Dutch Slough
Adaptive Management
and
Monitoring Plan

Version 2 ---- June 2016

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Figures

Introduction

The Dutch Slough Tidal Marsh Restoration Project will restore tidal marsh, riparian woodland, and other habitats to an 1178 acre site consisting of three parcels south of Dutch Slough and east of Marsh Creek, in northeastern Contra Costa County. The Natural Heritage Institute spearheaded an evaluation of the ecological values to be expected from restoring the site. The location of the site in the western Delta suggests that a number of species of concern could benefit by restoration of tidal wetland ecosystems, including all runs of Central Valley Chinook Salmon and Steelhead, Sacramento Splittail, and Delta and Longfin Smelt. Occasional sightings of both outmigrating salmon smolts and immigrating salmon adults suggest that Marsh Creek outflows attracted migratory fish to the area. Unlike most of the other possible restoration sites in the Delta, Dutch Slough is not deeply subsided and so could be feasibly restored to t tidal wetland. Thus, the consensus of biologists who were consulted was that the site had large potential benefit to aquatic species. This plan is largely focused on the benefits to fish species of concern.

In 2003, with funds from CALFED's Ecosystem Restoration Program and the State Coastal Conservancy's San Francisco Bay Area Program, the Department of Water Resources purchased the Dutch Slough restoration site, comprised of Emerson, Gilbert and Burroughs parcels. A technical advisory panel was formed to develop a restoration plan that was presented in the Conceptual Plan and Feasibility Report (PWA 2006). In addition, an Adaptive Management Plan (Cain 2008), an EIR (DWR 2010) and a supplemental EIR (DWR 2014) have been completed. For further details of the site and physical plans please see the supplemental EIR available at http://water.ca.gov/floodsafe/fessro/environmental/dee/dutchslough/index.cfm

The two primary objectives of the Dutch Slough project are to provide habitat for native species, and contribute to scientific understanding of habitat restoration in the Delta. To meet these objectives, an Adaptive Management Working Group (Group) was convened in 2004 to develop an Adaptive Management Plan (Plan) for the project. The Plan's primary purpose was to assess the ecological effectiveness of the restoration, and to pose hypotheses about marsh restoration that could be tested at the project site, the results of which would inform the design or implementation of future Delta tidal marsh restorations. For this project, because of the difficulty in changing the project's landscape after the introduction of tidal influence, use of the term 'adaptive management' does not mean that experimental results would lead to changes in the project's physical structure. The knowledge gained at the Dutch Slough site will be used to "adaptively manage" the design, structure, and implementation of subsequent projects in other locations in the Delta.

Work to restore the site is expected to begin in 2016. However, the restoration design has changed substantially from the design upon which the original Adaptive Management Plan was based. In addition, changes in the Delta environment, changes in our scientific knowledge of the Delta ecosystem, and changes in the regulatory and management environment will affect the implementation and management of the project to achieve its objectives. This plan addresses these changes and updates the original 2008 Adaptive Management Plan.

Species of interest for Dutch Slough

Located near the confluence of the Sacramento and San Joaquin rivers Dutch Slough is along the migratory corridor for all the migratory fish species of the Central Valley, including Chinook Salmon, Steelhead Trout, Green and White Sturgeon, and several species of lamprey. Many resident native fish species of concern are found in nearby waterways. The site is near populations of numerous endemic and unique plants, reptiles, amphibians, mammals and birds. The first task of restoration planning was to identify which species the site was most likely to benefit if properly restored.

Young salmon and Steelhead. All Chinook Salmon and Steelhead runs (hereafter 'salmon') could reasonably be expected on the restored site. Flows from Marsh Creek provide one of the few places where tributary flows could reasonably be expected to attract outmigrant smolts into an area for rearing before they enter salt water immediately to the west. In San Francisco Bay outmigrating salmonids have been found to move rapidly to the ocean once they leave freshwater. Because the Steelhead, Winter-run Chinook Salmon, and Spring-run Chinook Salmon are listed as threatened or endangered, intensive sampling of wild populations on the site must be restricted. Use of hatchery fish might help establish the value of Dutch Slough restoration for salmonids. For a review of the science and conceptual models underlying the adaptive management of Dutch Slough for salmon, see the draft Conceptual Model in Appendix 2.

Sacramento Splittail. Although not listed under state or federal endangered species acts, Sacramento splittail (hereafter 'splittail') have been a species of concern for several decades (Moyle et al. 2015). In wetter springs, splitttail spawn abundantly in the Yolo Bypass and the genetically distinct population centered on the Napa River overlaps extensively with the Central Valley population. Thus, Dutch Slough is situated where it is likely to receive large numbers of young splittail, of both genetic strains, in wetter years. Since splittail are not listed they can be monitored with sufficient intensity to test hypotheses from field data. In wetter years splittail abundance is often very high due to spawning in the bypass; in drier years spawning is much more limited but the nature and location of Dutch Slough may promote splittail spawning on site. For a review of the science and conceptual models regarding splittail at Dutch Slough see the 2008 DRERIP model in Appendix 2. Recent work shows that splittail consists of two genetic strains that overlap greatly in the western Delta, near Dutch Slough, but the distribution of each moves with the level of spring outflow (Feyrer et al. 2015). Thus, the location of Dutch Slough lends the site to a variety of in-depth work on splittail's natural history, population biology, ecology and genetic dynamics.

Delta Smelt. The lack of open water habitat limits expectations that Delta Smelt will occur on site. Productivity exported from the site might benefit nearby smelt populations in summer and fall months, by augmenting the trophic base in their nearby pelagic habitat. Flux of organic material, especially food organisms of Delta Smelt, can be readily measured and are included in this monitoring plan. However, directly documenting benefits to Delta Smelt in nearby large channels would require substantial off-site monitoring of a threatened species and is not included in this monitoring plan. This plan focuses on monitoring the restoration site and attempts to minimize any impacts on listed species.

Delta Smelt and Longfin Smelt may spawn on the site and larvae may occur, but there is no reason to expect them to be more than occasional visitors. However, if larval sampling indicates that the site has value as a smelt spawning ground, the need for additional monitoring or reduction of impacts of monitoring efforts on smelt recruitment would be evaluated.

Black Rail. Avian monitoring after 2008 added another listed species that was expected to benefit directly from the Dutch Slough restoration project, the California Black Rail. This plan addresses only aquatic species, but Black Rails are currently present in existing wetlands on the project site. The design of the project was modified to ensure that suitable habitats remained available to Black Rails. An avian monitoring program including Black Rails, at least in the first few years, would be valuable to inform future species management.

Project Objectives

The overall project objectives are:

1. Benefit native species by re-establishing natural ecological process and habitats.
2. Contribute to scientific understanding of functional ecological restoration by implementing the project under an adaptive management framework
3. Provide shoreline access and educational and recreational opportunities.

The 2008 Adaptive Management Plan for Dutch Slough aimed to offer early data on the functioning of restored wetlands in order to guide later restoration efforts. At that time, it was envisioned that all three Dutch Slough parcels would be restored at the same time, and that it would be restored years before other planned projects. Now Dutch Slough will be restored one parcel at a time, and it is one of many projects being pursued in parallel. Dutch Slough is no longer far enough ahead of other restoration projects work to influence their initial designs, but the ecological questions it is designed to address are still pertinent and can be compared with results from the simultaneous efforts and will likely inform future projects. Because the targeted eventual scale of Delta restoration has increased substantially since 2008, answering the questions outlined in this document will affect the design of much more future work.

The specific objectives of restoration and adaptive management at Dutch Slough have not greatly changed since the 2008 plan was published.
The restoration objectives, then and now, are:

- Reestablish the hydrologic, geomorphic, and ecological processes necessary for the long-term sustainability of native habitats and the plant and animal communities that depend upon them.
- Restore a mosaic of wetland and upland habitats.
- Contribute to the recovery of endangered and other at-risk species and native biotic communities.
- Minimize establishment of and reduce impacts from non-native invasive species.

The adaptive management objectives are:

- Generate information that will guide the design and effectiveness of future wetland restoration projects in the Delta.
- Generate information regarding the ecological function of different types and sizes of freshwater tidal marsh habitats and their value to native fish species, particularly splittail and juvenile salmon.
- Generate information regarding the processes that control the production and dispersal of both methylmercury and dissolved organic carbon in different types of wetlands. [Methyl mercury issues are generally a public health issue and monitoring plans are being developed separately by the Regional Water Quality Control Board].
- Provide the opportunity to establish field scale research projects at Dutch Slough to measure ecological processes and test the efficacy of management interventions for a variety of reasons including exotic species control, avian habitat enhancement, wetland species restoration, control of mercury methylation and subsidence reversal.

The measurable, ecological outputs or "performance measures" guiding design of the Dutch Slough experimental adaptive management project are more specific:

- Provide suitable habitat for juvenile salmon rearing (growth and survival)
- Provide suitable habitat for splittail spawning and rearing (reproduction, growth, and survival)
- Augment food production for pelagic organisms (phyto- and zoo-plankton abundance)

The Dutch Slough Adaptive Management Working Group envisioned an integrated monitoring and study plan to address the hypotheses to be tested. While those hypotheses are still the basis for the design of the restoration plan, this document only describes those methods for measurements for the “key” uncertainties (listed on page 10). This adaptive management plan includes no special studies requiring periodic intensive effort using tagged fish, mercury measurements, bird studies, etc. The physical design of Dutch Slough restoration as a “living laboratory” provides a powerful opportunity for addressing many issues not included in this plan, and it is hoped that other entities will use the site to study issues and hypotheses not specifically addressed here. Descriptions of other site-specific hypotheses developed by the Dutch Slough Adaptive Management Team for the 2008 Plan are retained in this document as guidance and examples of such future studies.

This updated 2016 Adaptive Management and Monitoring Plan is intended to guide the collection of data to assess how well the project meets the performance measures and the adaptive management and restoration objectives. This document recaps the 2008 Plan, updates the hypotheses to reflect 2016 scientific understanding and the revised final physical design of the restoration. This plan also incorporates more precise descriptions of the monitoring needed to address the hypotheses and the methods that are likely to be most appropriate on the Dutch Slough site.

Development and Direction of the 2008 Adaptive Management Plan

An Adaptive Management Working Group (henceforth 'Group') was convened in 2004 to develop an Adaptive Management Plan (Plan) for the Dutch Slough project. The Plan's primary purpose was to pose hypotheses about marsh restoration that could be tested at the project site, the results of which would inform the design or implementation of future Delta tidal marsh restorations. The Group's process included developing conceptual models, identifying and prioritizing key uncertainties, selecting hypotheses to guide research and design, and development of a conceptual restoration design.

Scientific Basis

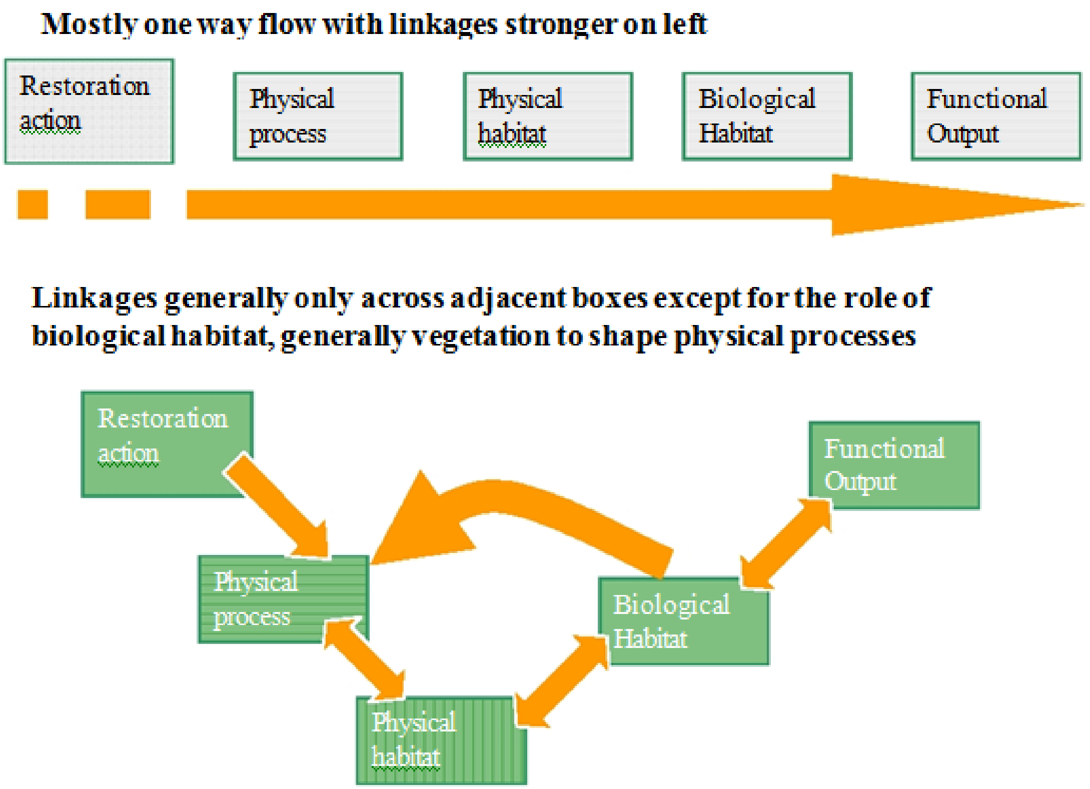
To provide a clear and scientific basis for the restoration of Dutch Slough the Group aimed to:

- refine existing and develop new conceptual models,
- identify and prioritize key uncertainties,
- select hypotheses to guide research and site design, and
- development of a conceptual restoration design.

The Group started with the tidal marsh restoration conceptual model developed by the CALFED Delta Habitats Group, refined it for the Dutch Slough project, and also developed models for Chinook Salmon growth and survival, and mercury methylation. Although these conceptual models approximate how ecosystems work and describe the relationship between restoration, physical processes, habitat, and species, there were significant uncertainties regarding how physical processes shape habitat and how habitats affect target species. Explicitly identifying these uncertainties and designing restoration actions or management interventions to generate information that can reduce uncertainty is the purpose of adaptive management.

While the Group was working, several members were also involved with a broad conceptual model system developed under CALFED – the Delta Regional Ecological Restoration and Implementation models (DRERIP). Many of the DRERIP models were published in 2008, at roughly the same time as the original Dutch Slough Adaptive Management Plan. The regional and all-inclusive nature of the DRERIP models did not address the needs of the Dutch Slough Plan which was focused on one site and two fish species. However, the DRERIP effort led to many scientists in the region identifying areas of scientific agreement and uncertainty. This led to the Group developing conceptual models specific to Dutch Slough.

Two of the most central conceptual models developed by the Group are shown in Figure 1. The top diagram shows the general pattern of the cascade of effect from restoration actions through changed physical processes that change physical and biological habitats that output a change in fish status. There is one feedback loop where vegetation in biological habitat can have a feedback effect on physical processes. The lower part of the figure elaborates on the processes, habitats and outputs involved.



