely to be important indicators of success of restoration projects, especially if combined with environmental indicators. For example, an increase in naturalized, desirable striped bass and American shad could represent an increase in the overall 'health' of the estuary, as determined by features such as higher turbidity. Conversely, an increase in non-native Centrarchids, along with an increase in the extent of aquatic plant beds and an increase in water clarity, would represent a negative trend in estuarine health (although regarded as positive by those who benefit from warm-water fisheries).

11. 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 (Table 2). Other native fishes that might have potential are currently too narrowly distributed or too low in abundance to be useful (e.g. Sacramento hitch, blackfish).

12. All surveys that capture multiple species of fish, mainly as juveniles, should be continued. In addition, 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 Other Metrics

This chapter has largely focused on ways to use abundance measures to determine status of fishes and response to environmental change, which primarily reflects the availability of data. However, there are measures of 'performance' of individual fish that could be useful, if samples are large enough, to see if 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:

- *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 probably growing rapidly. Most of the sampling programs listed in the chapter measure a sample of fish lengths. Measuring weights as well could be time consuming but add considerable additional information to each survey.
- *Growth*. Rapid growth (absolute or comparative) in fishes is often an indicator of optimal 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. For example, larger females 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 growth often requires sacrifice of samples of fish to obtain otoliths. Analysis of otolith geochemistry (stable isotopes), however, can also provide information on where the fish was living at various stages of its life cycle, suggesting potential for being influenced by restoration projects.
- *Histopathological condition.* A histopathology investigation of individual fish can determine factors such as contaminant and starvation stress. It is especially valuable when used in conjunction with otolith growth analyses.
- *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 do 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 Alternative Metrics

The following are possible metrics developed from the information presented above and other sources 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. We assume, however, that multiple metrics would be used for verification.

**A. Delta Indicator Species Metrics**

An indicator fish species metric would include measures of abundance and distribution of the following 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 to change in tandem; the species chosen for the indicator role would be treated individually but discussed together. Delta smelt and longfin smelt have become so rare their inclusion may not be meaningful, unless there is some recovery (unlikely), or they are used just as part of a presence/absence statistic. 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 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 the scale of distribution throughout the Delta, based on a grid (occupancy), although this would be correlated with abundance. The starting assumption would be that each species was once found at all stations, at least seasonally. This could be modeled on a score sheet used for evaluating the current status of all California inland fishes, native and non-native (Moyle et al. 2013).

**B. Delta Fish Assemblages Metric**

Another Delta fish metric would be a variation on the theme expressed in the Delta Indicator Species Metric. It would require using the combined results of all the various surveys on an annual basis. The basic questions being addressed would be: 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 what do those changes mean?

**C. Warm-water Resident Fish Species Metric**

A warm-water fishes metric would inherently be a negative one for native species, on the assumption that abundance and distribution of species associated with dense beds of *Egeria densa* and other aquatic plants or with shallow warm water reflect a Delta that is not favorable 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, 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 two.

### D. Suisun Marsh Nursery Metric

Suisun Marsh is being increasingly recognized as a major nursery area for native and other sensitive species that can rear in brackish water. A Suisun Marsh metric would track capture frequency of young-of-year striped bass, American shad, splittail, starry flounder, and staghorn sculpin, and note catch frequency of delta and longfin smelt as well.

### E. Invasive Species Metric

Ideally, management actions that are specifically designed to improve conditions for native and other desirable species, should make conditions less favorable for recent invasive species that have negative effects on native species. With an invasive species metric, it might be worthwhile to develop a metric that measured trends in the distribution and abundance of a selection of 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 jellyfish (effects don't seem to be major but plankton feeders), Brazilian waterweed, water hyacinth, and Mississippi silverside.

### F. Tailwater Native Fishes Metric

Rivers below dams in the Central Valley typically have the best developed 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 steelhead/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). Water released from dams to benefit estuarine fishes would/should take this into account. An Index of Biotic Integrity also might be appropriate in this situation.

### G. Outflow-Sensitive Species Response Metric

The Bay Institute (Bennett and Rosenfield 2017, draft) analysis proposes that Delta outflow is the key to a healthy ecosystem. 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 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 food was much more abundant than it is now. So, 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)[7].

### H. Index of Biotic Integrity

Indices of Biotic Integrity (IBIs) were proposed by Karr et al. (1986) as a way to evaluate how much the biota of a reach of wadable 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,

[7] 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 doesn't 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 been emptied.

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 1-5 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 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) expanded the use of IBIs 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 they 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) have 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 suggested as an indicator species or as part of an assemblage would have a conceptual or quantitative model associated with it that would include metrics discussed in Section 3.7. 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 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 smelt has declined to such low numbers that matching the results of the model with data is difficult.

*Sacramento splittail* were subject of a conceptual model of their life history dynamics in Moyle et al. (2004). While VSP-type parameters were largely unavailable, enough information was available to state a series of hypotheses that could be tested with further life history studies and, to some extent, a simulation model of population dynamics. While the study emphasized the many uncertainties in splittail life history, enough information on VSP-type parameters 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 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 the young-of-year for 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 (surveys 1, 4, 5). 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 fish. 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 Conclusions

The general method proposed here for evaluating the effects of major management changes in the Delta and associated rivers is to use abundance metrics that rely on data on fishes currently collected by diverse fish surveys (18) that encompass wide areas. An important difference here from current methods is our willingness to use data from both non-listed native species and non-native species, as well as listed species, to evaluate responses to proposed changes in flow regime and habitat restoration. Our contention is that using the responses of all fish species available should expand our ability to detect change, positive or negative, and then relate those responses to listed species or other species of interest.

We list eight possible methods that use Delta and river fishes for determining whether or not biological goals could be reached from flow and habitat improvement projects. In conjunction with the presumably dependent variables of flow and habitat variability, they include measuring, as indicated above: (1) important species abundance trends; (2) the distribution and abundance of a select group of fish species as indicator species; (3) the collective abundance and distribution of common non-native fish species; (4) abundance and condition (growth, etc.) of fishes using Suisun Marsh as a nursery; (5) abundance and distribution of invasive plant, invertebrate, and fish species; (6) a metric that measures response of native fishes to flow releases in tailwaters below dams; (7) responses of selected species of fish and invertebrates to changes in Delta outflow; and (8) an Index of Biotic Integrity. If our general approach is accepted, 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 more rapid assessment of changes than the surveys themselves. For some well-studied species (e.g., striped bass) VSP-type metrics could be used as well, as shown in Chapter four. Finally, using some landscape metrics (e.g., extent of aquatic macrophytes in Delta) simultaneously could provide additional insights into fish responses to a changing environment.

The first approach is tracking important species abundance trends. This is essentially the present method, 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 this method are the increasing rarity of a number of species, so changes in management are not likely to be detectable, and the dependence of salmonid abundance on hatchery production. An improved version of this method would be to use *both* distribution and abundance of a select group of native and non-native species that are 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 by the management actions.

Quantifying the distribution and abundance of select, undesirable non-native fishes (e.g., largemouth bass, Mississippi silverside) could be a metric that is 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 using Suisun Marsh as a nursery would work best in conjunction with other metrics although many of the important Delta fishes rear in the brackish-water Marsh in abundance (e.g. splittail, striped bass, and, historically, delta smelt). This metric 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.

Another alternative to the above eight 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 3.5. 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'), the best approach is likely to be using multiple approaches and metrics. This would be particularly true if some metrics are not sensitive to short-term changes, but useful for measuring long-term changes. Collectively, the fish metrics discussed here will start to tell the story (but not clearly) after 5-10 years at a minimum. Regardless, there will be a continued need for standardized surveys to continue and for new ones to develop that target the important fishes of today.

# 4 SALMON AND STEELHEAD

## 4.1 Introduction

The overall goal of this chapter is to provide recommendations for setting biological goals for populations of Chinook salmon and steelhead. We first describe key metrics for tracking viability of natural-origin populations of salmon and steelhead and their response to major restoration actions (Section 4.2). We emphasize the need to track progress using stock-recruitment relationships that account for density dependence, which is important even in depleted populations (Section 4.2.1). Next, we present a modeling framework for tracking productivity and abundance (Section 4.3.1) and for estimating population responses to major restoration actions (Section 4.3.2). Recommendations for tracking diversity and spatial structure are described (Section 4.4). Last, data requirements and limitations are briefly discussed (Section 4.5). We conclude the chapter with recommendations for setting biological goals (Section 4.6).

## 4.2 Criteria for Evaluating Population Viability and Response to Actions

The viability of salmonid populations 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 we recommend for tracking the status and trends of salmonid populations and their response to restoration actions. Abundance is typically measured as the number of spawners and the number of fry, fingerlings, or smolts. Intrinsic productivity determines the maximum rate of population growth, 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 the stability, resilience and persistence of a population (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., outmigration of fry and yearlings for Chinook salmon). Spatial structure and diversity typically reflect the diversity 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 and reflected by 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.

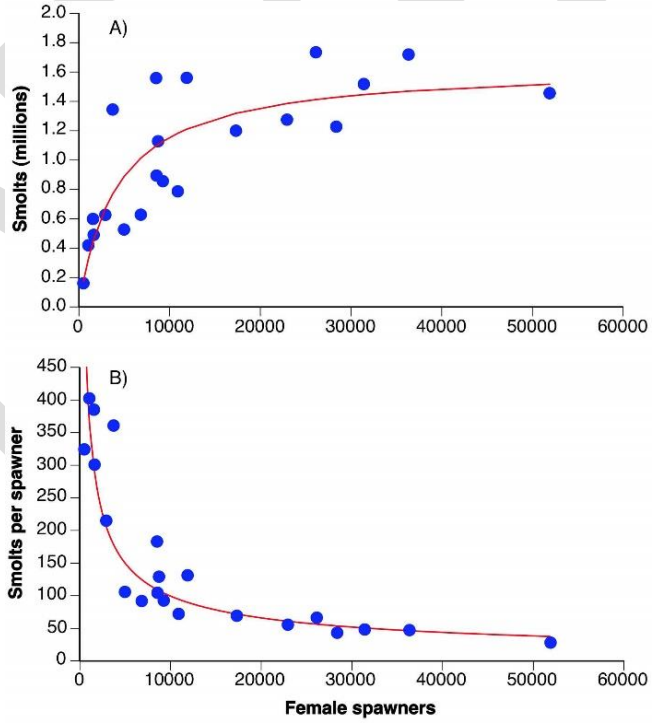
Although spatial structure and diversity are clearly important for maintaining population viability, development of biological goals in the Central Valley should focus primarily on abundance and productivity of populations and secondarily on spatial structure and diversity. Abundance and productivity provide a more direct and immediate measure of population viability. They also provide a more quantifiable means 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. 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 body growth or age at maturation. Abundance and productivity of a population should be evaluated within a framework that incorporates density dependence because there can be strong density dependence, even for depleted ESA-listed species (ISAB 2015; See Box 4.1). Furthermore, compensatory density dependence is critical to the resilience of populations because survival of individuals in the population increases when density is low. Given these findings, there is little support for the arguments made in the Stanislaus recovery report (SEP Group 2016) that density dependence is minimal and can be largely ignored.

**BOX 4.1: Density Dependence Within Depleted Salmon Populations with Supplementation**

Scientists often think density dependence is relatively weak 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% (e.g., 145 smolts/female * 0.5 females/spawner * 0.014 spawner/smolt = 1). 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

In this section we describe how a stock-recruitment 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-recruitment model describes the relationship between the number of parental spawners and the number of recruits they produce. We then describe how the recruitment relationship can be used to test for the effects of flow, habitat restoration, and hatchery salmonids on productivity of the natural population. 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 Section 7.3), are needed to estimate species- and watershed-specific stock-recruitment 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 (pHOS) can be calculated (See Box 4.2). The number of progeny should include total smolts and/or total adults produced by the parent spawners. Total adults should be pre-fishery recruits, 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-recruitment relationships to quantify abundance and productivity

In this section we present some simple mathematical models to articulate how parental abundance and environmental factors such as flow 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 test and quantify the effects of flow and habitat enhancement on parameters that influence status and trends. Finally, these models highlight experimental design and data problems 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 no density dependence in survival rates for simplicity, its influence on the growth of a 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 $\pi$ 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 that are data. 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., $\pi$=R/S), but only if there is no density dependence. In the absence of harvest a population will increase in abundance over time if $\pi$>1 and decline in abundance if $\pi$<1. Larger $\pi$'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 Central Valley 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 $\pi$ 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-recruitment model.

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

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

where $\pi_{trib}$ 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), and $\pi_{deo}$ is the survival rate from this life stage 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 $\pi_{trib}$ from say 50 to 100 smolts/spawner. However, this increase will not result in a positive trend for the population if $\pi_{deo}$ 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,

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

given annual exploitation rate 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). We note that there is little support for the arguments made in the Stanislaus recovery report (SEP Group 2016) that density dependence can be ignored because populations are well below historical abundances (see ISAB 2015 and BOX 4.1). It seems likely that reduced flow and habitat quality may reduce 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 will 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-recruitment relationships is a very common way of incorporating density-dependent effects in fish populations (Hilborn and Waters 1992, Myers 2001), and the Ricker stock-recruitment model is often used to model the production relationship for Chinook salmon (Chinook Technical Committee 1999),

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

where $\alpha$ represents a density-independent rate that depends on the proportion of female spawners, their fecundity, and the survival rate from egg-adult return, and $\beta$ 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 (Fig. 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 $\beta$*S->0, and maximum productivity (intrinsic productivity) in this form of the Ricker model is therefore calculated as $e^{\alpha}$ (the slope of the stock-recruitment curve at the origin). If we assume there is no density dependence in a population then $\beta$=0 and the stock-recruitment relationship is R=S*$e^{\alpha}$, which is equivalent to Eqn. 1 with $\pi$ from Eqn. 1 replaced with $e^{\alpha}$ (i.e., $\alpha$ represents intrinsic productivity $\pi$ in log space). The 1:1 or replacement line commonly shown on stock-recruitment 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-recruitment 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. Greater intrinsic productivity leads to an increasing space between the initial slope of the stock-recruitment curve and the replacement line and indicates a higher potential for population growth or exploitation (Figure 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 observed slope falls further from the replacement line.

A Ricker model and more generally, any stock-recruitment approach, can easily be modified to separate different life stages that may have different sensitivity to flow changes or habitat restoration. 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 3). This particular structure follows the conventional wisdom that the majority of density dependence occurs in fresh water and that effects of population-specific density on survival in estuary and ocean environments is 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 only grow 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.

### 4.3.2 Using stock-recruitment relationships to quantify benefits of flow and habitat actions

Incorporating effects of flow and habitat restoration into stock-recruitment relationships is straightforward and is often done by treating these variables as additive effects on $\alpha$. As an example consider,

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

Here, F is a covariate representing an index of flow at some point during the tributary incubation and/or rearing period, and $\gamma$ represents the coefficient for the flow effect. Stock-recruitment 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 $\alpha$. That is, under this form we expect the log of intrinsic productivity to increase linearly with F (productivity = $e^{\alpha+\gamma*F}$) and since carrying capacity is calculated as $\alpha/\beta$, we expect carrying capacity 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 as well, but we avoid this complication for simplicity here.