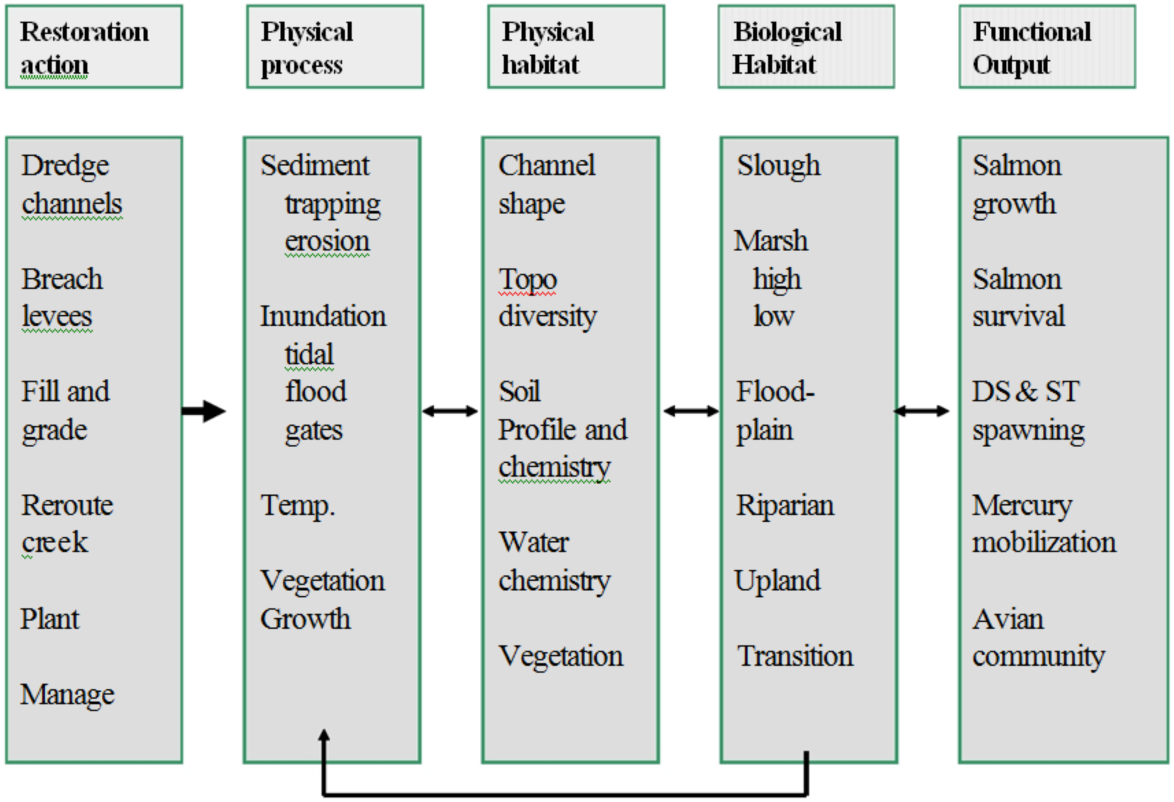


Figure 1a and 1b Two general conceptual models of the 2008 Plan showing how restoration actions affect physical processes, physical habitat, biological habitats and biological outcomes

Table 1: Categories of Uncertainties

1. Fish Uncertainties

a. Are target fish populations habitat limited?
b. Are target fish populations food limited?
c. Are target fish populations predation limited?
    fish predators?
    bird predators?
d. Are fish populations limited by contaminants?
e. Do Delta Smelt spawn in marshes or channels?

2. Geomorphic-Habitat Type Uncertainties

a. What are important characteristics of open water vs. dendritic marsh for fish growth and reproduction, avian habitat, and MeHg and DOC production and dispersal?
b. Role and value of large channels vs. small channels for fish and birds?
c. Role and value of large order channel networks vs. small order for fish habitat and MeHg and DOC production and dispersal?
d. Relationship of channel density to fish utilization?
e. Value and role of high and low marsh for fish growth and reproduction, avian habitat, and MeHg and DOC production and dispersal?
f. What is the transport connection (fish, food, sediment, HG, DOC) between marshes and channels?

3.Geomorphic Process Uncertainties

a. What factors influence slough channel development and sustainability?
b. What elevation of marsh plain will allow channel development or maintenance through scour?
c. How is marsh plain elevation influenced by sediment dynamics, peat accumulation, initial elevation, subsidence and compaction?
d. Will marsh plain accretion keep pace with sea level rise?
e. What is the lowest elevation tules will establish and persist?
f. How will system respond to extreme events?

4. Submerged Aquatic Vegetation Uncertainties

a. What is the relative stability of native SAV population?
b. Linkages of different SAV structure and fish habitat?
c. Role of SAV as habitat for invertebrates?
d. Fish and bird benefits of different aquatic plants?
e. Can we control SAV by managing submerged substrate?

Key Uncertainties Selected

All of the uncertainties identified in Table 1 are important, but it is not possible to physically design the Dutch Slough restoration to address all of these uncertainties. Many of these uncertainties cannot be effectively evaluated at Dutch Slough because they are controlled by factors outside of the project boundaries. Other uncertainties could be better or more cost effectively resolved through research at other sites. Some uncertainties could be evaluated on a smaller scale at the site, but the entire site would not be physically configured to specifically evaluate these other uncertainties.

The Group loosely applied the following criteria, in no order of priority, to identify which uncertainties could be most effectively addressed through field-scale experimental manipulations at Dutch Slough.

1. What variables/uncertainties have the greatest implications for the future cost and feasibility of marsh restoration elsewhere in the Delta?
2. What variables can we test at Dutch Slough? What variables can be just as easily tested elsewhere?
3. What design feature variables will maximize the chances of seeing a response?
4. What variables can be experimentally tested while still maximizing the restoration value of the project?
5. What variables can be experimentally tested without significantly increasing the restoration costs?
6. Which variables can we control?
7. Which variables need to be tested on a large scale vs. variables that can be effectively tested on a smaller scale within the larger project

Applying these criteria to the list of uncertainties, the Group focused on two issues: marsh plain elevation and size of restored areas. From a management viewpoint, size and elevation are likely to be crucial and expensive aspects of tidal wetland restoration in the Delta because much land in the Delta is currently too deep to function as tidal wetlands and parcels for restoration are likely to be of varying sizes. Therefore, knowing the minimum size and elevation necessary to achieve desired functions are crucial to successful management.

Key Uncertainty #1: Function of Marsh Plain Elevation

The Group recommended Marsh plain elevation for experimental design because:

- Marsh plain elevation controls the frequency of wetting and drying, which is a key driver that determines all of the following factors: vegetation type and character, access for native fish, habitat for fish and birds, residence time and primary productivity, mercury methylation and dissolved organic carbon formation.
- Marsh plain elevation is also a key cost factor, since it costs far more to restore high marsh on subsided lands.

Several of the geomorphologists and ecologists on the Group struggled with the distinction between high marsh and low marsh since low, emergent marsh was rare or non-existent in the historical Delta. Although there is not a clear natural distinction between lower and higher marshes in the Delta, the intent is to compare "lower" and "higher" marsh plains that differ enough in elevation to show different ecological responses, while not making the "higher" marsh so high that it becomes fill- limited or cost prohibitive.

The origins of higher and lower marshes vary substantially. High freshwater marsh in the Delta was formed naturally over the last 6,000 years as sea level rose and formation of organic soil from deposition of wetland vegetation kept pace (Atwater, 1982). The persistence of tule marshes even as sea level rose is evidence that biological accretion of the marsh plain was faster or equal to sea level rise and was apparently limited by the upper extent of common high tides. As a result, the elevation of natural marsh plains in freshwater environments generally corresponds with mean higher high tide.

Low emergent marsh, in contrast, is an artifact of human settlement in the Delta. Vegetated marsh plain lower then mean high tide occurs in the Delta where subsided islands have been intentionally or unintentionally restored to tidal inundation. Tules persist below mean lower low water in some freshwater tidal environments, because they can apparently tolerate frequent and persistent inundation. (Simenstad et al. 2000)

Group members cautioned that constructed high marsh may perform very differently than natural high freshwater marsh. Natural marsh plains in the Delta would have been characterized by hummocky, organic soils built over centuries by decomposing tules and rafted organic material including large woody debris. Constructing high marsh plain with earth moving equipment and mineral soil would provide a far different, and presumably less diverse, environment. Rather, they recommend that we not attempt to construct high marsh, but rather let it grow to high marsh from a constructed mean tidal elevation marsh plain. For this reason, and because constructing high marsh would cost significantly more, the experimental design will compare low marsh plain environments (-0.5 to 0.5 MLLW) to mid marsh (MTL) as described below. The group believed that constructed marsh plains will gradually accrete biologically until they reach an elevation approximating mean higher high water, provided that biological accretion of tules occurs faster than sea level rise, as it did during the last 6,000 years. A ten year pilot project by DWR and USGS on Twitchell Island measured tule accretion rates of more than one inch per year which far exceeds estimates for sea level rise.

What is the relationship between elevation of marsh plain:
    to salmon and splittail growth and survival?
    fish food production and access or transport to fish?
    splittail and Delta Smelt spawning?
    methylmercury formation and dispersal?
    Dissolved organic carbon formation and dispersal?

Key Uncertanty #2: Function of Marsh Plain Size

The size of a tidal marsh drives certain geomorphic attributes which in turn influence its ecological functions. Marshes are watersheds, just very flat. The larger the marsh, the larger the watershed and thus the larger the flows. Flow velocities control sediment scour and deposition, which influences formation and maintenance of channel networks within the tidal marsh. Flows also control the magnitude of exchange between the marsh and its surrounding tidal waters. Freshwater tidal marshes have the added characteristic that tule vegetation can grow to below low tide elevations, which means open water channel networks generally occupy areas deeper than perhaps 2-3 feet below MLLW. The Delta Historical Ecology report (SFEI 2012) identified tidal marsh channel depths of 10 feet or more below low tide. Surveys at Sherman Island, one of the very few large tidal marshes in the Delta, showed tidal channel depths several feet below low tide. Channel networks with emergent marsh banks thus persist throughout the tidal cycle and are connected at their deepest end to the marsh's tidal source. Delta freshwater tidal marshes therefore are expected to have a well-connected aquatic component present through all tidal cycles. The marsh plain elevation dictates total "watershed" volume and thus influences the magnitude of flows, and material exchange, between the marsh plain and channel network and between the channel network and adjacent tidal waters. Lastly, emergent marsh vegetation functions as a physical filter to material movement across the marsh plain.

Channel networks possess six attributes relevant to their ecological functions: density (length and surface area), width, depth, volume, sinuosity, and bifurcation. Channel geometry and scale influence potential fish access, residence time and primary productivity, wetting and drying of channel edge, and fate and transport of water quality constituents. The Group considered a range of channel geometry measures such as channel density, cross sectional area, and channel order. Channel density was eliminated from further consideration based on the assumption that higher density was better for fish and more consistent with the overall ecological objectives. Although channel cross section shape is important, the Group recognized that it was not a good experimental variable both because it would be difficult to control on a large scale due to constructability issues and because it is determined in part by marsh plain elevation, the other independent variable they wanted to design the experiment around.

The Group focused on the role of the size of the marshplain through the geomorphic attributes of network length, channel width and depth and channel order (i.e. degree of bifurcation).These attributes influence a number of meso- and macro-scale factors including water velocities, tidal exchange volume, the diversity and size of channels in a given marsh drainage, the exchange of nutrients and other water quality constituents between the marsh and neighboring sloughs, and the extent of low water refugia for target and predatory fish. As discussed in the hypothesis section below, small-scale marsh areas are more likely to fully drain on low tide while large areas are more likely to contain areas that do not fully drain, creating refugial habitats for small fish.

What is the relationship between the length of a channel network, channel order, width, and depth of tidal channels to:

- rearing and foraging habitat for salmon and splittail?
- avian predation?
- fish predation?
- access to marsh food supply?
- transport and dispersal of food, water, sediment, DOC, and MeHg into and out of the Marsh?

Testing hypotheses

Utilizing the Plan, the project has been physically configured to generate information about how native fish (particularly salmon and splittail) respond to different types of tidal wetland habitat, and to provide ecological benefits for a wide array of native species.

The Group developed a suite of hypotheses about tidal marsh restoration in the Delta and its likely ecological effects. During this process, however, it became clear that a field-scale experiment would only be successful if it was focused on a narrow set of variables. The hypotheses most likely to be effectively tested at Dutch Slough were determined to be those related to elevation and marsh size (incorporating channel geomorphology. The restoration has been designed to facilitate the testing of these hypotheses, by creating “cells” of three different sizes, and at two different elevations.

In addition, opportunities for research on other factors such as subsidence reversal, exotic species management, dissolved organic carbon formation, mercury methylation, and avian habitat have been accommodated in the final project design. These opportunities concerning issues not directly related to the target fish species are not included in this monitoring plan, but the hypotheses felt by the Group to be most important are described.

The 2008 Dutch Slough Adaptive Management Plan identified the species of interest and developed key hypotheses relating to the value of use of restored wetlands to aquatic resources. Consultation with a number of biologists (see list in Appendix 1) in January 2015 indicated that those key hypotheses and species were still appropriate. No new hypotheses about those species, nor new fish species to consider, were suggested so this plan continues with those species and the hypotheses pertaining to them that were described in the 2008 plan. These hypotheses are described at length later in this document.

The adaptive management hypotheses address issues of fish feeding, growth, and survival that are also the goals of tidal wetland restoration overall. Therefore, monitoring to address these hypotheses will inherently provide information on the success of restoration at Dutch Slough, permit comparison with performance of other restored wetlands in the Delta, and provide information relevant to the design of other wetland restoration projects.

The 2008 Plan also included ten water quality hypotheses and ten bio-geomorphic hypotheses that could be tested given the site’s conceptual design at the time. Changes in the design and implementation plan for Dutch Slough have led to the elimination of some hypotheses in this update.

The tension between restoration objectives and research objectives was a recurring theme during meetings of the Group that developed the 2008 Plan. Some members felt strongly that the research objectives should in no way reduce the restoration benefits of the project, others countered that since we did not know which type of treatments would have the greatest benefit for native fish, we must monitor different treatments so that future restorations could optimize their designs. Therefore the restoration has been designed to both benefit a wide array of species and to generate information about how native fish respond to different types of tidal wetland habitat.

The 2008 Plan made several assumptions that are no longer accurate for the final restoration design. The most important of these changes are as follows, and will be addressed in this current document.

- The 2008 Plan was based on simultaneous restoration of all three project parcels. This was later determined to be infeasible, and now the parcels will be restored one at a time, with construction of the first parcel (Emerson) beginning in 2016, the second (Gilbert) in 2017, and the third (Burroughs) beginning in 2019 or later. This change defers comparison of results from replicates on separate parcels.
- The 2008 Plan assumed that the northern, subsided portion of each parcel would be initially restored as subtidal open water, and that on-site research would guide the ultimate restoration of these areas. These design decisions have since been made.
- The 2008 Plan included re-routing Marsh Creek onto Emerson Parcel as one possible design option. The current design includes this re-routing as a definite part of the construction. The presence of this freshwater source on the Emerson parcel limits comparisons between its marshes and those of the other two parcels.
- The 2008 Plan included broad prescriptions for a baseline monitoring program that was never implemented.

The 2008 Plan did not include any monitoring prescriptions for testing the selected hypotheses, or identify who would conduct the research. This current document outlines monitoring methods for specific hypotheses, but also does not identify the entities that will carry out the monitoring or data analysis.

Changes Prompting the Revision of the 2008 Adaptive Management Plan

Both the ecological landscape and the regulatory environment for Dutch Slough have changed. In addition, the permitting process, public interactions, and further engineering work have led to substantial changes in the physical restoration plan for Dutch Slough that was the basis of the earlier Adaptive Management Plan.

Changes in the scientific understanding of the Delta, its component species and its ecological structure are covered in the later section on scientific basis and conceptual model.