This framework provides a statistical approach to quantify benefits of increasing flow or habitat restoration. In the absence of information on tributary 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 during different times of the year affects recruitment. The value of $\gamma$ would quantify the effect of changes in F, with larger positive values indicating a more beneficial effect of higher values of F. An example of this approach is shown in Figure 4, which shows the relationship between egg deposition from steelhead in the Cheakamus River, BC, in the

spring and the resulting size of the fry population in the fall. As expected, fry abundance increases with egg deposition but with diminishing returns due to density dependence (black line). Variation around the stock-recruitment relationship is explained by a statistic quantifying rapid fluctuations in flow during the summer when newly emergent fry are utilizing shoreline areas destabilized by fluctuations largely resulting from flow regulation. The model predicts better than average fry abundance at a given egg deposition in years when fluctuations are lower (red lines pointing upward) and lower than average fry numbers when fluctuations are higher (red lines pointing downward). This covariate model results in a substantive improvement in the amount of variation in fry abundance that is explained, and is a better predictive model based on information criteria.

A number of logical extensions to this model 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[8]) then parameters in,

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

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

can be estimated. This formulation will likely lead to better estimation of $\gamma$ than if estimated using only spawner and adult recruitment data because smolt data allows the procedure to remove variation caused by smolt-adult survival rate. This model could be expanded by using a density-dependent relationship for spawners and resulting fry, and a second density-dependent relationship for fry to smolt stages. However, this model would require robust estimates of the resident fry population as well as the number of smolt migrants, and would require modification to account for potential outmigration of fry. While development of this more complicated version is possible, it would only be possible to apply in very well-studied systems where abundance of all these juvenile stages is well quantified.

As in any regression model, sufficient variation in independent variables (S and F) is required to estimate the coefficients that quantify their effects. Strong variation in F across water year types will probably allow estimation of $\gamma$ even with modest changes in flow within water year types (e.g., dry, above normal, etc.) due to changes in

[8] adult returns prior to fishing means that harvests and spawner counts are included in the adult return values.

unimpaired flows (http://calsport.org/fisheriesblog/?p=2378). Estimates of $\gamma$ can in turn be used to predict the expected increase in productivity and carrying capacity that will result from an increase in unimpaired flows. It may be that $\gamma$ 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 unimpaired flows results in a relatively modest increase in predicted recruitment.

A highly relevant model would include both flow and habitat effects,

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 across years. If both F and H increase by similar amounts over the same time period, they will be strongly co-linear and coefficients $\gamma$ and $\delta$ 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 across different water type years, $\gamma$ and $\delta$ are potentially identifiable depending on the frequency of different water year types. 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 $\kappa$ would quantify the additional benefit of habitat at higher flows compared to lower flows, which may be a sensible prediction in some settings (e.g., Yolo Bypass or unconstrained river reaches where floodplains 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 that has received a habitat enhancement 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 enhanced habitat but also on differences in observed juvenile densities in enhanced and un-enhanced 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 salmonid benefit, whose effect on productivity could be tested by using this metric to represent F or H.

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

Hatchery-origin spawners can make up a large proportion of the total Chinook salmon escapement in many CV tributaries, especially for fall-run Chinook (Figure 5; Palmer-Zwahlen et al. 2018, Willmes et al. 2018). Production relationships predicting recruitment as a function of spawning stock must therefore account for hatchery-origin fish that spawn in the wild. 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 for wild-origin fish. This can be incorporated into a stock recruitment model using,

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

10b) $S' = S*(1-pHOS*\phi)$

where S' is the adjusted number of spawners, pHOS is the proportion of hatchery-origin spawners in the escapement to a tributary, and $\phi$ is the degree to which the productivity of hatchery-origin spawners is impaired (see Box 4.2). In the case when only natural-origin spawners return to a tributary, pHOS=0 and therefore S'=S. If for example 50% of the returning spawners that spawn in the wild are hatchery in origin and these fish have no ability to produce recruitment in the following generation (i.e., $\phi$=1), then S'=S*(1-0.5*1) = S/2. In this simple example only natural-origin spawners are therefore included in the calculation of recruitment.

We recognize that this modeling approach is an over-simplification of wild-hatchery interactions. While pHOS is readily quantifiable through escapement surveys, $\phi$ can only be estimated in systems where the genetic pedigree of all returning spawners and subsequent recruits can be determined. Thus, the value of $\phi$ in the majority or perhaps

all of CV tributaries will have to be assumed by applying values from a limited number of well-studied systems (Box 4.2). However, that application will be highly uncertain because the degree of reproductive impairment of hatchery-produced fish depends on many factors such as previous levels of pHOS, the percentage of natural-origin fish used in the hatchery broodstock (pNOB), and the level of domestication in the hatchery. In addition, "carry-over" effects related to genetic characteristics have also been shown. For example, surviving recruits from hatchery-origin parents that spawned in the wild have been shown to have reduced productivity relative to spawners with no hatchery-origin history (Araki et al. 2009). Ultimately, the history of hatchery management and the level of hatchery fish spawning in the wild will determine the net effect of pHOS on S'.

Quantifying the impact of hatchery-origin spawners on subsequent recruitment will therefore be uncertain. This in turn will make it challenging to evaluate effects of increased flow and habitat enhancement on productivity. For example, if spawning populations which receive flow or habitat enhancement have a considerable hatchery-origin component and do not respond to increases in flow and habitat, it will be difficult to determine whether the lack of response was due to hatchery-origin fish dominating the escapement (pHOS and $\phi$ are large resulting in a small value of S' in equation 10b which limits recruitment) or because flow and habitat enhancement do not have a substantive effect even in the absence of hatchery-origin fish ($\gamma$ and $\delta$ in equation 8 are small values). In the majority of cases we will therefore be testing the combined effect of flow increases, habitat enhancement, and hatchery effects. In our view, this is a major limitation in evaluating potential benefits of flow and habitat enhancement.

While the negative effects of hatchery salmon on wild salmon are well established (Box 4.2), if site-specific quantification is needed, one experimental option would be to exclude hatchery-origin spawners from several tributaries using a weir. This approach would work for winter and spring-run Chinook and steelhead where all hatchery-produced juveniles have the adipose fin clipped but would not work for fall-run Chinook where only 25% of juveniles are clipped. Another option would be to reduce hatchery production for a series of years and compare responses to flow and habitat enhancement during periods with substantively different levels of pHOS. However, such experimental manipulations of pHOS would result in significant social conflicts and also substantively lengthen the period of evaluation.

**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 whether 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. In addition to relative reproductive success studies, 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 impacts or benefits of a hatchery on a naturally spawning population depends 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. 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 (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 substantive 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 allow the response of natural-origin salmonids to management actions to be more accurately accessed (Naish et al. 2008).

## 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)[9]. CA HSRG (2012) recommends that pHOS be <5% when the natural population is not integrated with the hatchery population. For "integrated populations," CA HSRG recommends a PNI that exceeds 0.50, whereas the HSRG in the Pacific Northwest recommends a PNI of 0.67 or higher (HSRG 2014)[10]. Importantly, PNI values represent a means for tracking hatchery practices and adverse effects of hatchery salmonids on the natural population. Tracking pHOS in a watershed serves a dual role because it is also important for evaluating changes in productivity and abundance, as described above.

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

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

[10] Proportionate Natural Influence (PNI) = pNOB/(pNOB + pHOS), where pNOB is the proportion of natural origin spawners in the hatchery broodstock and pHOS is the proportion of hatchery origin spawners in the natural spawning population. For example, a PNI of 0.67 is produced when pNOB = 10% and pHOS = 5%, or when pNOB = 20% and pHOS = 10%. This theoretical approach to hatchery management reduces genetic effects but does not eliminate effects.

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 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.

Rainbow trout (*O. 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 Central Valley 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 the non-anadromous form (Johnson and Lindley 2016).

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 Central Valley 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 human decisions to build dams that block migrating salmonids. 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 tributaries within a watershed. A population with a broader spatial distribution in

a watershed is less vulnerable to catastrophic events and may have less density dependence in survival rates.

## 4.5 Estimating Status, Trends, and Flow-Habitat Construction Effects

Quantifying status, trends, and flow and habitat enhancement effects on Central Valley salmonid populations will depend on the availability and quality of data as well as a variety of estimation issues. This section of the chapter 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) describes the monitoring plan for Chinook salmon spawning escapement in tributaries of the Central Valley. 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, though we have not found any brood tables during our review (see Appendix Section 7.3 for example). We recommend that brood tables be developed for natural-origin salmon. 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 natural 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, hatchery salmon 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 Central Valley 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 Central Valley was 69% (Fig. 5; Palmer-Zwahlen et al. 2018). More comprehensive estimates of pHOS for spring and winter-run populations are needed (Johnson and Lindley 2016).

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 but are likely lower owing to their earlier return timing that reduces exposure to later fisheries (Johnson and Lindley 2016). Ocean harvest of Central Valley steelhead is extremely rare and considered an insignificant source of mortality (Johnson and Lindley 2016).

Although CDFW uses CWTs to estimate hatchery-specific contributions of Chinook salmon to harvests, we are not aware of attempts to allocate ocean and lower mainstem harvests of natural-origin fish to their natal streams (Barnett-Johnson et al. 2007, Kormos et al. 2012, Palmer-Zwahlen et al. 2013, 2018). Genetic stock identification and CWT data, and run reconstruction methods (e.g., English et al. 2007) can be used to allocate harvests of natural origin salmon (by age) to their natal streams in the Central Valley. Genetic assignments will be challenging for fall-run Chinook salmon owing to extensive straying of hatchery-origin fish.

Age composition of returning salmonids is needed to assign harvest and escapement back to the correct brood year so that annual recruitments 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 will therefore not represent the age structure of the latter group. Thus, age composition of natural-origin salmonids needs to be collected every year in both fisheries and on the

spawning grounds. Zabel and Levin (2002) report that recruitment models based on averaged age composition can lead to a substantial overestimate 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 data for natural-origin salmon across all CV tributaries and over time is needed to evaluate the existing data.

Chinook run identification (spring, fall, late-fall, winter-run) is needed to assign Chinook salmon to the appropriate run type. 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 problematic (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).

Tributary outmigrant estimates are required to establish juvenile-based stock-recruitment relationships (e.g., equation 7a). 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 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 only 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-recruitment 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. 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 2008, Newman and Brandes 2010). In the Sacramento River, Perry et al. (2018) found that survival rates of late-fall hatchery-origin juvenile Chinook salmon 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 will not increase survival rates through the southern portion of the Delta; and, 2) very modest increases in flow in the southern Delta from increasing the percentage of unimpaired flows of tributaries are unlikely to improve survival rates to the point where gains in production of juveniles from tributaries will lead to substantive increases in adult recruitment. In the context of our quantitative model described above, $\pi_{deo}$ may be so low that the product of $\pi_{trib}$ and $\pi_{deo}$ 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% (Buchannan 2013), which are considerably higher than values for Chinook over the same area. The higher survival rate is thought to be due to the larger size of steelhead. The value of applying survival rates of tagged hatchery steelhead to natural-origin steelhead 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. As noted above, marking 100% rather than 25% of fall-run hatchery Chinook salmon would improve estimates of natural-origin salmon abundance and trends.

Sample size is an important factor to consider in estimation of fish responses to flow and habitat enhancement. For example, the number of years where escapement or smolt production is measured before and after an increase in flow will have a large effect on the precision of the estimated flow effect. Bradford et al. (2005) found that monitoring had to be conducted over a period of 4-6 generations (20-36 years for a Chinook population with a maximum age-at-return of 5 years) given a substantive 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-recruitment 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-recruitment 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 larger variation in flow among water year types.

Thus, even if flow changes due to an increase in unimpaired 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 unimpaired 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.

The situation for 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 our ability to accurately estimate this abundance (given uncertainty in pHOS and the lower reproductive ability of hatchery-origin fish). In our view 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.

<u>Statistical Approaches for modeling stock-recruitment data</u> are well established (Hilborn and Walters 1992). In the modeling component of this chapter we have described how to include covariates such as flow and habitat directly into a stock-recruitment 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 uncertainty in the input data and are less prone to bias resulting from serial correlation in input data that can affect traditional stock-recruitment 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 flow and habitat covariate effects. This latter feature will be useful if flow and habitat effects are small.

## 4.6 Recommendations for Setting Biological Goals

Viable Salmon Population criteria (abundance, productivity, diversity, and spatial structure) are well-established metrics for evaluating populations. We recommend that VSPs form the basis of biological goals for Central Valley 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.

Our recommended framework for evaluating abundance and productivity accounts for density dependence in survival rates by using a stock-recruitment relationship (Sections 1.3, 1.4). This approach is needed because density dependence occurs even at low salmon densities, and compensatory density dependence provides resilience and stability during periods of declining abundance. This framework, including the approach to quantify benefits of increased flow and habitat, 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 flow and habitat actions.

Intrinsic (maximum) productivity can be estimated from spawner-to-smolt or from spawner-to-adult stock-recruitment models; the former reflects conditions solely within the natal watershed, while the latter is also influenced by conditions in the Delta and ocean. An obvious goal for intrinsic productivity is that adult recruits/spawner must exceed one, which represents the lower limit for a viable population. Another biological goal for intrinsic productivity during the spawner-to-smolt stage could be based on the value needed to produce a viable population after considering survival during the smolt to adult stage. Trends in intrinsic productivity can be evaluated over time using a state-space approach to determine whether conditions are improving (Peterman et al. 2003, Peterman and Dorner 2012). Lastly, annual estimates of R/S, which does not explicitly consider density dependence, can be examined over time to evaluate whether a population heavily supplemented with hatchery spawners is sustainable if hatchery fish are removed.

Key abundance metrics include the number of adults (or progeny) produced by spawning parents and the number of spawners that leads to maximum production of

juveniles and/or future adults. The latter value can be derived from the stock-recruitment relationship.

Two of the most relevant diversity metrics include the pHOS and age composition of the returning natural-origin salmon. These metrics are also needed to reconstruct salmon recruitment to the watershed so that stock-recruitment parameters can be estimated. A reduction in pHOS is a sensible biological goal if the objective is to maximize productivity of the naturally-spawning population. A biological goal for age composition might be desirable considering that 1) diversity in age reduces risk to catastrophic events, 2) more old/large Chinook are produced from natural-origin compared with hatchery-origin salmon, and 3) an increase in natural-origin subyearling Chinook migrants might reflect improvement of shallow water habitats in the Delta.

We also recommend a covariate stock-recruitment estimation approach for quantifying benefits for salmon and steelhead associated with flow, habitat improvements/restoration, and changes in pHOS. Ideally, this approach would use data collected before and after restoration actions, and possibly use data from watersheds without actions that could serve as a control for environmental change. Water year type may provide necessary contrast to quantify fish responses to flow actions.

Monitoring of salmon in the Central Valley has improved in recent years, but key metrics such as those described above 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 to support stock-recruitment analyses. When data are deficient, assumptions may be used. However, these assumptions and lack of data will lead to greater uncertainty in estimates of the status and trends of the population, and uncertainty in estimates of the potential benefit of restoration actions.