Changes in the Delta environment

Changes in the environment surrounding the restoration site include frequent summertime blooms of harmful algae, particularly *Microcystis* spp. in Dutch Slough and other nearby channels of the western Delta (summarized in Lehmann 2015). Toxins from *Microcystis* have been found at elevated levels in tissues of splittail (Ger 2009), a target species of restoration work. Splittail spawn generally between March and May, so the value of Dutch Slough as a spawning and early rearing area is likely unaffected by summertime blooms of *Microcystis*. However, permanent residency of splittail at Dutch Slough might expose them to increased risk of exposure to microcystin toxins in their food and habitat. The presence of young salmon in the Delta does not coincide with the summertime and fall *Microcystis* blooms so the value of the site to salmon is likely unaffected. Blooms of *Microcystis* in the low salinity waters of the western Delta in 2009 were likely responsible for the apparent displacement of Delta Smelt into higher salinity water further away from Dutch Slough (Baxter et al. 2015). In the original Adaptive Management Plan, Delta Smelt were expected to benefit from Dutch Slough restoration primarily through enhanced summer and fall food abundance in the channels of the western Delta near Dutch Slough. If *Microcystis* blooms occur in the channels near Dutch Slough, that could greatly reduce the benefits of trophic augmentation for Delta Smelt .

Changes in the regulatory and management environment

Since 2008 two changes in the management of the Delta have changed assumptions underlying the original approach of the first and second objectives.

The Delta Plan by the Delta Stewardship Council calls for restoration actions in the Delta to have a monitoring and adaptive management plan. The original Dutch Slough Adaptive Management Plan preceded the guidelines; this plan addresses those requirements and aligns the plan with other large-scale and simultaneous wetland restoration efforts, rather than the pioneering effort it was originally aimed to be. New long-term management plans for the Delta call for more acreage of valuable ecological habitats. The California EcoRestore program targets restoration of 30000 acres of wetland habitat, including 9000 acres of tidal wetland by 2018. Thus, Dutch Slough restoration is now proceeding in concert with many other restoration sites resulting in opportunities for comparison of how different sites meet similar goals and evaluation of the factors affecting restoration success. Such comparisons of early restoration work should provide valuable guidance for later projects. Dutch Slough provides a unique structure and location to experimentally test important ecological processes that cannot be addressed elsewhere, particularly the importance of wetland elevation and restoration site area.

Changes in the Restoration Design

Timing.

The most significant change to the restoration plan is that the three parcels will be restored in sequence, rather than simultaneously. The westernmost parcel, Emerson, will be restored first with landscape preparation in 2016 and levee breaches in 2017. Gilbert Parcel restoration will begin in 2017, and Burroughs last of all in 2019. This sequence of restoration has several implications for changes relative to the 2008 Dutch Slough Adaptive Management Plan:

1. Because Emerson comprises only two large test areas, the effects of different sized restoration areas is deferred until the other parcels are restored, but the effects of different elevations can be addressed.
2. Since monitoring of restoration response will begin at different times, we will better be able to identify changes through time that are due to the evolution of wetlands on site vs changes driven by climate or other regional drivers.
3. Early results from Emerson Parcel may provide important insights for later site-specific monitoring on Gilbert and Burroughs parcels

Structure

The physical restoration design for Dutch Slough has changed substantially; these changes are listed below and described in detail in the Supplemental EIR. The designs (from figures 3.2 and 3.4 from the SEIR) of the original and final designs are reproduced below (figures 2 a,b,c). Some of these changes have direct relevance to the adaptive management and monitoring plan. The changes in the design, with their expected impacts on restoration goals, include:

1. Marsh Creek now will have a fluvial connection to the southern end of Emerson Parcel, and provide freshwater flow onto the site. The current channel will still transport tidal flows and some outflow during high flow events, but the new channel through the Emerson parcel is designed to carry most of the water into and out of Marsh Creek. Freshwater passing through the Emerson Parcel may serve to attract salmon smolts that commonly engage in rearing in non-natal streams during their outmigration (Murray and Roeneau 1989; Scrivener et al 1994; Maslin et al. 1997) . Adult splittail may be similarly attracted onto the site by spring flows from Marsh Creek, although the triggers of their reproductive behavior are not well known.

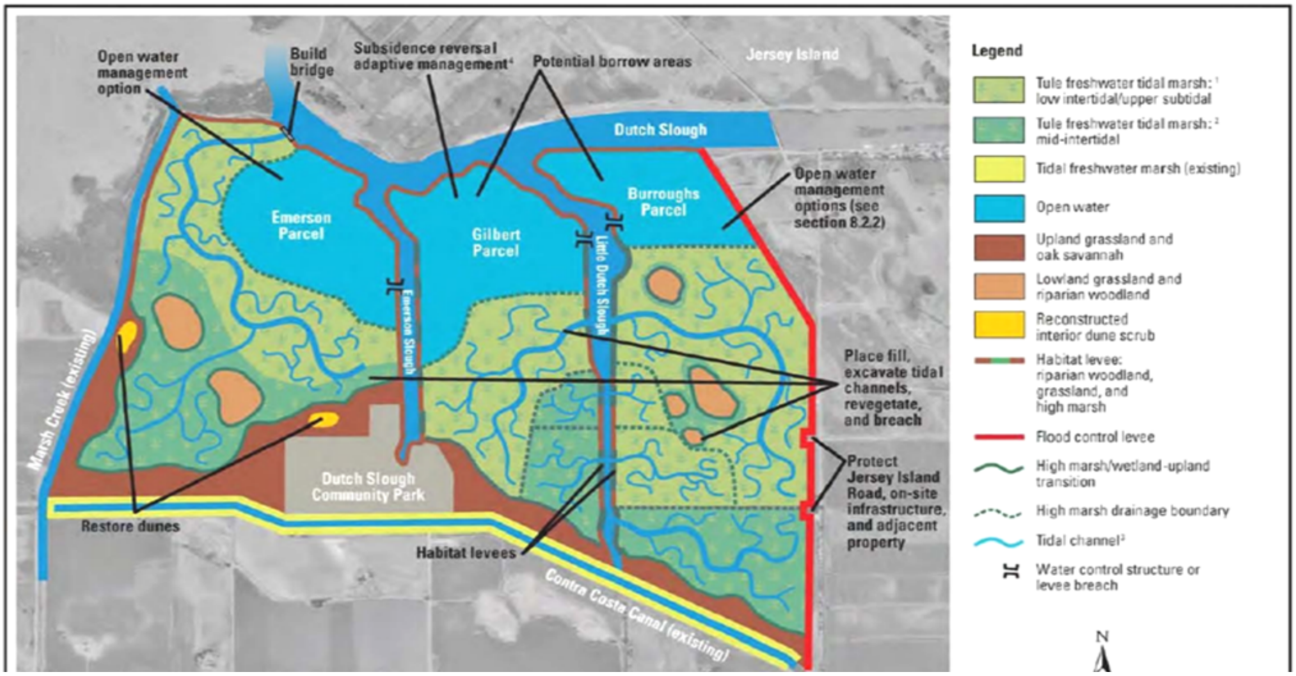
2. The southern boundaries of all three parcels now have a flood control levee, bordered by a riparian zone. This riparian replaces the upland, dune and savannah habitats n the original design. These changes have little impact on the aquatic aspects of restoration and monitoring, but provide suitable habitats for terrestrial and riparian species.

3. Open water areas at the north end of Gilbert and Burroughs parcels are now replaced by non-tidal marsh, and pasture and riparian areas, respectively. These areas provide nesting and foraging areas for Swainson's Hawks, Black Rails, Giant Garter Snakes and other terrestrial and marsh species.

4. The final design is presented in the third figure below. The change from the figure in the Final SEIR is that the levee breaches from Little Dutch Slough onto Gilbert are combined into one main breach, with the four sub-units all drawing their water from one main channel. In addition, the cross levee separating managed from tidal marsh has been moved northward, increasing the acreage of tidal marsh. This design makes all four sub-

units into parts of one large hydrodynamic unit and makes the two cells of intermediate size on Burroughs more comparable in area to those on Gilbert.

5. The alignment of the eastern flood protection levee is moved from the eastern Project boundary to an alignment on higher ground, and in a location that reduces fill volumes and cost. The new levee alignment will result in decreased acreage of tidal marsh compared to the 2008 design, but will provide foraging habitat for Swainson’s Hawk and other raptors**.**



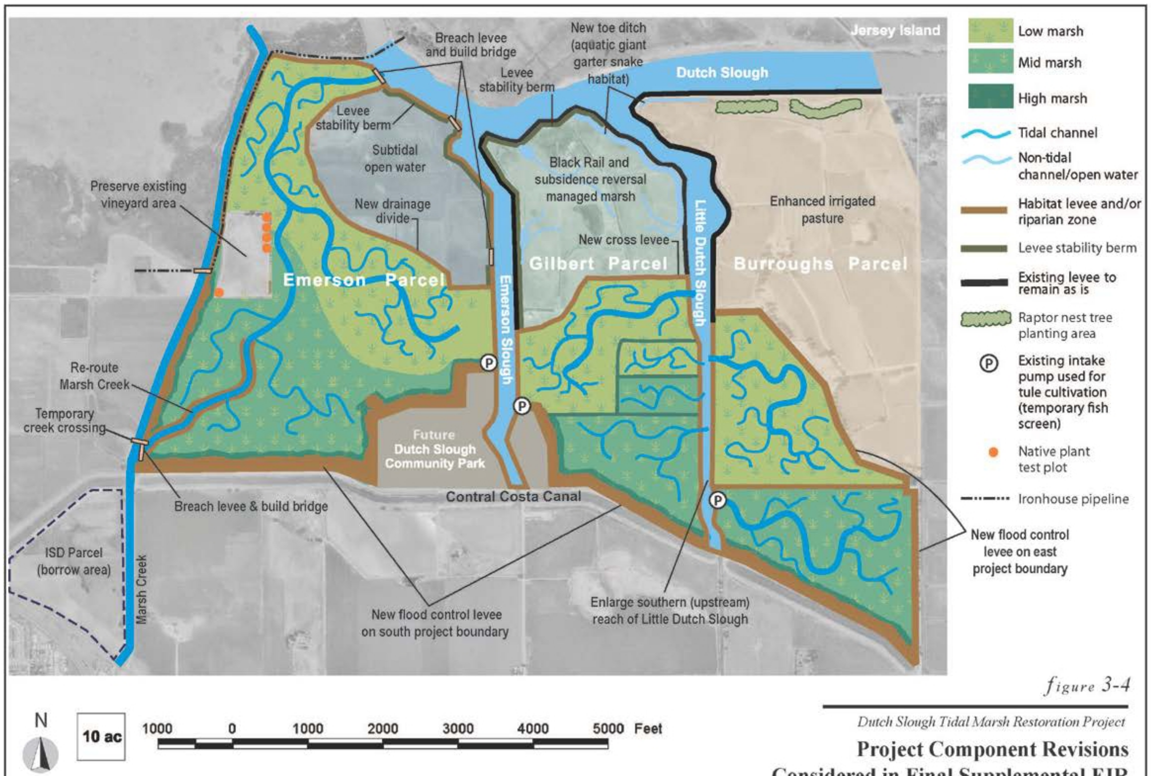


figure 3-4

Dutch Slough Tidal Marsh Restoration Project

Project Component Revisions Considered in Final Supplemental EIR

ESA PWA

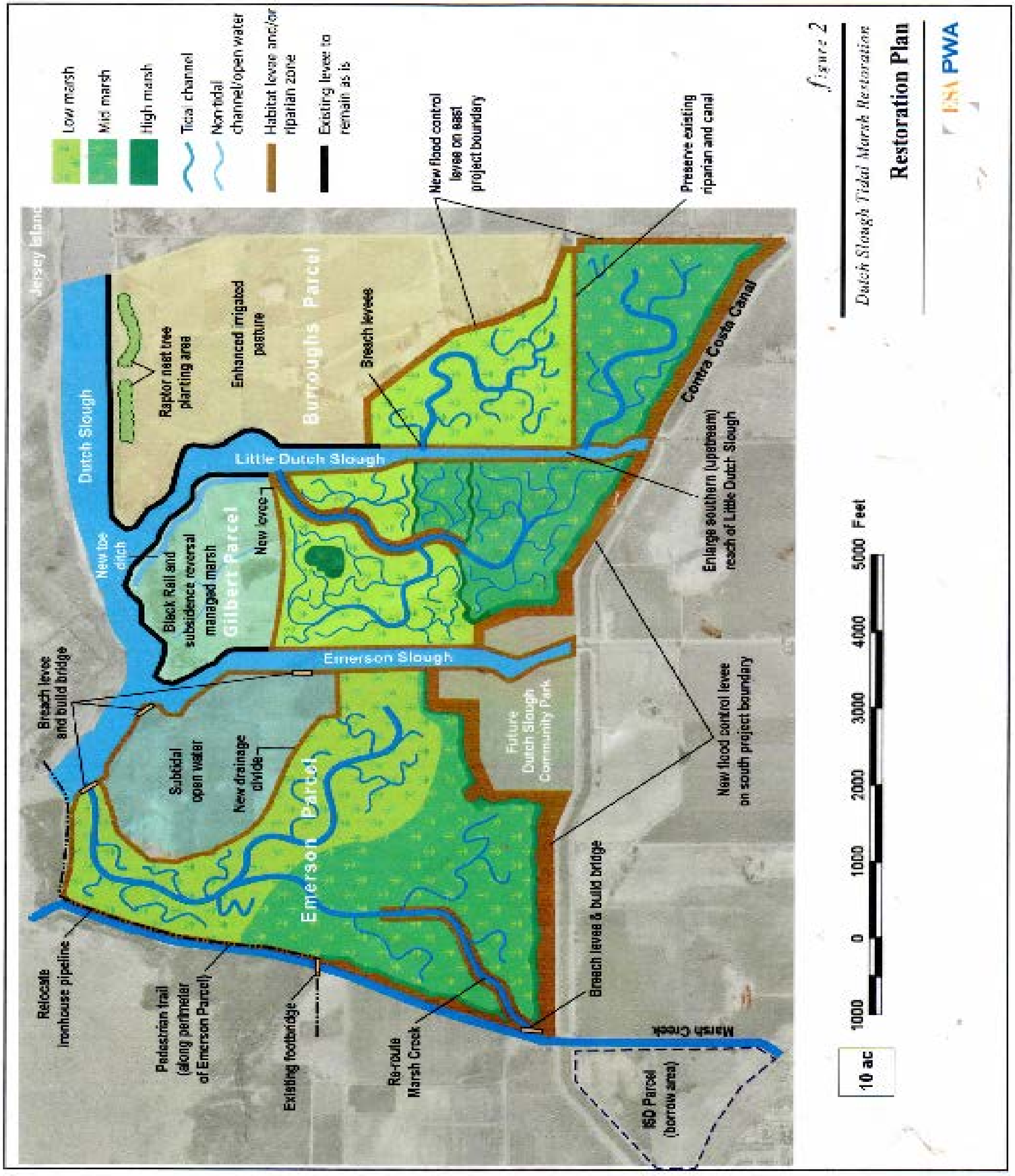


Figure 2a, 2b, and 2c Original restoration design (2a), the EIR published version (2b) and the full page final design (2c)

Hypotheses and Metrics

This plan focuses on the adaptive management aspects of restoring and monitoring Dutch Slough. Performance monitoring measures how well the site is meeting its goals and how performance varies from year to year and under different conditions of weather and salinity distributions. Because of the physical design of the restoration on each parcel, different aspects of performance are expected to vary with the elevation and size of each cell. The degree to which different performance metrics do vary from cell to cell will allow determination of the importance of size and elevation in contributing to the growth and survival of splittail and salmon on site. Thus, performance monitoring and adaptive management monitoring grossly overlap in their tools, techniques and application. We describe seven performance measures in this plan and include guidance on how to monitor and assess them.

Adaptive management monitoring extends beyond simple performance monitoring in that a higher level of detail in space is required. Also, application of the results of adaptive management monitoring is greatly enhanced by knowledge of the mechanisms causing differences in survival or growth. Adaptive management monitoring includes measurements of the food, predation and physical habitat features that likely would produce differences in survival and growth. Results from such adaptive management monitoring greatly increase the ability to replicate at other sites any enhanced performance measured at Dutch Slough.

The structure and location and conditions of Dutch Slough make the site suitable for an array of simultaneous studies on geomorphic process, water quality effects and avian uses. We update and include an array of hypotheses on these topics from the 2008 Plan, but do not include guidance on how the hypotheses would be addressed.

Restoration Performance Hypotheses

The primary species-specific goals of tidal wetland restoration at Dutch Slough are to enhance growth and survival of young, outmigrating salmonids and enhance spawning and rearing of splittail in years with drier springs. High levels of primary productivity may contribute to increased trophic support in the summer and fall for Delta Smelt. These expectations are summarized in the following seven hypotheses that mostly address the expected outcomes of restoring Dutch Slough to tidal wetland and so they focus primarily on measuring the overall abundance and condition of fishes on site.

<u>Salmonid and Splittail Benefits</u>. Dutch Slough lies near the confluence of the Sacramento and San Joaquin rivers so that all young migratory salmonids must pass near it before they can reach the Bay and ocean. Spring flows from Marsh Creek may attract young salmonids onto the site for a final round of non-natal rearing before entering salt water and simple dispersal during the tidal and diel cycles of migration is likely to cause salmonids to enter the site. Splittail spawn in wetland habitats in all years, and in wetter springs vast numbers are produced on the flood bypasses and are then found throughout the Delta. For salmonids and splittail that occur on Dutch Slough, once it is open to tidal action, we hypothesize that:

PM1 Aquatic Habitats at Dutch Slough will improve rearing for young salmonids and spawning and rearing habitat for splittail. Metrics to be monitored are fish community composition and abundance via beach seines and other gear from December through June.

PM2 Dutch Slough will support better growth of young Salmonids and Splittail than river channels and will be comparable to growth rates on floodplains and other restored sites. Metrics of fish length/weight ratios and presence of external parasites or spinal deformations are expected to show better health than data from nearby channels. Additional metrics, if warranted, could include otolith analysis and evaluation of liver condition.

PM3 Dutch Slough will provide enhanced survival of young fish on site. Metrics of abundance of predatory fish, diet composition of predatory fish, and size distribution of young salmonids and splittail are expected to show lower incidence of predation in comparison to nearby channels.

Delta Smelt Benefits. Dutch Slough is near the usual summer and fall abundance center of Delta Smelt, that are thought to be food limited (Baxter et al 2015). Primary and secondary production from Dutch Slough may substantially contribute to growth and survival of Delta Smelt in nearby channels. Most other proposed restoration sites are not situated as closely to the summertime and autumn low salinity zone and cannot be expected to either provide much off-site export of productivity or to allow smelt to move into adjacent channels as may happen at Dutch Slough. Dutch Slough provides none of the habitats usually occupied by Delta Smelt so benefits are most likely to accrue off-site. Hydrodynamic modeling and special studies would be needed to combine with monitoring of material flux off-site to approximate Dutch Slough's contribution to nearby pelagic habitats. This monitoring plan only addresses monitoring of conditions on-site at Dutch Slough. For Dutch Slough contributions to nearby trophic webs that support Delta Smelt, we hypothesize that:

PM4. Dutch Slough will export, on every ebbing tide, substantial biomass of zooplankton of suitable size and food value for pelagic fish, including Delta Smelt. Zooplankton abundance is expected to be measurably greater in water exiting Dutch Slough on ebbing tides than that entering on rising tides. Community composition of exported zooplankton is expected to be of species and life-stages reported from gut analysis of Delta Smelt (Slater and Baxter 2014).

Restoration effects and evolution. The hypothesized benefits to fish populations are based on a number of ecological processes that will develop, change, and interact through time. Some of these processes will be swift and clear, such as the initial inundation providing new aquatic habitats and shortly thereafter populated by aquatic organisms. Other processes will be much slower and more subtle, such as sediment accretion or erosion, changes in channel depths causing changes in plant species composition with subsequent changes in predation risks for young salmon. Most wetland ecological processes will vary substantially from year to year based on variability in the weather and long-term trends associated with climate change.

If fish monitoring data from on-site support PM hypotheses 1-3, i.e. target fish are abundant, growing and surviving in Dutch Slough, then the project will have met its intended goals. But abundance, growth and survival will vary from year to year and other restored sites may do better or worse than Dutch Slough. Therefore, monitoring data on the ecological processes that support fish, and how they change through time, is essential. Thus, we hypothesize that:

PM5. Sediment dynamics and organic accretion will change the elevation and topography of the wetlands through time. Sediment capture and organic matter accumulation will maintain relative elevation of Dutch Slough tidal marsh in pace with climate change and sea level rise. Metrics of sediment deposition will reflect sediment loads in neighboring channels and will vary with river flow and weather conditions in the Central Valley, with deposition greatest in years of high precipitation. Increases in sea level and storm intensity associated with climate change will affect sediment dynamics more in areas of lower elevation, where inundation is more prolonged each day. Sediment loads entering the Delta have fallen sharply since the year 2000 (Schoellhamer 2011; Schoellhamer et al. 2013). Organic material accumulation on the carbon sequestration marsh projects on Sherman and Twitchell islands has been roughly 5-8 cm per year (Bryan Brock DWR pers. Comm.) Such accumulation rates exceed projected sea level rise rates (Cayan et al. 2009). The relative contribution and ability of organic accretion to keep up with sea level rise at Dutch Slough could be tested by measuring the ratio of organic to inorganic material in sediment cores.

PM6. Initial conditions and subsequent sediment dynamics will determine plant species composition and vegetation architecture, thereby determining the opportunities for refugia and feeding sites for target fish species. Metrics of channel complexity and length are expected to increase from initial configuration because tidal energy in constructed channels will enhance erosion upstream (aka headcutting) to produce a network of smaller channels into the tidal plain. Metrics of emergent vegetation cover are expected to show, at least, an initial expansion from growth of planted tule cores. Increased channel and vegetation complexity will enhance feeding, growth, and survival of target fish as measured for PM Hypotheses 1-3. Thus, the suitability of the site is expected to increase in the years following inundation. Establishment and growth of aquatic vegetation (all types) will result in localized decreases in water velocity, promoting sediment accretion, decreases in depth and increases in water clarity. These physical changes will in turn provide more habitat for submerged and emergent aquatic vegetation, and perhaps produce changes in plant species composition. Changes in plant species composition will likely lead to changed abundances and composition of the fish community on the site.

Other Measures of Performance

The restoration design of Dutch Slough has incorporated a number of assumptions that should be evaluated. Data gathered to assess the performance hypotheses will also allow assessment of the broader questions of success, including:

1. Did the planting of tule plugs before levee breaching promote the establishment of desired vegetation better than areas colonized naturally? Did plugs lead to better vegetation cover in lower elevations?
2. Did the channel design produce velocities and water turnover rates sufficient to reduce the coverage of floating aquatic vegetation? Where deep water areas were designed, were they deep enough to prevent the dense growth of *Egeria?*
3. Are constructed channel networks stable through time? Do they promote headcutting of smaller channels into more dendritic networks? Do they fill in excessively with sediment or vegetation? Are constructed elevations stable through time or is there differential sediment capture causing the higher elevations to erode and deposit into the lower marsh?
4. Do target fish access the restored site in measurable numbers? Are invasive species occurring at levels of concern?
5. Do other desirable species, especially Giant Garter Snake, Black Rails and Swainson's Hawk use the site?
6. Does public access to the Emerson parcel affect its wildlife value?

Most of these questions can be answered from the data gathered as part of this monitoring plan and are likely to be valuable in planning other restoration sites.

Adaptive Management Hypotheses

Wetland restoration at Dutch Slough is designed to meet and better assess the uncertain needs of targeted fish in tidal wetlands and how management can best meet those needs. The number of planned tidal wetland restoration sites in the Delta has increased greatly in the years since the initial design of Dutch Slough, increasing the importance of addressing these unknowns. These uncertainties are summarized in the following five hypotheses. Although these hypotheses are stated positively, each is followed by a brief contrast of the ecological ideas that support and refute them.

These hypotheses provide a major basis for the adaptive management aspect of Dutch Slough restoration and are referred to through the rest of this document as Adaptive Management Hypotheses AM1, AM2,… AM5. In general these hypotheses address the processes that are likely to control the results of the performance of restoration in meeting the restoration objectives contained in the performance measures in PM 1-7. As such they mostly focus on differences among the cells and on differences of the physical habitat and food organisms in the cells of different elevation and size.

AM1. Food Resources for splittail and juvenile salmon will be greater in lower marsh than higher marsh due to increased duration of inundation and increased water column volume in the photic zone. See Figure 3.

Justification: Water will remain on the marsh plain for longer periods and more water per area will be available for photosynthetic activity that drives primary productivity and produce greater total abundance of prey populations. However, in the higher

marsh lower water volumes may concentrate food in the water column and allow more efficient feeding.
Monitoring required: Bathymetry, sediment dynamics, vegetation structure and composition, inundation duration of each cell, primary productivity, benthic and littoral secondary productivity and composition.
General evaluation approach: Compare trophic dynamics across cells and with other restoration sites at diverse elevations.
Limitations. Comparison can be done independently on each parcel since each parcel contains cells of different elevation. Different dates of restoration for the three parcels will limit ability to compare across parcels, especially in the first few years as vegetation establishes and substrates settle.

AM2. Splittail and juvenile salmon growth will be greater in lower marsh and channels due to increased duration of feeding opportunities. See Figure 3.
Justification: Greater food resources in lower marshes as described under #1 combined with longer feeding opportunities since low marshes drain for less time during each tidal cycle. Splittail are primarily benthic feeders, so epibenthic feeding may be constrained by duration of inundation. Salmonids, feed epiphytically along the edges of channels and may experience less difference in access to fringing vegetation as falling tides remove access similarly in marshes of either elevation.
Monitoring required: Length and weight of young salmon and splittail in each cell. General evaluation approach: Compare across cells and with size distribution of young salmon and splittail at monitoring sites in the Delta. If differences are found, special study of otolith growth rings would be required to assess if differences are effects of different growth rates or size-specific survival differences in the different cells.
Limitations: Comparison can be done independently on each parcel since each parcel contains cells of different elevation. Fish may move between cells and thereby reduce measured differences, so special studies using caged or tagged fish may be necessary.

AM3. Fish growth will be greatest in intermediate- and large-scale channel marsh areas, because higher order networks are more likely to maintain feeding opportunities during low tides. See Figure 4.
Justification: While elevation will control the duration of inundation (as in hypothesis #2 above), smaller marshes will provide fewer refugia for small fish at low tide, limiting feeding opportunities. Large marshes will provide larger and more numerous refugia at low tide. This hypothesis assumes that smaller marshes will always drain completely, but they may not drain completely during late winter and spring months when Delta water levels are often higher than normal due to river inflow. To the extent target fish (juvenile salmon and splittail) are most abundant in late winter and spring, the actual extent and effect of marsh drainage may have less effect on feeding and growth when it is most important.
Monitoring required: Size distribution of juvenile splittail and salmon.

General evaluation approach: Compare across cells and with size distribution of young salmon and splittail at monitoring sites in the Delta. If differences are found, special study of otolith growth rings will be required to assess if differences are effects of different growth rates or size-specific survival differences in the different cells.
Limitations: This comparison cannot be done until cells of different size are monitored, thus inundation of the Gilbert Parcel is essential because it contains both small and medium sized cells and permits comparison with the two large cells on Emerson Parcel, that will be flooded first. Inundation of Burroughs, the last parcel, will allow better evaluation of how intermediate sized cells function.

AM4. Fish survival will be greatest within intermediate-scale channel marsh areas, because higher order networks will harbor predators and lower order networks lack sufficient refuge during low tides. See Figure 4.
Justification: The amount of channel inundated to a depth of over 0.5 meter at low tide is a function of marsh size. This hypothesis is based on the assumption that most predator fish require depths of greater than 0.5 meter and would therefore be more abundant in the largest marshes or at the mouth of small marshes that completely drain on each tide cycle. Intermediate sized marshes would drain enough to limit predator habitat in the marsh channel, but would not drain so completely as to force target juvenile fish into the main sloughs where predators are known to be abundant.
Monitoring required: Size distribution of juvenile splittail and salmon. Predator surveys to establish distribution and diet.
General evaluation approach: Predation is generally size selective, taking smaller individuals disproportionately. Predator distribution and diet reflect where predators are foraging and what they are foraging upon. If prey size distribution, predator distribution, or predator diet composition change in cells of different size and complexity, special studies using tagged fish would be indicated.
Limitations: This comparison cannot be done until cells of different size are restored and monitored. Thus, inundation of the Gilbert Parcel is essential to address this hypothesis because Gilbert contains both small and medium sized cells and permits comparison with the two large cells on Emerson Parcel that will be flooded first.

AM5. Low marshes will export more food resources to adjacent Delta channels than mid elevation marshes due to increased food production on low marshes, greater tidal prism, and generally greater proximity to channels.
Justification: Greater productivity and longer inundation will move marsh production into surrounding channels where it can augment the pelagic food chain. However, greater access and use of food production by fishes residing in the lower marshes may reduce the amount available for export off-site.
Monitoring required: Concentration and community composition of primary and secondary productivity at the discharge location of each cell.

General evaluation approach: Estimate flux to establish if there is a net movement of productivity off the restored site. Compare across cells to establish any differences in the productivity due to either differences in water volume, differences in growth or consumption of primary of secondary production.
Limitations: The fate of exported productivity cannot be assessed by onsite monitoring.

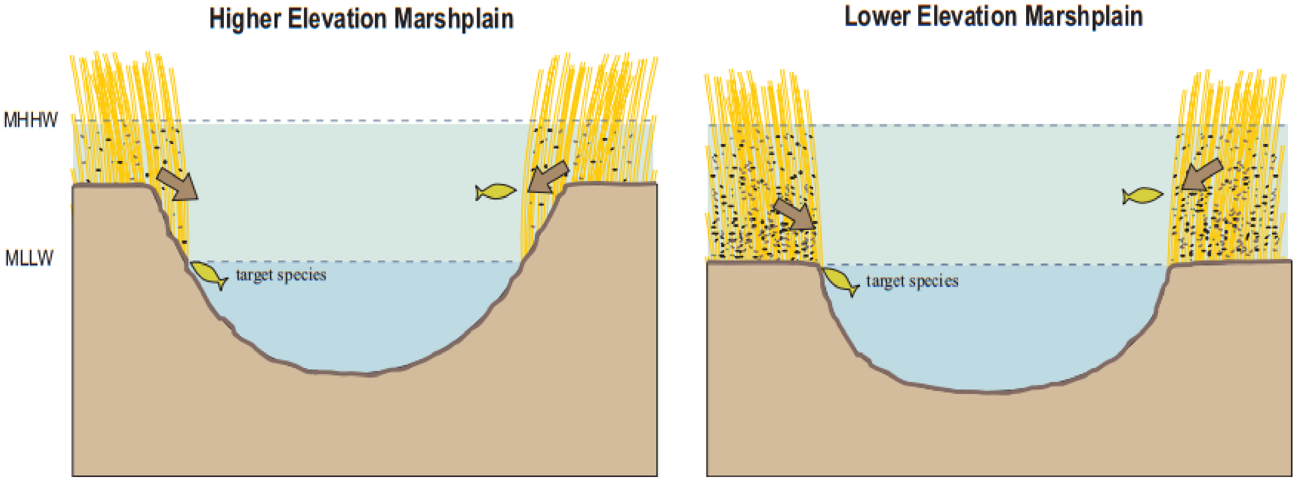


Figure 3. Lower elevations of marshplains support greater food production and more epiphytic feeding opportunities, but the benthic and edge areas are more similar..