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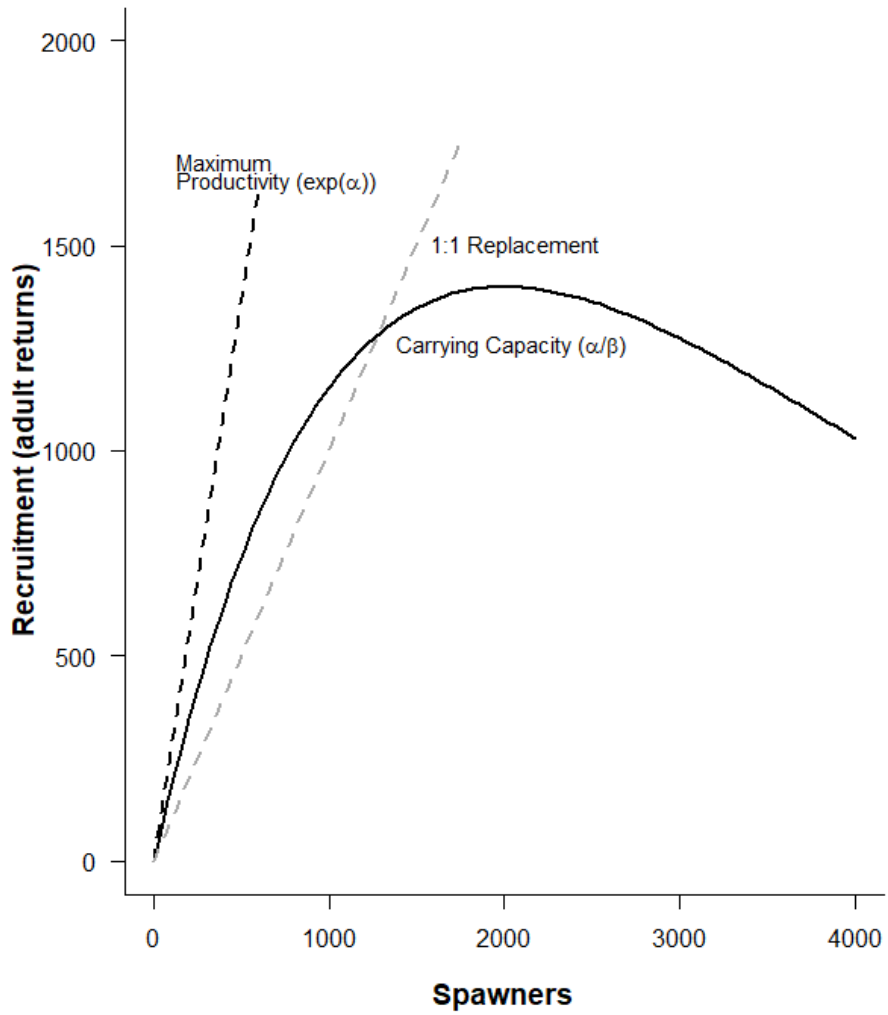


Figure 4.1. Example of a stock (escapement)-recruitment relationship (thick black curved line). The initial slope of the curve (dashed black line) represents maximum (intrinsic) productivity and is calculated as $\mathbf{e}^{a}$. The intersection of the stock-recruitment curve and the 1:1 replacement line (dashed grey line) represents the carrying capacity of the population (escapement = recruitment) in the absence of harvest and is computed as $\boldsymbol{\alpha/\beta}$.

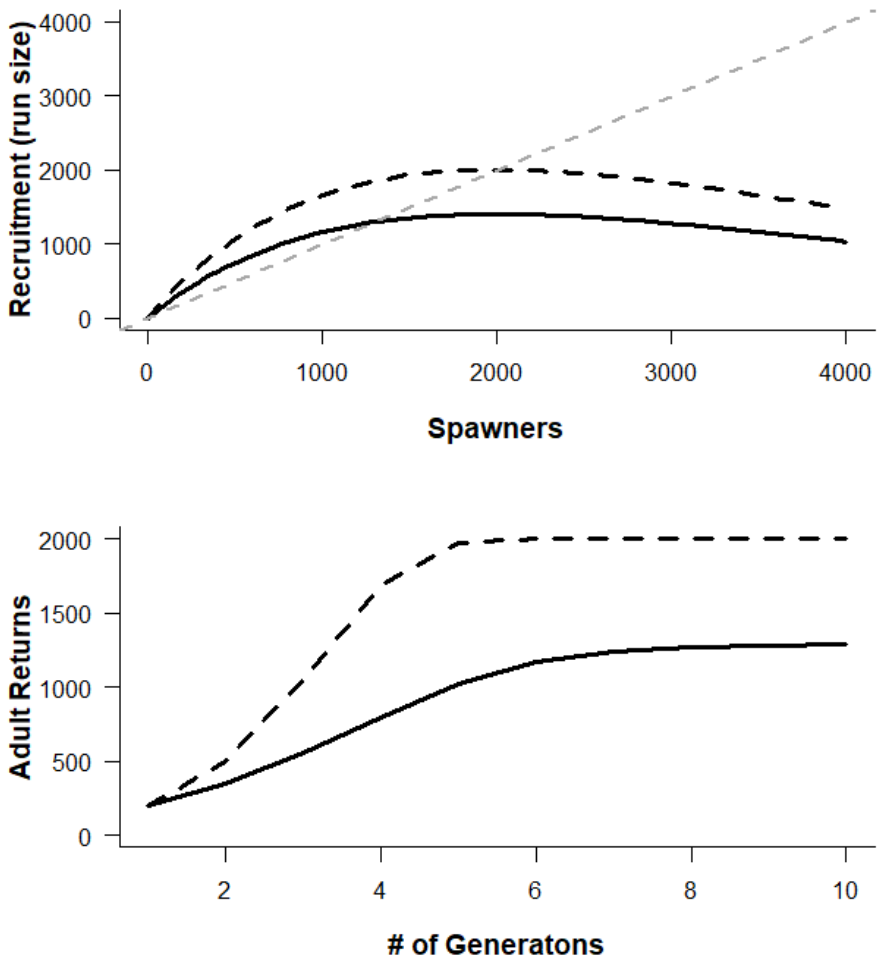


Figure 4.2. Comparison of two stock-recruitment curves (top panel) where the intrinsic productivity for the base relationship (solid line) is increased by 40% (dashed line), and the resulting increase in abundance over 10 generations beginning with an escapement of 200 fish (bottom panel). This figure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

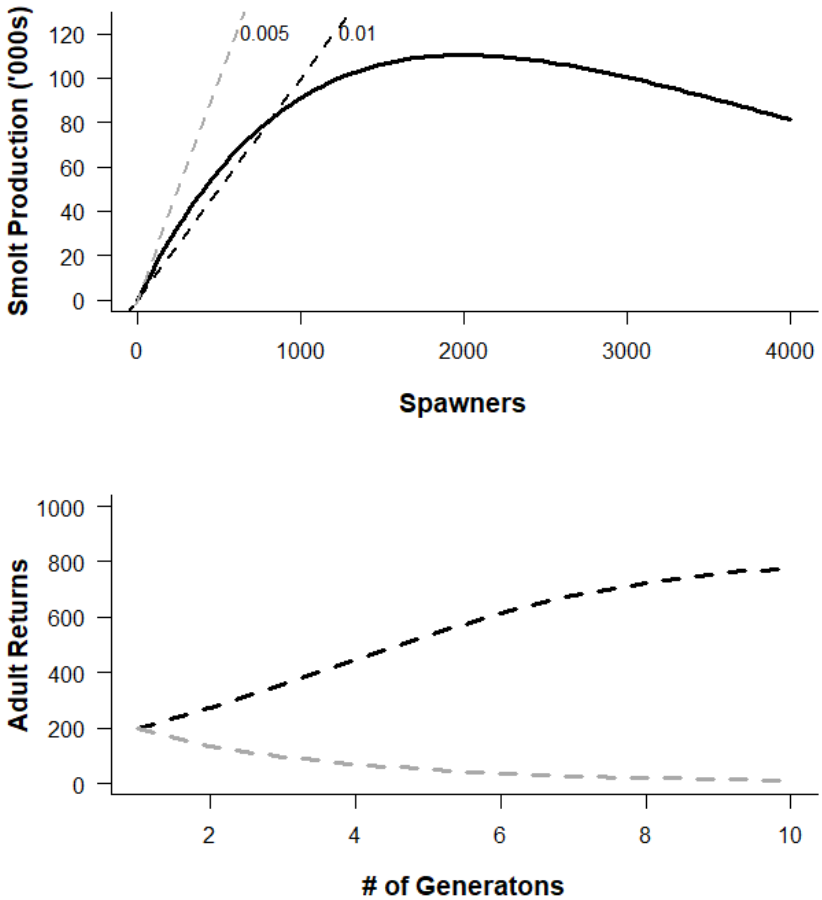


Figure 4.3. An example of a relationship between parental spawner abundance (escapement) and the resulting number of smolts produced (top), and predicted abundance trend of adult returns that depends on this recruitment curve and low and high smolt-adult survival (bottom). The freshwater stock-recruitment 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.