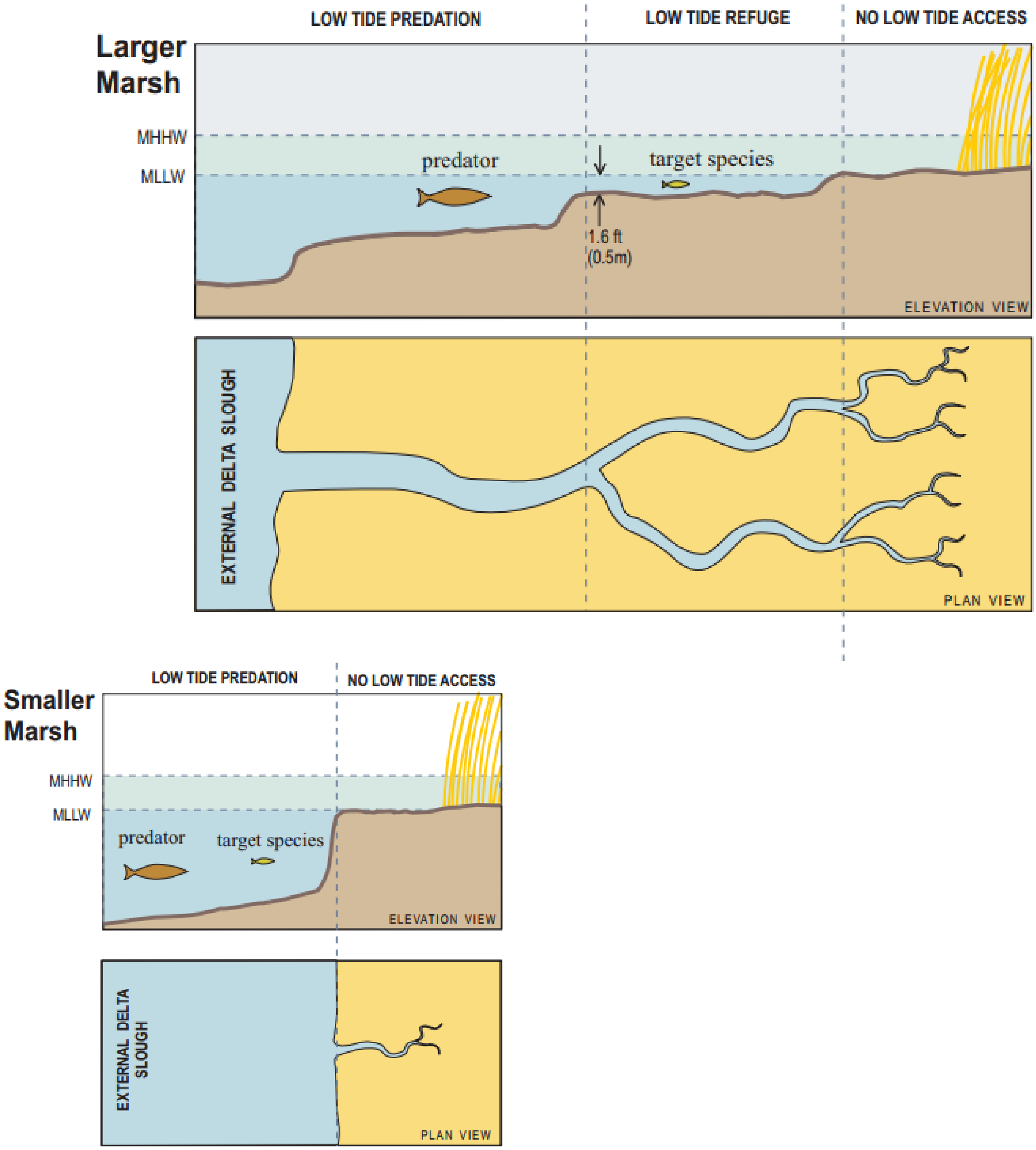


Figure 4. Hypothesized differences in predation risk in small vs large marshes, where large marshes provide more areas of intermediate depths that are too shallow for predatory fish to feed but deep enough for small fish at low tide

The three sizes of experimental plots at Dutch Slough span the range of marsh sizes where channel (thalweg) depths change substantially (Figure 5). Given the data presented in Figure 4, the smallest plots may not support predation refugia, but construction of channels may allow the size:depth relationship to be different. Results from Dutch Slough could be critical in assessing minimum size for successful performance of restored parcels.

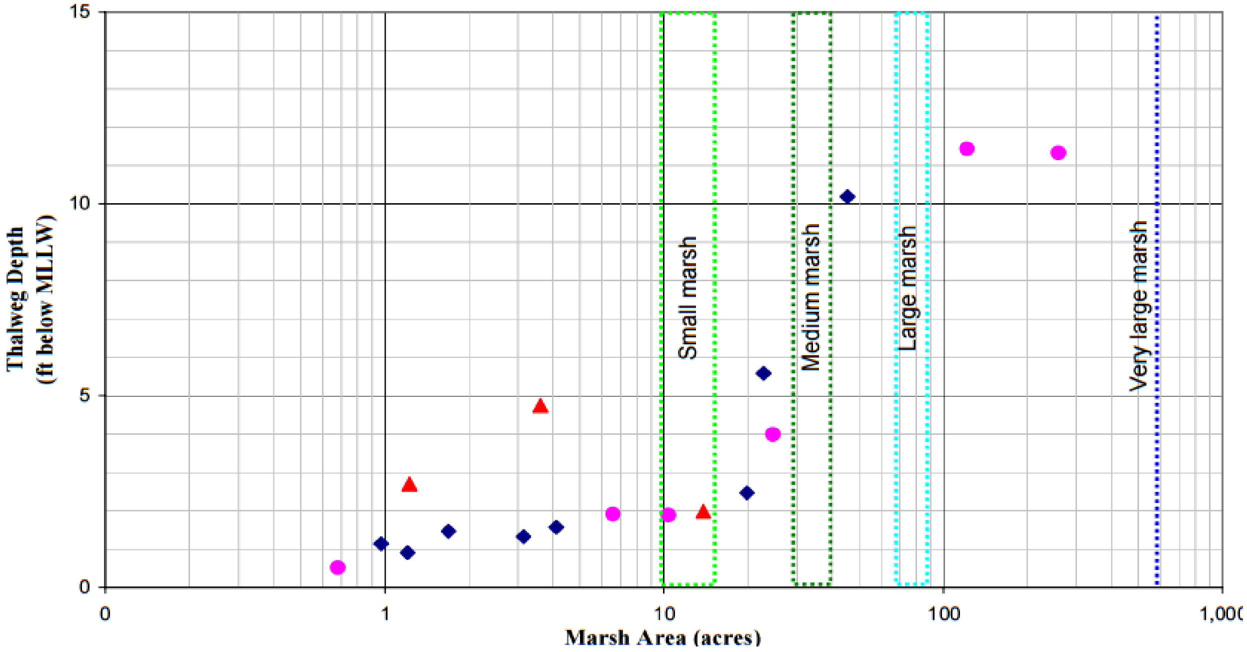


Figure 5. Thalweg (Channel) depth supported by tidal wetlands of different area. This relationship may shift in relatin to tidal energy at a site and with structure of the site.

Directly testing these hypotheses might be done by releasing tagged fish, tracking isotopes off-site and into the neighboring waterways, and other methods that are beyond the usual scope of 'monitoring.' This monitoring plan includes no elements of such special studies, although such studies are consistent with the plan, are the basis of some aspect of site design, and could be based on early results of the monitoring described in this plan. Such studies are vital and should be promoted by outreach and cooperation with local universities, management agencies, and other scientists.

Indirect testing of these hypotheses can be accomplished in the course of routine monitoring on and immediately adjacent to the Dutch Slough site. For example, chlorophyll concentrations, invertebrate concentrations, predator concentrations and target fish size and abundance in the six different test areas could support or refute the assumptions described above. The use of the Tidal Monitoring framework developed by the TWM PWT will ensure comparable from other sites Results from similar, simultaneous restoration monitoring would add depth and confidence in conclusions from Dutch Slough, but this plan does not assume anything about monitoring of other sites.

Bio-geomorphic Hypotheses (not included in monitoring plan)
The structure of restoration, eight units of two different sizes and three different elevations, lends the site to addressing many hypotheses beyond the targeted fish questions above. The following hypotheses are derived from the original 2008 monitoring plan, but changes in the plan for Dutch Slough and new knowledge have led to the elimination of some others. The revised plan to restore the three parcels of Dutch Slough sequentially would allow some of these hypotheses to be tested in a similar sequential fashion, using the initial restoration of Dutch Slough as a control site.

A number of significant uncertainties about tidal wetlands in the Delta can be addressed at Dutch Slough with little effort needed beyond that needed to address the Performance and Adaptive Management hypotheses. In several cases below we suggest small changes (*in italic letters*) to the restoration plan that would enhance the testing of these hypotheses.

BG 1. Marsh plains will accrete at rates equal to or greater than sea level rise in a variety of wetland environments. Non-tidal marshes on Twitchell and Sherman islands are accreting organic material 2-3 inches per year. Accretion rates in tidal wetlands and at different tidal elevations can be assessed simply by comparing elevation changes in the different experimental units of Dutch Slough. *Feldspar markers could be deposited on the as-built surface shortly after construction* and periodic sediment cores could be used to measure accretion from that surface over time.

BG 2. Subtidal emergent marshes (below MLLW) will accrete orders of magnitude faster than open water (including vegetation) areas with the same elevation. This hypothesis is based on the assumption that the presence of emergent vegetation is a catalyst for both physical and biological accretion processes that enable accretion rates to cross a threshold from very slow (mm per year) to relatively fast (cm per year). This could be tested by measuring accretion rates and sediment cores in restored emergent marshes compared to open water areas at Dutch Slough or in nearby Big Break and Franks Tract.

BG 3. **Tule vegetation established on subtidal elevations prior to tidal inundation will persist at elevations 1 foot below MLLW, but will not survive at greater depths**. Confirmation of this hypothesis is critical to knowing the minimum elevation for establishing tidal marsh and achieving the accretion threshold addressed in hypothesis 3 above. If tule vegetation persists at greater depths, then it may be possible to restore non-tidal subsidence reversal ponds to tidal marsh at a lower elevation. This hypothesis can be tested by measuring the presence, density, growth, and persistence of emergent marsh established at various subtidal elevations. *Tules could be cultivated on a variety of surface elevations above and below MLLW prior to tidal inundation* and their survival and growth monitored after tidal inundation.

BG 4. Invasive submerged Aquatic vegetation (e.g. Egeria) will be less successful in colonizing subtidal areas that are vegetated with emergent tule marsh or native Submerged aquatic vegetation (i.e. Stuckenia) . Colonization of shallow water zones by invasive, exotic submerged aquatic vegetation (SAV) is a major, pervasive problem. Pre-inundation cultivation of tules at subtidal elevation as described in hypothesis 4 may preempt establishment of undesirable SAV, even if the tules do not persist. Even dead, submerged tules may prevent establishment of SAV. *Randomly distributed quadrats with dead tules and with no tules(and in suitable sites Stuckenia) interspersed among the restoration sites would allow testing of the importance of tule presence in preventing the establishment of undesirable submerged aquatic vegetation.*

BG 5. Rice straw bales can be used to build-up subsided surfaces and sequester carbon without degrading water quality or marsh vegetation growth. Rice straw bales could be used to build-up subsided lands, but there are concerns about how rice straw bales could affect water quality or marsh vegetation. If rice straw bales are buried under 1-3 feet of mineral soil, tules and other wetland vegetation may be able to grow normally on the restored surfaces. Vegetation over shallower soils may be limited by competition for nutrients with organisms decomposing rice straw in the shallow soil. Decomposing rice straw could also increase DOC or other undesirable water quality constituents. *This hypothesis could be tested through a pilot project using rice straw to rebuild subsided surfaces as a substrate for restored marsh on Emerson* and using the information gained to guide use of such bales on Burroughs and Gilbert.

Water Quality Hypotheses (not included in monitoring plan)
Many of the water quality hypotheses discussed below involve measuring fluxes of MeHg or DOC off of restored marshes. The accuracy of the flux measurements will depend in large part on the ability to accurately quantify tidal inflow and outflow. Therefore, a high priority of the monitoring program will be to develop a numerical hydrodynamic model and a reliable method for measuring flow into and out of the various marshes to calibrate the model. It may be efficient to do this through establishment of a network of tidal gauges and optimal backscatter devices. Utilizing a calibrated model is an important first step for cost effectively testing hypotheses one through four below.

These hypotheses are presented to stimulate others to add on to this document, and to begin an integrated monitoring approach for the site. However, these hypotheses are not included in the rest of this monitoring plan.

WQ 1. Methylmercury production will be greatest on mid elevation marshes characterized by periodic wetting and drying and lowest on perennially inundated emergent marsh. This hypothesis is based on the assumption that flooding of vegetated wetlands or uplands or fluctuating water levels during tidal cycles could stimulate microbial methylation of inorganic mercury, increasing concentrations of methylmercury in water and biota. Measurements of methylmercury in fish from the Bay-Delta reveal that fish have higher levels in areas subject to periodic inundation such as high marshes and seasonal floodplains (Ye, et al, 2006). In contrast, fish from perennially inundated areas such as the Central Delta have relatively low levels of methylmercury suggesting that the more frequent and continuous inundation that would occur on lower marshes would result in lower methylmercury production (Slotton, 2007). Methylmercury production could be measured with soil and water samples from small plots, caged fish bioassays from various marshes, and on larger scales by measuring import and export of methylmercury or its surrogate (DOC) on tidal cycles. Monitoring techniques at Dutch Slough would be based on methods recently developed by the USGS, Moss Landing Marine Laboratory, and the UC Davis Mercury Group as part of the CALFED funded mercury monitoring program.

WQ 2. Methylmercury and DOC flux from the marsh to Delta channels will be greatest during extreme low tides during spring tide cycles. Data collected by both USGS and the Moss Landing Marine Laboratory indicate that fluxes of methylmercury and DOC from tidal marshes fluctuate greatly across the tidal cycle and that exports from marshes are greatest during extreme low tide events when tidal sloughs, banks, and associated pore water drain from the marsh. Methylmercury levels may concentrate in the pore waters where residence time and microbial activity are both high and then diffuse to adjacent channels and sloughs when the pore water drains from the soil. This hypothesis could be tested by measuring fluxes of DOC and methylmercury at the mouth of different marshes across the tidal cycle or by collecting water samples from marsh channels across the tidal cycle.

WQ 3. Total methylmercury and DOC flux from the restored marsh to the Delta will be highest on small-scale, mid elevation marshes that drain frequently and lowest on large-scale, low marshes that seldom drain completely. This hypothesis is based on the assumption that methylmercury production will be greatest on higher marshes (hypothesis 1 above) and that fluxes will be greatest on smaller marshes since they are more likely to drain completely on low tides (hypothesis 2). This assumes that DOC and methylmercury produced on the marsh plain will be efficiently exported out of the marsh. On larger marshes, in contrast, it would be cycled within the marsh. Pore water rich in DOC and methylmercury would not have time to drain from the marsh before flood tides would re-disperse it across the marsh plain. However, to the extent that tidal prism and total volume of water draining lower marshes is greater per unit area, it is possible that total methylmercury and DOC flux from lower marshes could be greater than mid elevation marshes even if hypothesis 1 and 2 are correct. This hypothesis could be tested with monitoring approaches described for hypothesis 1 and 2 above.

WQ 4. For a given marshplain elevation, total methylmercury and DOC export from the marsh to Delta channels will be greatest per unit area for small drainage areas and least for large marsh drainage areas. This hypothesis is very similar to hypothesis 3 above and could be tested with similar monitoring methods. This hypothesis may be particularly policy relevant since it may be far easier to implement many small restoration sites then it is to implement one large restoration site. If this hypothesis is correct, an equal area consisting of many small sites may have significantly greater water quality impacts than an equal area of a few large sites.