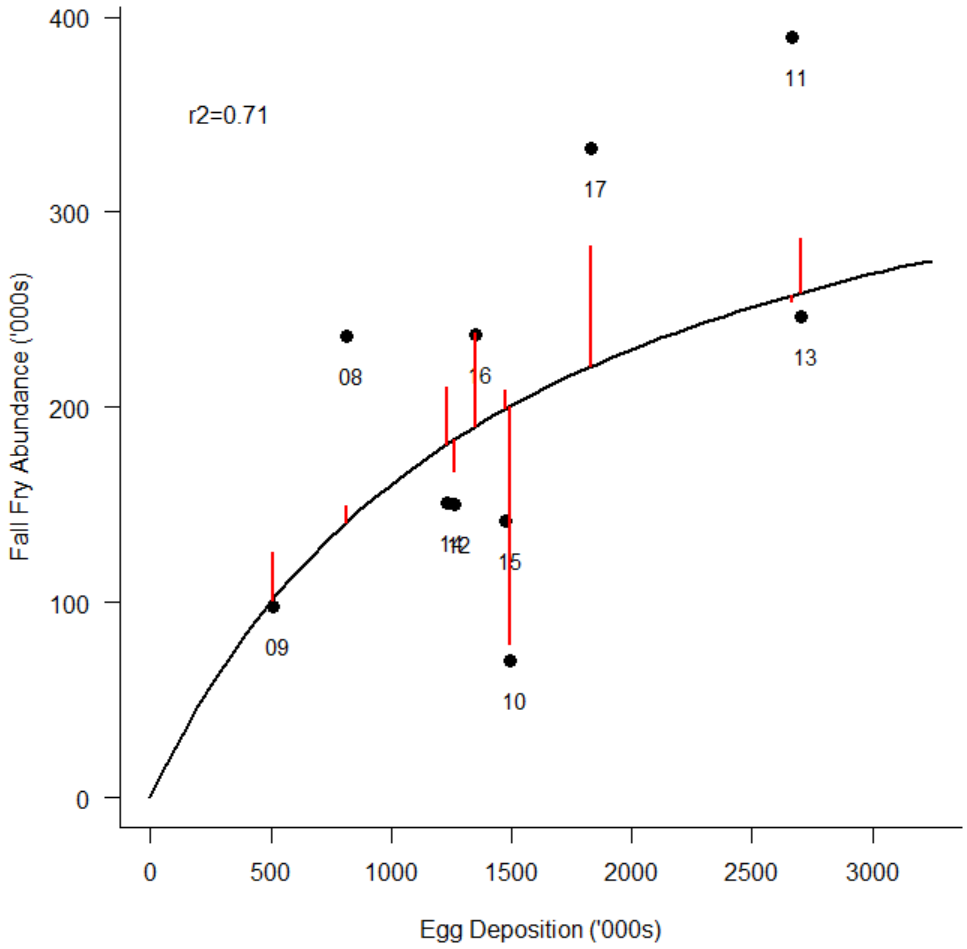


Figure 4.4. The relationship between steelhead trout egg deposition in the Cheakamus River, BC in spring, and the resulting number of fry in the fall. The black line represents the estimated average stock-recruitment relationship, while the red lines show year-specific predictions based on a statistic quantifying the extent of fluctuations in flow over the summer when newly emergent fry are utilizing immediate shoreline areas. Positive values (red lines extending upwards from black line) show years when there are lower fluctuations during the summer. The text beside each point denotes the year of spawning and fry capture.

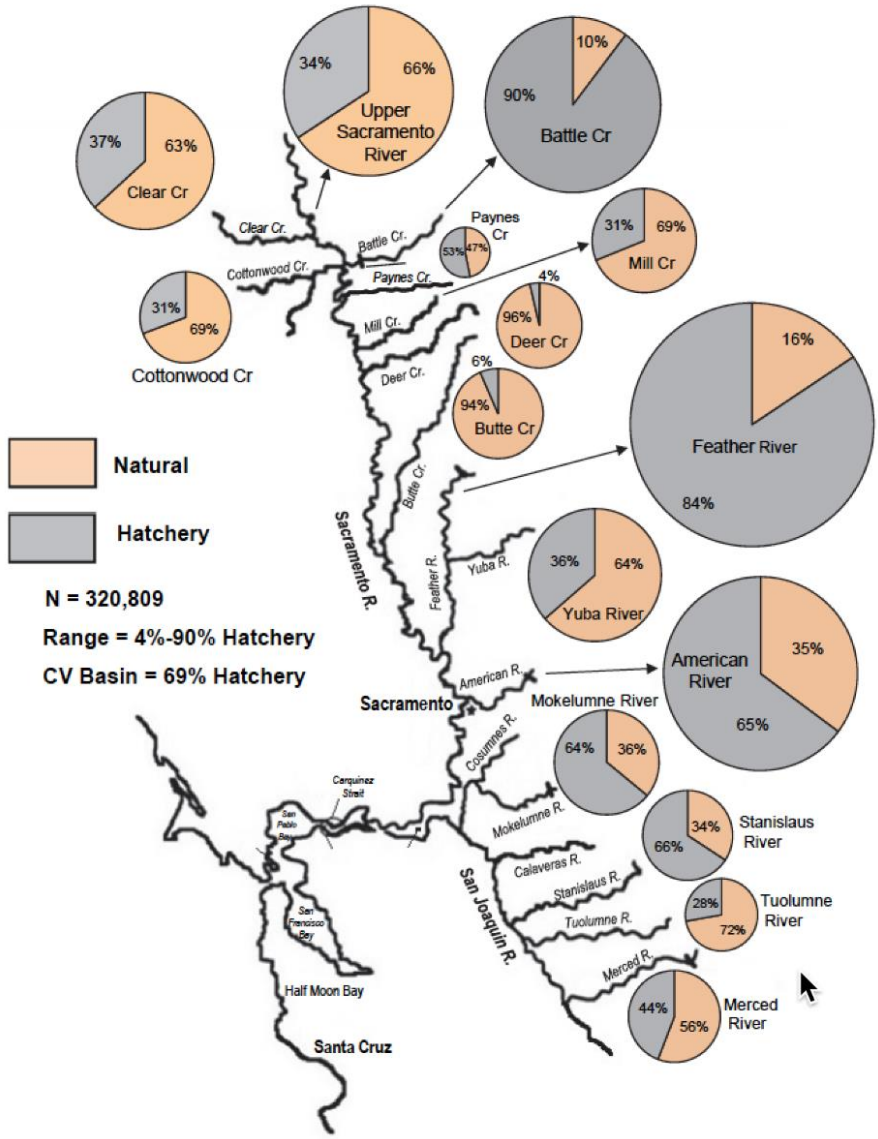


Figure 4.5. Proportion of natural and hatchery origin fall Chinook salmon on Central Valley spawning areas, 2013. Source: Palmer-Zwahlen et al. (2018).

# 5 CONCLUSIONS

The Charge to the Panel was to develop scientifically defensible methods for formulating biological goals for narrative objectives in the Bay-Delta Plan. We focus on methods for determining ecological responses to management actions including manipulations of flow and other actions such as habitat restoration. The principal focus of our analysis was therefore on monitoring and evaluation based on selected metrics that would be most useful for assessing responses of species of concern and identifying structural components and processes underlying those responses. The biological goals were formulated so they could be assessed with current monitoring programs and existing data; however, realizing the inadequacy of monitoring for some geographic areas and specific topics, we also considered additional methodologies that could detect expected variability and that have proven to be valuable in other monitoring and evaluation programs.

Our work was divided into three main topics: salmonids, other fishes, and ecosystem structure and function. We discuss these in reverse order so that each chapter builds on the previous chapter.

Chapter two looks at structural and functional components of the aquatic ecosystems of the rivers and the estuary, emphasizing components that support native fishes by providing habitat elements such as physical configuration, water-quality attributes, food, and shelter from predators. The chapter distinguishes between monitoring and evaluation programs focused on river ecosystems and those focused on the estuary, because of the importance in the estuary of tidal oscillations and the response of salinity to variations in freshwater flow, and the much longer and richer data sets available for the estuary than the rivers. Several structural ecosystem elements would provide essential underpinning to analyses of fish responses: these include physical (e.g., temperature, flow, and conductivity or salinity) and chemical properties (e.g., nutrients, dissolved oxygen), as well as biotic structural components including the main primary producers (algae and plants), the microbial community, and aquatic invertebrates. Few if any functional components are monitored, though information on some would be helpful in understanding responses of fish and thereby refining actions. Measurements in the rivers should include informative bioassays using attached algae and assessments of aquatic insects. In the estuary, phytoplankton primary production can be estimated

from existing monitoring programs, and secondary production of zooplankton can also be roughly estimated. We emphasize, though, that little monitoring of the kind needed for these assessments now occurs in the rivers, and even in the estuary some important components remain unmonitored, such as the vegetated margins of the estuary. These gaps should be filled as needed to support the overall focus on salmonids and other native fishes.

Chapter three proposes a general methodology for evaluating management changes on the abundance of fishes in the rivers and upper estuary using metrics collected by diverse agency surveys throughout the region. The list of native fish species provided by the Board included several whose abundance is so low that no amount of monitoring will reveal their responses to management actions. Therefore, we strongly recommend expanding this list to include other, unlisted native fish species and non-native fish species to evaluate responses to changes in flow regime and habitat restoration. Expanding the array of fish species will enhance the ability to detect responses, to the extent that fishes respond similarly. We discuss eight possible methods for fish assessment that provide alternative views of the fishes' responses. A selected group of fish species should be designated as indicator species, either individually or as assemblages of species with similar requirements. Environmental and other non-fish metrics, as noted in Chapter two, are needed 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, such as those for salmon, are available or could be developed for three species (striped bass, delta smelt, splittail) that can help managers predict effects of change; similar models should be developed for other selected species as well.

Chapter four provides recommendations for setting biological goals for salmon and steelhead in the rivers and estuary while also providing a framework for quantifying responses to flow and habitat restoration. Viable salmon population criteria (abundance, productivity, diversity, and spatial structure) are well-established, and biological goals should be derived from these criteria, especially productivity and abundance. We recommend a framework for evaluating salmonid abundance and productivity that incorporates density dependence in a stock-recruitment relationship. Two critical metrics are the proportion of hatchery-origin spawners in the wild and the age composition of returning natural-origin salmon. We identify an estimation approach for quantifying responses of salmon and steelhead to changes in flow, habitat, and the

proportion of hatchery-origin spawners. Key metrics should be consistently measured each year in every watershed of interest. Improvements in monitoring and evaluation programs are needed to reduce uncertainty associated with evaluating progress towards biological goals for natural origin salmonids, including goals related to flow and habitat actions. A definitive assessment of the effects of flow and habitat improvements on a salmonid population would likely take at least a few decades of intensive data collection, whereas responses of specific life stages (e.g., egg-fry survival rate) can be determined over shorter time frames.

Ultimately, goals for populations of Chinook salmon, steelhead, other fishes, and for the ecosystem should be integrated so as to quantitatively evaluate current conditions, future trends, and the potential benefits of management actions. This evaluation will be most effective if the actions are implemented and managed as experiments, with clear identification of the key unknowns and a design to resolve them, and a responsive feedback process for changing or refining goals, models, and assessment. Success will be dependent on well-designed quantitative goals, well-implemented monitoring programs that inform progress towards those goals, support for data management and analyses, adaptation to new information, and sustained funding for long-term assessment that will enable distinction between environmental variability and responses to management actions.

DRAFT

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# 7 APPENDICES

## 7.1 Charge Questions and Brief Answers

*Ecosystem Structure and Function*

The materials found in the Ecosystem Structure and Function chapter are designed to answer most of these charge questions. Specific questions, however, might require some additional attention. A short answer is provided for each of the modified charge questions for this chapter on ecosystem structure and function.

> **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, and the Bay and Delta are much better monitored than the tributary rivers. The Panel recognizes that there are good arguments for adding more functional measurements to assessment metrics (e.g. primary production, ecosystem respiration, secondary production, decomposition rates, nutrient uptake rates, predation rates and patterns, etc.), but the data base to which to compare is 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 review and synthesis 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 included organic matter decomposition, nutrient cycling, metabolism, pollutant dynamics, and community dynamics. The four categories listed in the charge question are very broad, but 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.

> **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 would of necessity change as the physical configuration, flow dynamics, salinity, and community structure changes.

> **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 what gaps are identified in ongoing monitoring in the Delta and in the tributary rivers. In addition, monitoring should be evaluated from the perspective of bringing together monitoring efforts for 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 community. Structural elements such as temperature, dissolved oxygen, turbidity, salinity (conductivity), and toxic constituents to fish should be monitored. Other structural components such as algal community structure, aquatic invertebrate structure, and fish community structure are common monitoring constituents in well-regarded monitoring and evaluation

programs (see the box on the Healthy Waterways Initiative Program in Australia). 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 is 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.

> **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 question with limited boundaries or constraints on the individual questions. Climate change alone opens up so many pathways that most certainly 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 open up an infinite number of potential interactions. One 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 characterize the extent and duration of flooding and model flow fields on the flooded landscape.

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### *Native Fishes*

The following are questions posed in the Charge to the Panel that we address in the native fishes and fish assemblage chapter (Chapter three). Brief answers to each question are also provided, to direct readers to appropriate parts 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 most 'other fishes' analyses. 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 distribution and abundance as indicated in diverse surveys of Delta and riverine fishes (see "Surveys for fishes").

> **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 this section is devoted to answering this question but see "Other Metrics" in this chapter.

> **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?

Yes, this is discussed in "Discussion: what does this mean for establishing metrics?"

> **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, but we determined that the broad term "indicator species" was more applicable. Umbrella species typically require large blocks of habitat to persist, so their conservation results in non-target species being conserved at the same time, in a subset of habitats. By indicator species

we mean 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 occupy the same general area. In this case our main concern is for Delta indicator species, 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.

> **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?

We provide some additional possibilities in "Other Metrics".

> **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 covered in "Other Metrics".

> **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 of resident, non-pelagic fishes in the Delta is needed and that 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, making 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.

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### *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 we recommend evaluating VSP within a density dependence framework, as this was not adequately addressed in Sep Group (2016) (Section 4.2, 4.3, 4.4). Separate biological goals should be created for each VSP parameter.