WQ 5. Soil substrate and vegetation characteristics influence methylmercury levels. Vegetation and soil type may influence the size of the reactive mercury pool, the rate of microbial activity and corresponding methylation, and the cycling of methylmercury. This hypothesis could be tested by establishing numerous small scale experimental plots with a range of soil and vegetation types. Measurements would include reactive mercury pool as well as methylmercury levels in soil, pore water, vegetation, and resident biota. Techniques developed by USGS and SFEI as part of the CALFED mercury monitoring program would be used (CBDA, 2007). Vegetation type, that is closely correlated to elevation on the marsh plain, will influence rates of mercury methylation and DOC formation. Certain plants may alter the rhizosphere by production of organic acids or release of dissolved oxygen, both of which are likely to affect mercury methylation rates. The effects of plant species on water quality could be investigated in test plots on a small scale.

WQ 6. Soil amendments such as iron will limit mercury methylation. Soil amendments could be added to experimental plots to evaluate mitigative effects. Data collected from the experimental plots could be compared with one another and with data generated from sampling conducted from the general restoration area to evaluate the respective impact on methylmercury production.

WQ 7. The diversion of Marsh Creek onto the restoration site will not significantly increase the methylmercury levels in restored marshes. Although there is an abandoned mercury mine and elevated mercury levels in the upper Marsh Creek watershed, fish from the mouth of Marsh Creek and nearby Big Break actually have the lowest mercury levels of all fish measured in the Bay-Delta watershed (Slotton, 2007). Even if inorganic mercury levels are high in Marsh Creek, they will not necessarily increase mercury methylation in restored marshes since levels of methylmercury may be controlled by microbial activity in methylating environments, not the total amount of mercury available. This hypothesis can be tested by comparing methylmercury levels from marshes connected to Marsh Creek with similar marshes not connected to Marsh Creek.

WQ 8. The diversion of Marsh Creek onto the restoration site will filter, trap, and/or bio-remediate pollutants in Marsh Creek and thereby reduce pollutant loads to the Delta. Water quality in Marsh Creek is poor. Creation of a wetland at the mouth of Marsh

Creek may improve Marsh Creek water quality before it enters the Delta. Alternatively, wetlands at the mouth of Marsh Creek may simply accumulate heavy metals such as copper or increase the levels of methylmercury. Even if the hypothesis is correct, acute toxicity events (pesticides, herbicides, chemical spill during low summer flows) in Marsh Creek could significantly harm biota in the restored wetland at the mouth of Marsh Creek. For this reason, it is not prudent to assume that a wetland at the mouth of this urban creek can both provide stable habitat for biota and filter pollutants that would otherwise flow into the Delta. This hypothesis could be tested by measuring Marsh Creek water quality upstream, in, and downstream of the restored wetland, and before and after restoration.

WQ 9. Tidal marsh restoration will increase production of dissolved organic carbon, result in net positive export of DOC out of the restored marsh, and thereby increase the level of DOC at the Delta's drinking water diversions. There are three parts to this hypothesis: 1) increase production of DOC, 2) export out of the marsh, and 3) entrainment in drinking water diversions. Even if restoration increases both production and export, it will not create negative water quality impacts unless DOC is transported from the restoration site to the drinking water intakes when water is being diverted. Due to Dutch Slough's westerly location in the Delta, DOC produced at Dutch Slough will most often be transported westward into Suisun Marsh and San Francisco Bay and therefore is not likely to increase DOC at drinking water intakes. Westward movement of DOC from Dutch Slough will be most pronounced in periods when net westward flow is greatest (presumably winter and spring). Conversely, the potential for eastward flow and dispersion is greatest when net-flow is lowest (summer and fall). Therefore, evaluation of the timing of net DOC production at Dutch Slough is directly relevant to questions regarding the impact of Dutch Slough on DOC concentrations at the Delta drinking water diversions. Furthermore, since some type of DOC is more likely to form trihalomethanes (THM), monitoring efforts should focus on the timing and potential transport of these more reactive species.

Avian Hypotheses (not included in monitoring plan)

California Black Rail (*Laterallus jamaicensis coturniculus*). State Threatened. The species inhabits remnant in-channel islands and managed wetlands throughout the Delta, and is associated with emergent wetland vegetation interdigitated with woody riparian species (Tsao et al. 2015). Tidal connectivity was found to be a strong predictor of presence along with the specific vegetation associations.

Monitoring for California Black Rail at Dutch Slough was initiated in 2011, and detections of the species have been made annually through 2015 on Gilbert and Burroughs parcels, and within waterside habitats west of Emerson parcel (Tsao and Melcer Jr. 2012). Within the parcels, the species is using emergent wetland vegetation associated with levee through-seepage. The species requires specific field survey methods (i.e. call-playback methods) and a careful sampling design (e.g. occupancy surveys) to address low detection probabilities (Richmond et al. 2008).

Black Rail Hypotheses

BR 1. Increasing the extent of wetland vegetation and especially tidal wetlands will increase the abundance and distribution of California Black Rail at Dutch Slough. Metrics are rail abundance and distribution in comparison to wetland development.

BR 2. Providing high water refugia (woody riparian species) will buffer the Dutch Slough population during high tides, storm events, and exacerbated stress due to sea level rise (Tsao et al. 2015). Metrics are population surveys following storm events, with comparison to Black Rail populations lacking nearby riparian refugia.

Swainson's Hawk (*Buteo swainsonii*). State Threatened. The species uses riparian tree species for nesting substrate, and wetlands, grasslands, and certain agricultural lands for foraging and post breeding staging and molting throughout the Delta.

Monitoring for the Swainson's Hawk at Dutch Slough was initiated in 2005, and at least one pair has been found nesting on site through 2015 (Bradbury 2005, Melcer Jr. 2010, Tsao and Melcer Jr. 2012). The species utilizes both nesting substrates (i.e. large trees) and foraging habitats (e.g. fallowed lands, irrigated pasture) within Dutch Slough.

Potential impacts of restoration actions at Dutch Slough are being offset through mandated habitat construction and protection on-site. Within the Central Valley, wetland and riparian land cover types provide essential resources for breeding Swainson's Hawks given the dominant extent of agricultural land uses (Schlorff and Bloom 1983).

Swainson's Hawk Hypothesis

SH1. Restoration of agricultural lands to riparian and wetland land cover types are equally beneficial to the species. Metrics include monitoring of the abundance, level of use, and nesting occurrence of Swainson's Hawks at Dutch Slough with comparison to surrounding agricultural areas and with parcels at Dutch Slough before and after restoration.

Avian Community Composition and Abundance. Birds are diverse in their life history requirements, and in many cases can serve as indicators of overall ecosystem health. Birds and bird communities respond readily to changes on the landscape, and can be used to evaluate management actions. They are also relaltively easy and inexpensive to survey.

Monitoring avian community composition and abundance at Dutch Slough was conducted in 2013-14 as part of a larger effort to define baseline estimates of species richness, diversity, abundance, and densities for birds throughout the Delta (Melcer Jr. 2014, Melcer Jr. 2015). Thus, protocols and pre-project data exist for this type of monitoring.

Dutch Slough provides habitat for species of special concern, and other sensitive species (e.g. Northern Harrier, Yellow-breasted Chat, White-tailed Kite).

Avian Community Hypothesis

AC 1. Quality and extent of tidal wetland habitats will increase through time after levee breaching on each parcel and will support increasing diversity and abundance of desirable avian species. Metrics would be subsumed in ongoing avian monitoring, with comparisons to wetland extent and quality.

Scientific basis and conceptual models for 2016

In anticipation of the need to be able to compare how restored sites differ in their function, the Department of Fish and Wildlife (DFW) has recently led an effort to develop a standardized tidal wetland monitoring framework to allow valid comparisons across sites, while recognizing that many sites will have different goals. In 2014 DFW formed a Tidal Wetlands Monitoring Project Work Team (TWM-PWT) within the Interagency Ecological Program to bring together about 45 scientists interested in developing a common monitoring approach. The team revisited the conceptual models developed under the CalFed Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) models, many of which were published (SFEWS 2008, ones not published are available here: (http://www.dfg.ca.gov/erp/cm_list.asp). These models are being used to guide the design of other restoration work, such as Prospect Island and Lower Yolo Ranch . Since many of the DRERIP models were developed more than seven years previously, the TWM-PWT first worked to update the models. The model figures were restructured using the format of the recent Delta Smelt conceptual model (Baxter et al. 2015). The model texts were updated with the wealth of information acquired in the last seven years.

From the outset, Dutch Slough aimed to design the site in order to test particular hypotheses as the basis of an adaptive management strategy. The natural division of the site into three parcels (Emerson, Gilbert and Burroughs), the diverse topography on the parcels, and the relatively large size of the site dispose Dutch Slough to an active Adaptive Management approach.

Conceptual Model Overviews

The TWM PWT conceptual model structure uses an overall model of tidal wetland dynamics to link seven models of ecosystem processes with models for the needs of young Chinook Salmon and Delta Smelt. The linked ecosystem processes are Tidal Wetland Overview, Foodweb, Aquatic Vegetation, Clam Effects, Contaminant Effects, Wetland Evolution, and a generalized Transport model. Each model begins with a visualization of the processes at work, the physical changes they produce and the impact of those changes on the target fish species or the other aspects of the ecosystem that affect those fish. Drafts of these conceptual models are attached to this report as appendices 2-9. These new conceptual models and the draft overall wetland monitoring framework are the basis of this effort.

Salmon.

The Conceptual Model for Chinook Salmon in delta wetlands was developed by Pascale Goertler, Kris Jones, Joseph Kirsch, Louise Conrad, and Ted Sommer as part of the PWT effort is and is attached as Appendix 2. Figure 6 modifies the general Delta salmon model figure in Appendix 2 by removing those processes unlikely to be important at Dutch Slough – largely the contaminant, flow, and entrainment effects. Soil tests of the land and Dutch Slough's documented land use as a working dairy suggest little concern for contaminant effects on salmon and splittail. Flows near Dutch Slough are dominated by tidal flows; these twice daily flows in both directions of 300,000 to 600,000 cfs swamp any physical effects of river flow. No water diversions are near the site and so entrainment effects are unlikely.

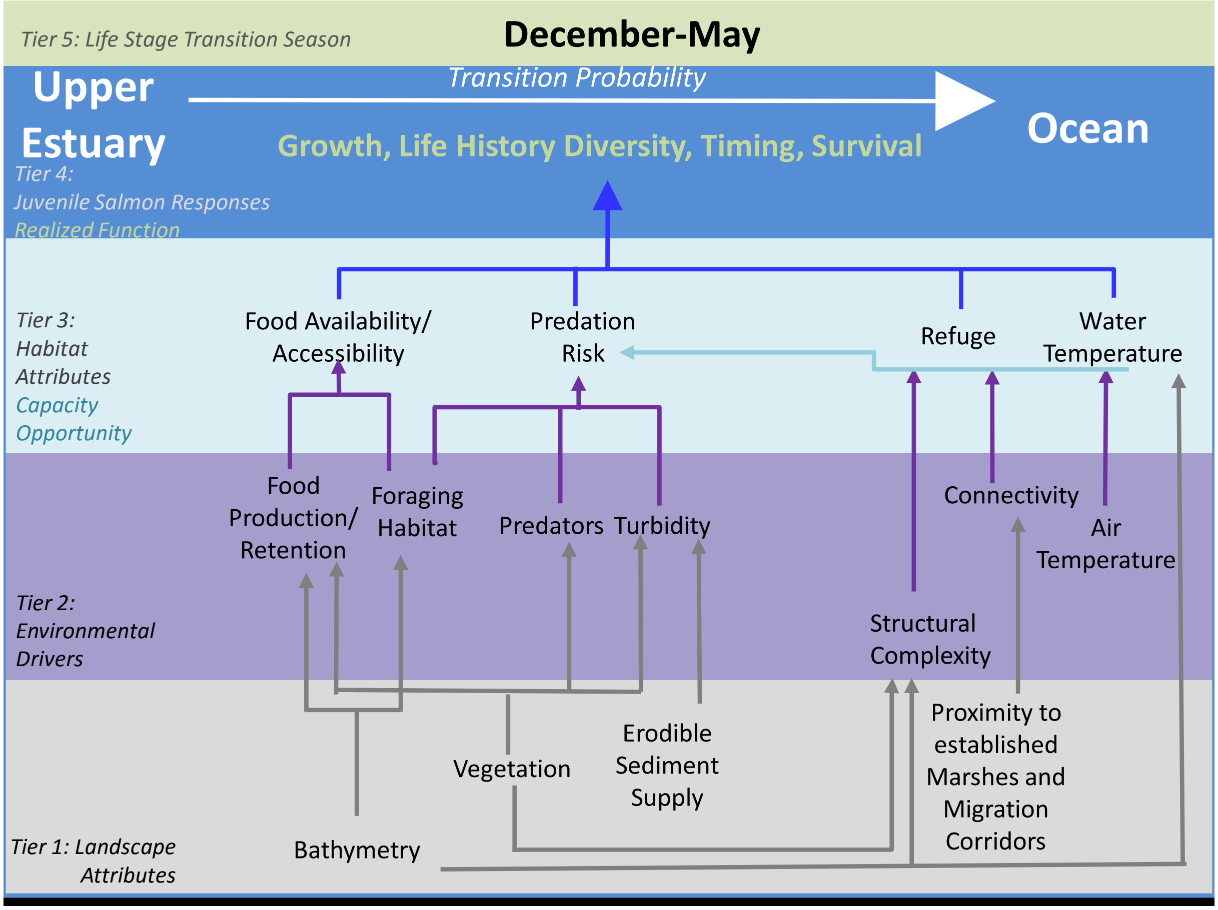


Figure 6. Thalweg (Channel) depth supported by tidal wetlands of different area. This relationship may shift in relatin to tidal energy at a site and with structure of the site.

As with most conceptual model figures from the TWM PWT, the bottom layer of 'Landscape Attributes' illustrates how the placement and nature of the site feeds into processes that control the environmental drivers on the site. For example the Landscape attribute of 'Bathymetry' directly affects the structural complexity of the site, the primary productivity of the site (because deeper waters are less well-lit and less productive), and the water temperature on the site by mediating the interface between water and air over every tidal cycle. The ‘Landscape Attributes’ and ‘Environmental Drivers’ may change substantially from year to year or through longer time frames, but they always provide the context for evaluating the monitored conditions on Dutch Slough. Together they directly affect fish ‘Habitat Attributes’ via their controls on ‘Food Abundance and Accessibility’, ‘Predation Risk’ and ‘Refuge,’ and the physiologically fundamental variable of ‘Water Temperature’. Thus, the Habitat Attributes control the desired outcomes of restoration at Dutch Slough, the Growth and Survival of juvenile salmon. This figure broadly identifies the physical and biological features that need to be monitored to assess how the effects of different elevations and sizes in the restoration design affect the growth and survival of young salmon at Dutch Slough. Details of the specific hypotheses and metrics are discussed below.

<u>Splittail</u>.

Some of these environmental aspects may be of different importance for young splittail compared to salmon, particularly because direct production of splittail is likely to occur on site (which is not an issue for salmon). The conceptual model for splittail has not been updated by the TWM PWT from the 2008 version in Appendix 3. However, figures 7A and 7B are comparable to the models figures developed by the TWM PWT. Feyrer et al. (2015) summarizes much of the current understanding of splittail biology and serves as the textual explication of the model.