> **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 noted in the charge should be monitored as a means to fully evaluate population dynamics of salmon and steelhead populations, but we recommend several metrics that should be the focus for establishing biological goals and for evaluating progress towards those goals (Sections 4.2, 4.3, 4.5.1). 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 as they also reflect diversity and spatial structure (Section 4.2). Biological goals should also be established for the proportion of hatchery-origin spawners in the population (pHOS), as this diversity metric has critical implications for evaluating abundance and productivity of natural populations (Sections 4.5.1, 4.8.1).

> **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?

We recommend an approach that specifically tests for the impact of flow and habitat actions on population productivity and abundance (Section 4.5). This approach accounts for variation in discharge due to increases in both the percentage of unimpaired flows and water year type. The latter variation will likely provide more contrast which will help determine the population-level response to flow.

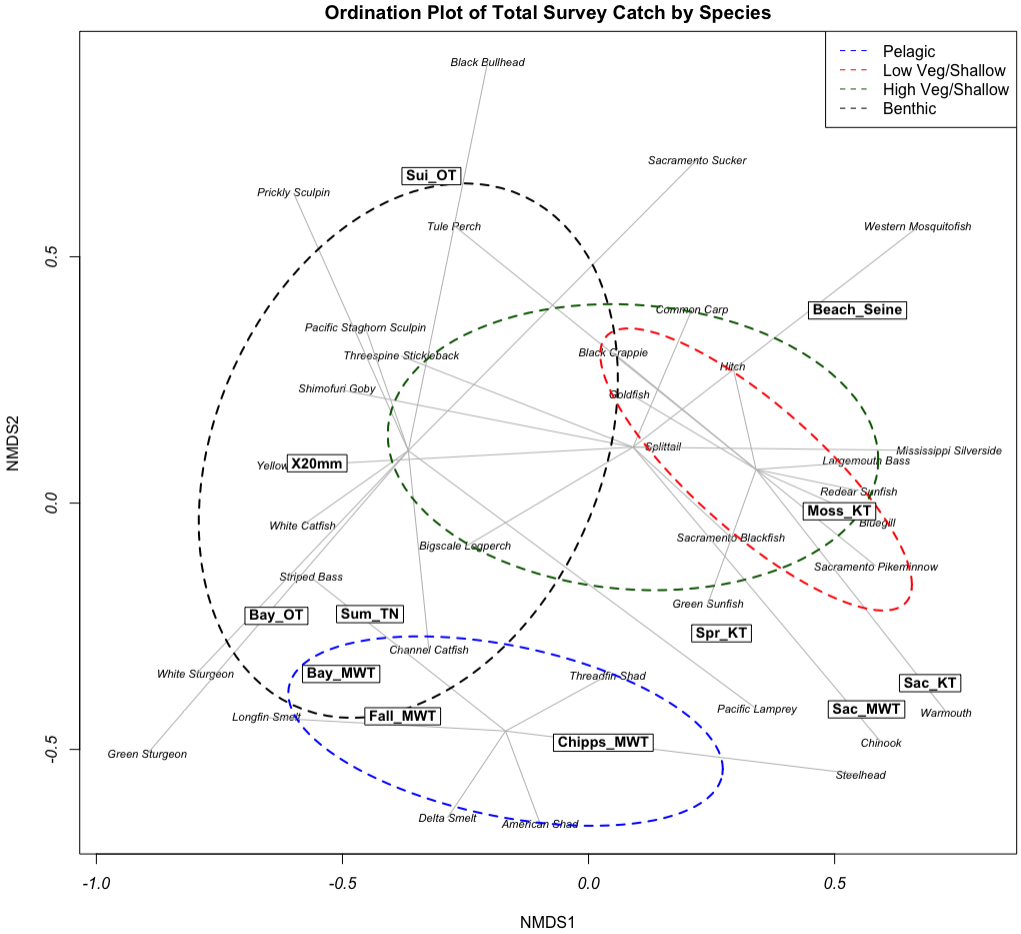
**Charge Question 4.** Under the existing network of monitoring programs, what site-specific and system-wide monitoring methods does the Panel recommend using to assess achievement of the biological goals, including each VSP parameter? For example, could rotary screw traps be used to effectively monitor 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 we 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. We strongly recommend accurate estimates of the spawning population, along with pHOS, as spawning escapement is the basis for constructing "brood tables" as discussed in Section 4.3 and used in the modeling described in Section 4.4.

**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 1.5.2). The proposed stock-recruitment analysis will likely require 15-20 years of data to provide moderately reliable inferences of effects of flow and habitat and abundance and productivity (Bradford et al. 2005). We recommend the framework described by Lindley et al. (2007) for evaluating extinction risk.

## 7.2 Native Fishes and Fish Assemblages




Appendix Figure 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 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 |

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|---|---|---|---|---|---|---|---|---|---|
| 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 |
| **Survey** | **Agency** | **Years of Sampling** | **Timeframe** | **Method** | **Location(s)** | **Number of Stations** | **Sampling Intensity** | **Survey Purpose** | **Area Sampled** |
| 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 |

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|---|---|---|---|---|---|---|---|---|---|
| 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.

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Appendix Table 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, *A. 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 |
| Sac. Hitch, *Lavinia exilicauda* | Yes | SSC (3.1) | 1 | 0 | 2? | RA |
| Sac 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 |
| Sac. 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, *A. 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 |

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|---|---|---|---|---|---|---|
| Chinook salmon (fall run), *O. 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, *L. microlophus* | No 50+ | None | 3 | +1 | 1 | AB |
| Green sunfish, *L. cyanellus* | No 50+ | None | 3 | +2 | 1 | CO |
| Largemouth bass, *Micropterus salmoides* | No 50+ | None | 4? | +1 | 1 | AB |
| Spotted bass, *M. punctulatus* | No 50+ | None | 3 | +1 | 1 | CO |
| Warmouth, *L. 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 |

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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). See Appendix Table 2-4.
    -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)

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**Appendix Table 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 | Floodplain | Within Delta, | Benthic | Low |
| Goldfish | SFW | Warm | M | 3 | Delta FW | None? | Veg | Low |
| Hitch | SFW | Cool | M | 3 | Delta | Up-river | Pelagic | Low |
| Sac blackfish | SSW | Warm | M | 3 | Delta | None? | Pelagic | Low |
| Sac pikeminnow | SFW | Cool | L | 4 | River | Up-river | Pel (ad) Edg (j) | Mod |
| Splittail | EFW | Cool | M | 3 | Floodplain | Up-river | Benthic | High |
| Sac 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 |

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|---|---|---|---|---|---|---|---|---|
| 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 |

**Explanation of columns**

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

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Appendix Table 3.4. Scores and categories used to assess the vulnerability of California's freshwater fish fauna to climate change (From Moyle et al. 2013).

| **Category** | **Score** | **Description** |
|---|---|---|
| -2 Critically vulnerable | <17 | The species is extremely likely to be driven to extinction by year 2100 without conservation measures |
| -1 Highly vulnerable | 17-22 | The species is on the path towards extinction as the result of climate change |
| 0. Less vulnerable | 23-37 | The species is likely to decline or become more limited in distribution but extinction unlikely by 2100 |
| +1 Least vulnerable | 28-32 | The species is likely to be relatively unaffected by climate change, with range and populations remaining stable |
| +2 Likely to benefit from climate change | >32 | The species is likely to increase in range and abundance as the result of climate change |

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## 7.3 Salmon and Steelhead (example brood table)

Example of a brood table constructed from adult Kvichak River sockeye salmon in Bristol Bay Alaska, 1962-2017. Age 1.2 refers to 1 winter in fresh water and two winters at sea before returning to spawn. Returning progeny include 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 |