Unlike salmon, splittail are expected to use the site throughout the year and all life stages are expected to be present on the restoration site. Different life stages will use the site differently, and some individuals may occupy the site for multiple years. Many young splittail will likely migrate offsite after a period of rearing, as that is their behavior on other spawning grounds. The primary benefit of Dutch Slough restoration will be to provide spawning grounds and early rearing habitat near where both genetic strains of splittail occur. In wetter years the Central Valley race primarily uses floodplains for spawning and the Napa River race use the freshwater connection of Suisun Bay to access spawning habitat there. In drier springs when floodplains are not available for spawning and the freshwater connection to wetlands of the Napa River are not available spawning is presently greatly restricted (Feyrer et al. 2015) and spawning and rearing habitat at Dutch Slough may be of great importance. Young splittail will occupy the restoration site primarily during the same time as young salmon (February-June).

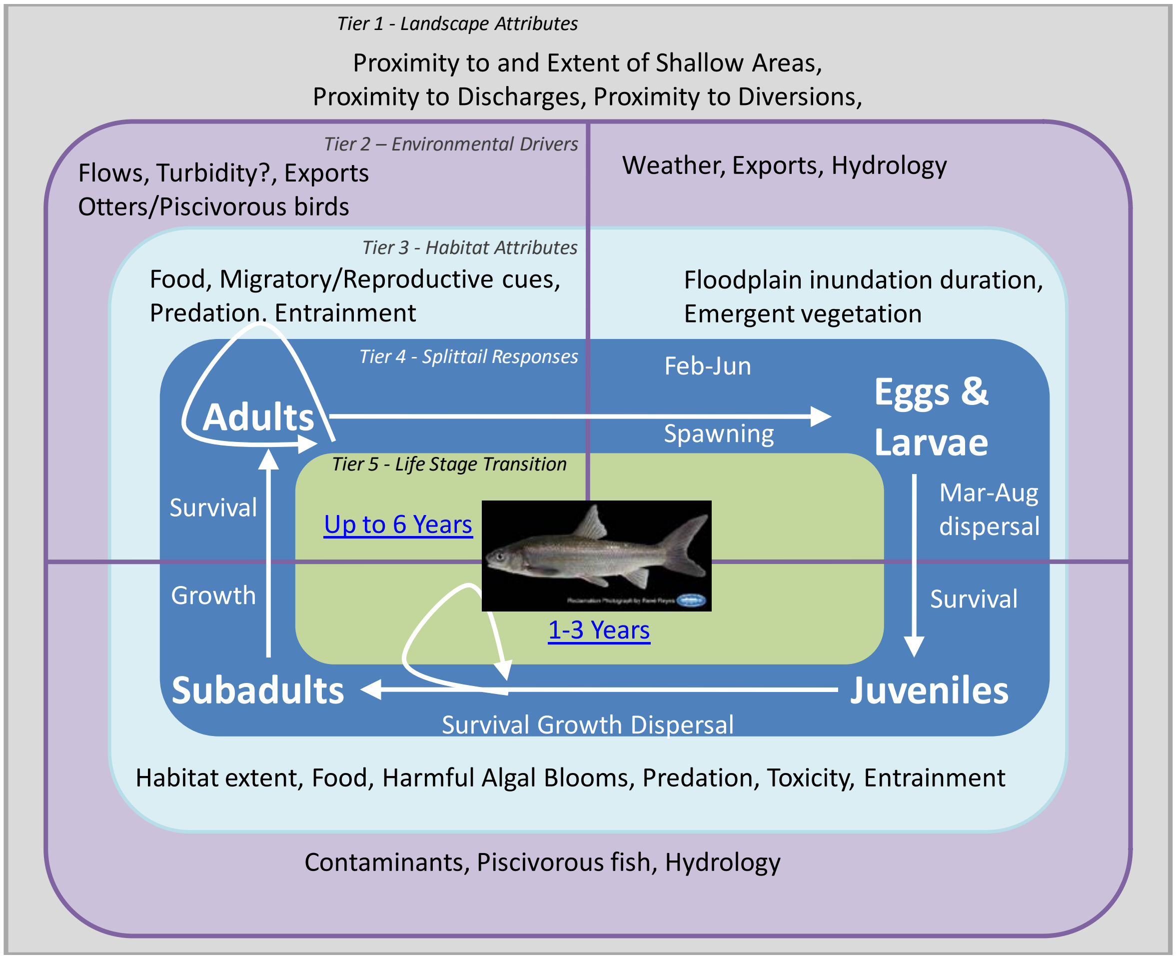


Figure 7A, General figure for conceptual model of splittail, based on format of the MAST Delta Smelt model Baxter et al. 2015.

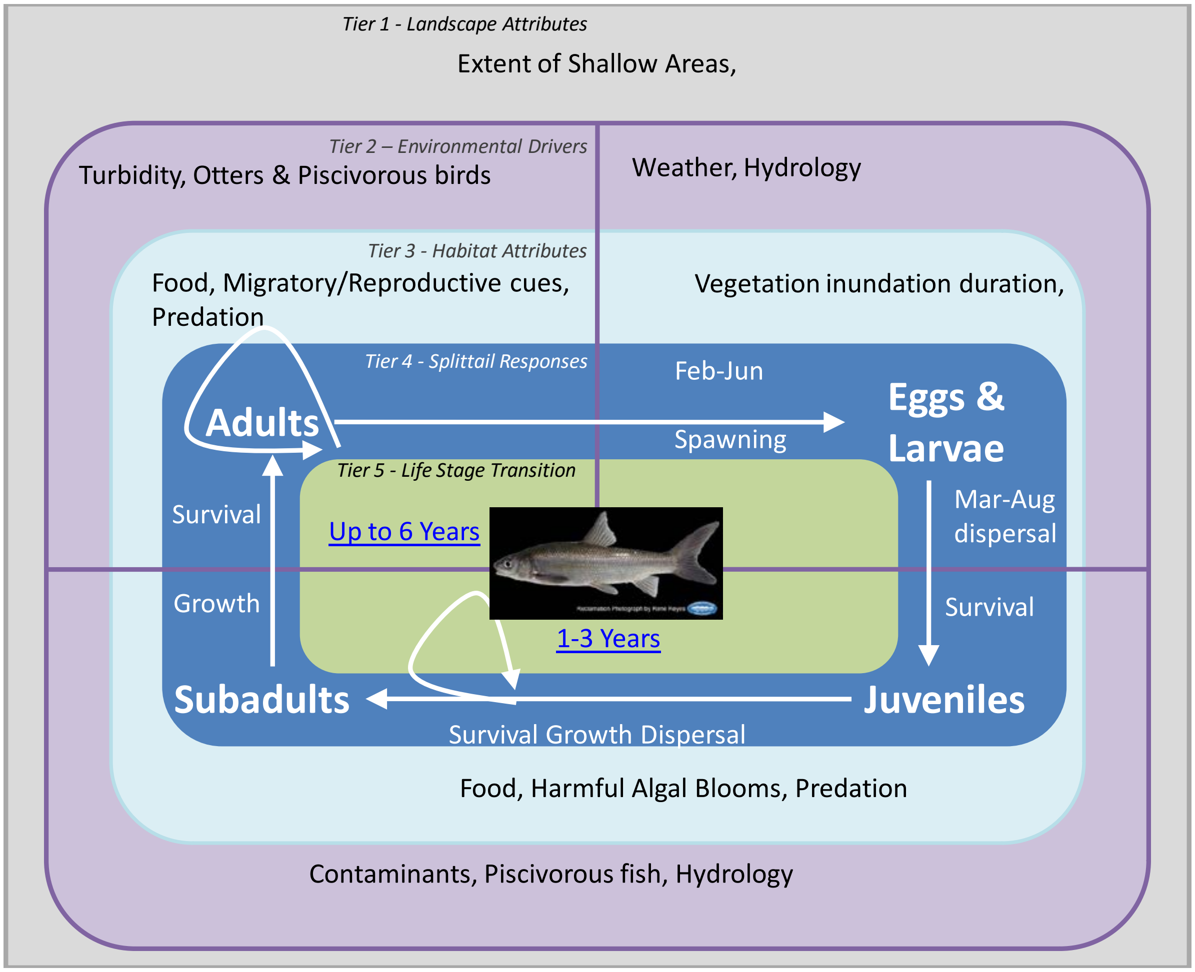


Figure 7b. Figure 7B Conceptual model figures for splittail modified for the location, history, and conditions of Dutch Slough.

Other models. The ecological process models that support these species models are included as Appendices 3 through 9 for wetland marsh dynamics, food web, aquatic vegetation, clam effects, contaminant effects, wetland evolution, and transport.

Monitoring Strategy and Analytical Framework

Performance monitoring to address PM1-PM7 will identify the status and trends of native species on the site and allow comparison with similar status and trends monitored elsewhere. Such monitoring and comparison will allow assessment of how the project performs as a restored ecosystem supporting native species and its likely ability to maintain that performance through time.

Performance of Dutch Slough as a functioning ecosystem and contributor to target fish growth and survival must be evaluated in the changing annual context of the broader landscape of hydrology, water quality, fish production and abundance. Data for evaluating the monitoring data to assess performance of Dutch Slough rests on three comparisons when fish are likely abundant on site from January through May:

1. Comparison with the size distributions of fishes of interest on-site and in nearby un-restored habitats. Data from long term fish and zooplankton monitoring conducted by Interagency Ecological Program (IEP) and academic programs, as well as newer fish programs, including UC Davis's North Delta Arc study, and the USFWS Liberty Island study, to assist in placing project effects in a landscape context . Most importantly, comparisons will be made with the wealth of data on the size, variance in sizes and abundance of young salmon and splittail in the USFWS juvenile fishes sampling program.
2. Compare changes through time as restoration proceeds and the ecological structure and functions evolve on site. This approach will gain particular relevance as the parcels are restored and data can show whether the age of the site or the annual fluctuations in weather and fish conditions are more important in the performance of Dutch Slough.
3. Comparison with results from other restored sites. Details of monitoring on other sites are beyond the scope of this plan. However, the consistent use of the standardized monitoring framework being developed by DFW will enhance the comparability of data gathered at Dutch Slough with data gathered on other sites.

Adaptive management monitoring will address critical uncertainties addressed in AM1-AM7 to help explain the observed status and trends and to develop answers to the critical uncertainties that the different elevations and cell sizes are designed to address. The difference between performance monitoring and adaptive management monitoring is twofold:

1. Performance monitoring addresses the entire restored site whereas adaptive management monitoring must be pursued on the scale of the eight test cells at the different elevations and sizes.
2. Performance monitoring is concerned only with the final outputs of fish abundance and condition, and the likely trends of the abundance and condition of fish on site. Adaptive management monitoring will determine the importance of different processes in driving the abundance and condition of fish on site.

Monitoring is also likely to identify issues that will require special studies to address unresolved processes or emerging new issues.

The Adaptive Management questions are addressed on site by comparing monitoring data collected in the eight different subunits that are of two different mean elevations and three different sizes. Figure 8 shows the eight cells of different elevations and sizes. The two largest cells are on Emerson and receive water both from their common connection to Dutch Slough and from the inflow from Marsh Creek at the southwestern corner of cell A. All other cells receive water from Little Dutch Slough. Cells 1 and A will be inundated first and immediately provide data on the development and function of tidal wetlands at the two different elevations. However, because they are of similar size they can allow no assessment of the impact of size on restoration site function. Subsequent inundation of the Gilbert parcel will provide further comparisons of the role of elevation, but will also allow comparison of two sites (2 & B) which are roughly ¼ the size of the other sites on that parcel (3 & C). Finally, inundation and monitoring of Burroughs parcel will add two more cells (4 & D), each at different elevations, but at intermediate size like 3 & C on Gilbert. Together these eight cells allow evaluation of differences due to elevation, size, and the intrinsic differences across the parcels.

Monitoring to address the adaptive management hypotheses is broadly similar to the monitoring of performance overall, but uses a finer spatial scale to allow comparison of different cells within the site. In addition, the value of comparing the different cells as their physical structure affect fish growth and survival is greatly enhanced by monitoring those factors hypothesized as responsible for differences among cells. Thus the adaptive management monitoring includes assessment in the eight different cells of food production throughout the year, the physical habitat differences, feeding success and predation risk in the months when young salmon and splittial are abundant on site.

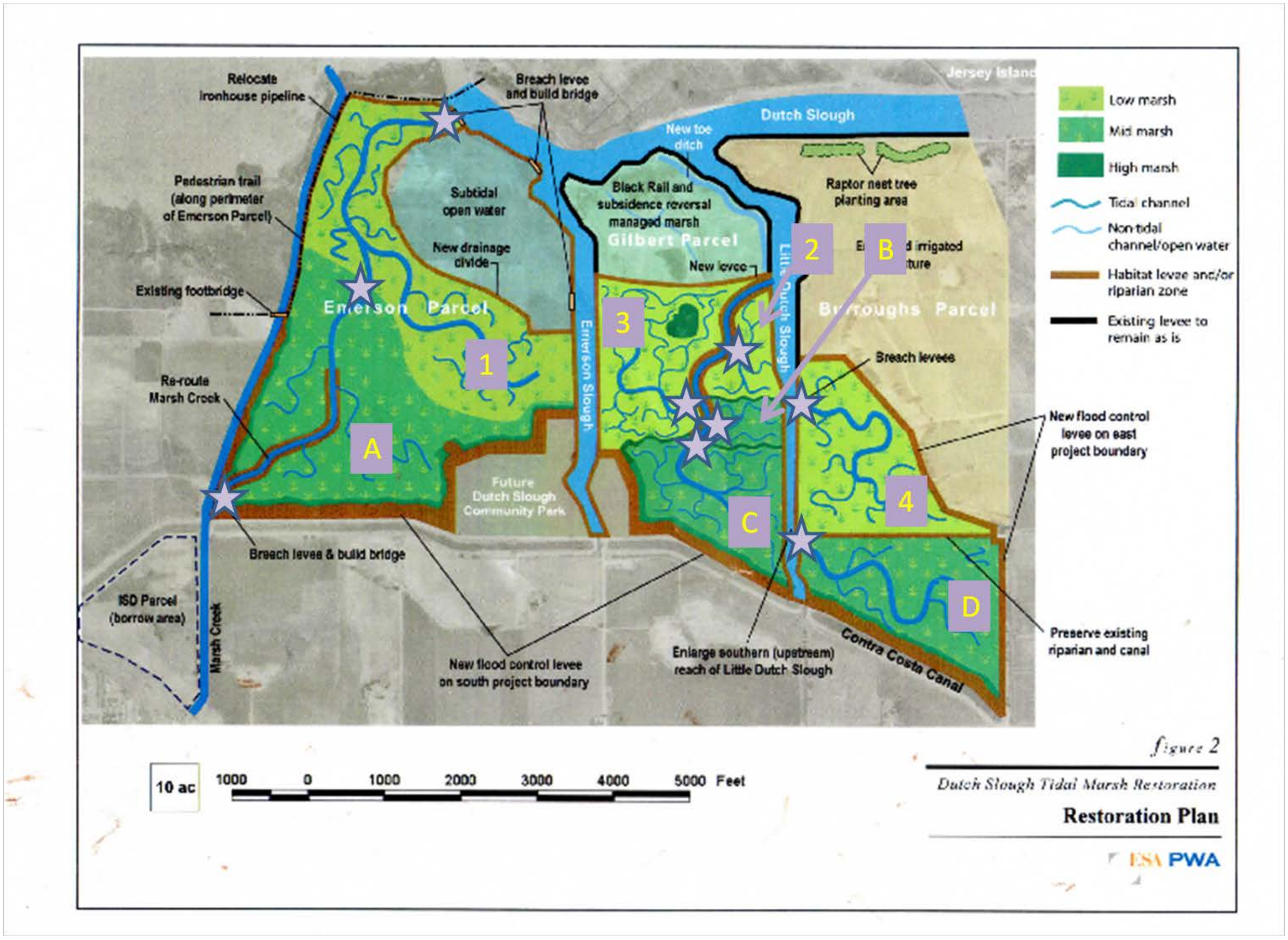


Figure 8. Experimental structure of restoration site showing the eight test sites; Numbers indicate low marsh elevation, Letters indicate mid-elevation marsh. A and 1 are largest in area, B and 2 are the smallest, 3,4 C,D are of middle size. Stars indicate location of sondes.

Sampling frequency and intensity

California’s Mediterranean climate has major impacts on species’ life histories and ecosystem productivity within the San Francisco Estuary (Figure 9).

Larval fish assemblages shift seasonally, as would be expected given the spawning habits of the adults (Feyrer 2004; Sommer et al. 2004). Outmigrating salmonids tend to arrive in the Delta in concert with weather and/or flow events in the late winter or early spring. Delta Smelt begin their migration from the low salinity zone into fresh water for spawning during the “first flush” of high turbidity water that accompanies early winter/late fall storms (Grimaldo et al. 2004, Grimaldo et al. 2009; Sommer et al. 2011). Longfin Smelt migrate from the ocean or San Francisco Bay into the estuary during late fall/early winter (Moyle 2002; Fish et al. 2009). During the summer, most young-of-the-year Delta Smelt have moved into the Low Salinity Zone of the estuary and Longfin Smelt have moved downstream into brackish water or to the ocean (Fish et al. 2009).

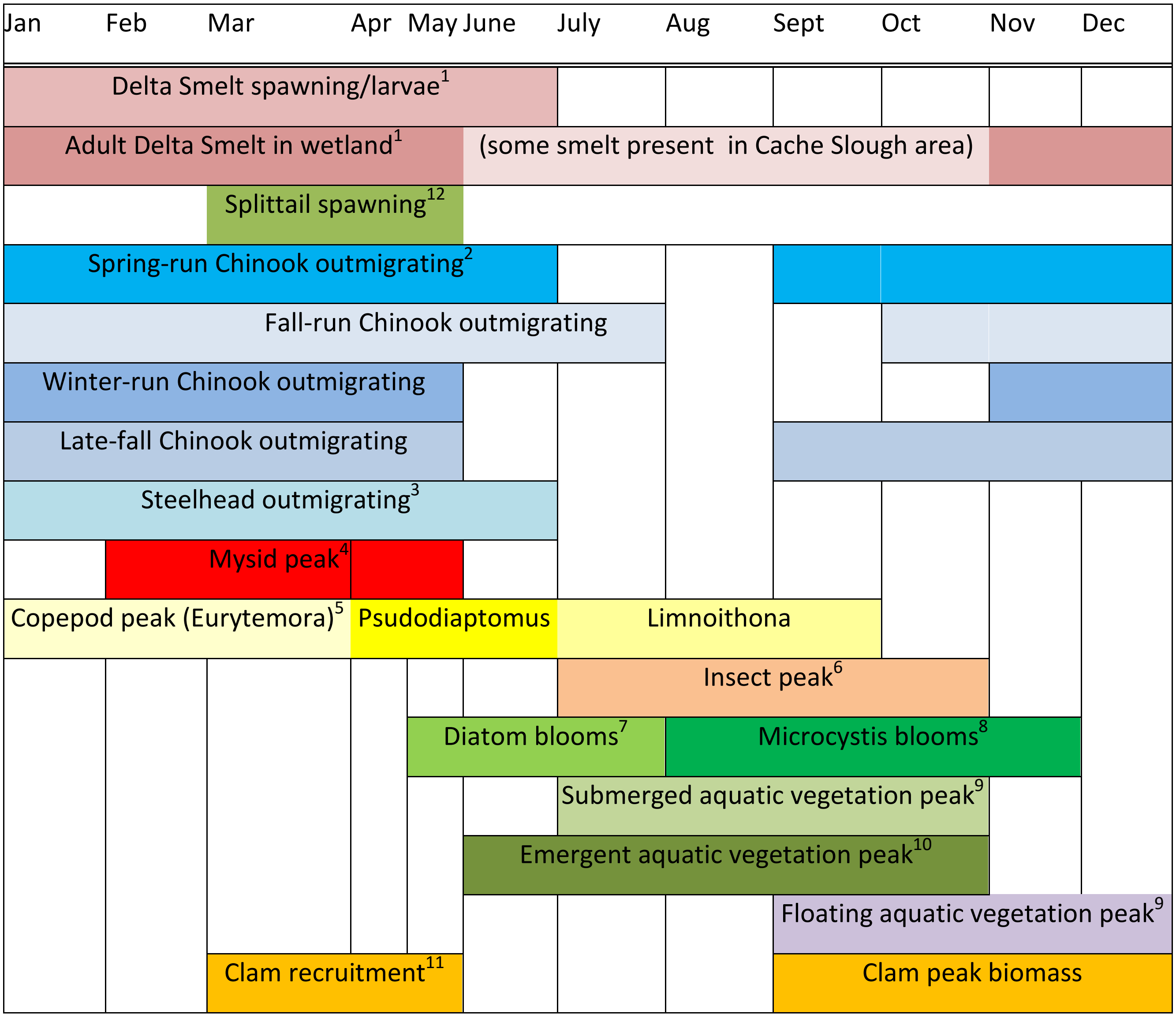
Changes in wetland primary productivity follow seasonal changes in temperature and hydrology. Lower average flows and higher residence times common from late spring through the fall allow greater phytoplankton production (Lucas et al. 2003, Lopez et al. 2006; Lucas and Thompson 2012). Summer is also the season of greater macrophyte growth due to longer day lengths. The macrophytes primarily considered here and in the Aquatic Vegetation conceptual model are herbaceous aquatic or wetland species including emergent vegetation, such as tules (*Schoenoplectus spp*.), floating aquatic vegetation (FAV), like water hyacinth (*Eichhornia crassipes*), and submerged aquatic vegetation (SAV), many of which are non-native and considered pests such as Brazilian waterweed (*Egeria densa*), grow vigorously in the summer months and often die back in the months when young salmon and splittail are present. While macrophytes provide relatively little immediately bio-available organic carbon when compared to phytoplankton, they do contribute to the food web via detrital pathways and are important in providing structural heterogeneity and supporting physical processes of wetland evolution (see Aquatic Vegetation and Wetland Evolution models).

Thus, the season of greatest primary productivity is distinct from the months when target fish populations are in the estuary. This dichotomy implies that the short term dynamics of primary production in the summer are less important to supporting the foodweb that salmon and splittail depend upon than the accumulation of secondary productivity available when the fish are on site in the winter and spring. Therefore, primary productivity, as it affects the target fish, can be adequately assessed from measurements at the peak and conclusion of the growth season, we use May and October to assess the annual variability in primary productivity.

The abundance and condition of fish in Dutch Slough will change rapidly from month to month and vary greatly across years. Fish abundance and condition in Dutch Slough will be impossible to evaluate without knowledge of fish abundance and condition elsewhere in the Delta (for example see Diefenderfer, Thom et al. 2011) For migratory species, abundance and condition of juveniles outmigrating through the Delta are inseparable from conditions upstream and the health and abundance of their parents. Thus, fish sampling must be intensive and frequent and techniques must be comparable to sampling at other sites, weekly or biweekly

In contrast, the habitat value of Delta wetlands changes slowly. Sediment pulses may be sporadic and of short duration, but the changes in the bathymetry of the channels and marshplain that affect fish use are of relatively large scale and long duration. Thus, less frequent and less precise monitoring can be sufficient to detect changes.

The routine long-term monitoring described here should be supplemented with special short-term studies as needed to test specific hypotheses of the relationships between fish, food, and tidal restoration sites.



1. Baxter et al. 2015, see Delta Smelt Model
2. Yoshiyama et al. 1998, and Kevin Reece (DWR), pers. comm, see Salmon Model
3. Moyle 2002
4. Hennessy and Enderlien 2013, M. Young, UC Davis pers. comm, see food web model
5. Hennessy and Enderlien 2013. Bollens et al 2014, see food web model
6. Howe et al 2014, M. Young, UC Davis, pers. comm. , see food web model
7. Glibert et al 2014
8. Lehman et al 2015
9. Hestir et al 2008, see Aquatic Vegetation Model
10. Miller and Fujii 2010, see Aquatic Vegetation Model
11. Corbicula and Potamocorbula; Thompson and Parchaso, 2012, 2013, see Clam Model
12. Feyrer 2015

Figure 9. Seasonality of organisms in the upper San Francisco Estuary.

Regional comparisons

Restoration sites can require special monitoring if they are likely to be affected by historical uses of the land on the site or by nearby stressors. The Dutch Slough site was a dairy farm for many decades. We are aware of no significant discharges, entrainment risks, upstream land use or nearby pesticide use that might require monitoring beyond that needed to evaluate its ecological effectiveness as a restored tidal wetland and fish nursery.

Dutch Slough is one of the first tidal marsh restoration sites of significant size in the Delta. Therefore, much of the monitoring we propose is new in this landscape. We target a planning horizon of only 3-5 years, after which the goals, methods, metrics, frequency and intensity of this monitoring program should be reviewed and compared with other monitoring programs that will have been designed and implemented subsequently. In addition, the current plan to restore the three parcels of Dutch Slough sequentially will allow monitoring plan revision and incorporation of new technologies as Gilbert and Burroughs parcels are restored

Metrics, Methods, and Schedules

Fish and their prey, monthly sampling January-May
Sampling in Dutch Slough is likely to be most informative when fish are most likely to be present at measurable abundances. The target fish species, salmonids, splittail, and smelts, vary greatly in their abundance from month to month. With four runs of salmon and a variable seasonality of Steelhead, young and adult salmonids can be found in the Delta in most months except August. However, the peak abundances of young salmon and Steelhead in the Delta is between the first storm flush after December and when Delta water temperatures rise above 25 C, usually by June. Splittail are resident in the Delta. The main invertebrate foods of young salmon and splittail vary greatly with short-term changes in the physical environment. Therefore, sampling recommended here is on a monthly time step for the months from January through May. In each of the five months sampling will be simultaneous for zooplankton, larval fish, and young fish, especially fry and smolts and adults of resident species.

Young salmonids and splittail are mostly found in the Delta from January through May, feeding primarily on detritivores and relying on primary production from the preceding year. Thus, the role of wetlands in feeding these target species is mediated through a benthic or detritivore trophic linkage that captures summertime productivity and delivers it to young fish four to six months later.

Larval fish sampling.

Adaptive management of Dutch Slough requires knowledge of the relative abundance, growth and survival of larvae in the different environments provided by the eight experimental units. Larval fish sampling can be done simultaneously with macrozooplankton/Neomysis sampling. The primary larval species of interest is splittail.
Field crews will sample larval fish during daylight using a 2m long net (500µm nylon mesh) with a 0.2m mouth diameter attached to a metal O-ring frame and a flowmeter attached in the

center. The crew will deploy the net alongside a canoe or kayak via a davit sample for 10 minutes. In channels too small for boats, shorter duration deployments from shore may be necessary. Upon gear retrieval, the crew will rinse the net from the outside with a water hose to wash down all contents into the cod end. The crew will remove, measure, and release any non-larval identifiable fish from the cod end, then pour the remainder of the cod end into a jar containing 10% buffered formalin and rose bengal. Lab personnel will identify all preserved fish to the lowest possible taxon; up to 100 of each taxon will be measured (fork length) and the remainder will be counted.

Small fish sampling.

Fish in the size range of 30 mm to 30 cm are the primary fishes of interest in Dutch Slough. This encompasses most salmon and young-of-year splittail but also includes adults of smaller species such as threespine stickleback, prickly sculpin, and various non-native gobies. In general, fish with higher condition (i.e. biomass per unit length) are thought to have higher survival, fecundity, and growth rate (Blanchard et al. 2003, Grorud-Colvert and Sponaugle 2006). Thus, much of the performance evaluation of Dutch Slough will rest upon the abundance and condition of juvenile fish in comparison with fish in the main river channels, on other restored wetlands, and between the eight different cells comprising Dutch Slough. The USFWS Juvenile Fish Monitoring Program samples fish weekly at the stations near Dutch Slough from January through June. Ideally fish sampling on site would also happen on a weekly basis to allow the best comparison with USFWS data, but every other week is probably adequate to determine if fish are using and benefitting from occupying the site.

All fish captured will be identified to species, measured and weighed. In addition to measuring fish, it is also important to monitor their condition using Fulton's condition factor (Anderson and Neuman 1996) . Fish condition provides information on the general health of fish and shows if at-risk fish species are in better condition than USFWS samples taken in nearby channels.

Fish sampling will always be accompanied by measurement of environmental and physical metrics at each location: specific conductance, temperature, DO, pH, turbidity, depth, and substrate composition. Water samples will be collected at each fish sampling site , filtered in the field and brought back to the lab for measurement of chlorophyll a concentrations. These grab samples will be compared with data from the nearest sonde. Sampling with seines will occur at repeated stations, where artificial substrates will permit sampling at all tidal heights. Sampling will occur at randomly selected sites within each experimental unit, stratified to include different depths and habitat types.

Seines. Varying largely with the timing and degree of winter and spring storms, salmonids enter the Delta in some years more as smolts or as fry. In dry springs splittail spawn in upstream areas, in Napa Marsh, or in fringing wetlands in the Delta and produce small year classes in such years. In wetter springs when the Yolo Bypass is inundated extensively, production of young splittail is usually several orders of magnitude greater than otherwise (Moyle et al. 2004). Thus, interpretation of fish size and abundance as a measure of Dutch Slough performance requires knowledge of their size and abundance in nearby areas in each year, as sampled with comparable gear.

Juvenile fish are sampled throughout the Delta by the US Fish and Wildlife Service using both trawls and seines (Figure 10).

The channels of Dutch Slough will be too narrow and shallow to allow boats and trawls, so comparability of data will require the use of 15m beach seines. Because most of the beaches in the Delta have been rip-rapped, USFWS beach sampling occurs often on boat ramps. The channels of Dutch Slough will not be rip-rapped, but their U-shape and friable nature will make seining difficult and less comparable with the USFWS data. Thus, part of the restoration design in each unit will include a permanent sampling substrate similar to the boat ramps used by USFWS. On average the USFWS program samples about 47 cubic meters. There is a wide overall variability in the volume sampled from site to site, but for the sites near Dutch Slough the range in volume during 2012-2015 was a usual maximum of 120 m^3 and a usual minimum of around 30 m^3.

To sample comparably to the USFWS, the sampling substrates in Dutch Slough will extend along the channel for 15 m and extend into the channel about 10 m with a grade of about 15 cm for each m. They will be made of interlocking precast concrete planks with a textured upper surface to allow traction for pulling in seines (Figure 11). In addition to allowing high-quality data gathering, these solid substrates provide work and rest areas for the field crews. Sample locations of the substrates on Emerson Parcel are illustrated in Figure 12. They are located in such a way as to be out of sight and inaccessible from the public walkways, near the center of each test unit, in proximity to each other, and close to the mouths of small channels where young fish are likely to aggregate and feed. Such sampling substrates could also allow relatively secure short term storage of trawls and other bulky gear and thereby facilitate sampling in difficult weather. At one end of the substrate and on the opposite shore, pilings will facilitate the placement of fyke nets (discussed below).

Beach seine sampling will occur during daylight using a 15m long x 1.2m high (3mm delta square mesh) net using protocols developed by the USFWS (Speegle, Kirsch et al. 2013). One person will walk out into the water (up to 1.2m in depth) holding one end of the net to measure the width and depth of the seine site. The second crew member will walk to the first crew member and place their seine pole where depth was recorded. The first crew member will walk parallel and the length of the shore and note seine length and site depth. Both crew members will haul the beach seine up on the shore, leaving the cod end in water. The crew will fill a tub with water and place the cod end in the tub along with any fish caught in the “wings” of the seine. The crew will measure and weigh up to thirty fish of any single species (fork length), and count and identify all fish. Abundance of all shrimp and jellyfish species will be estimated.

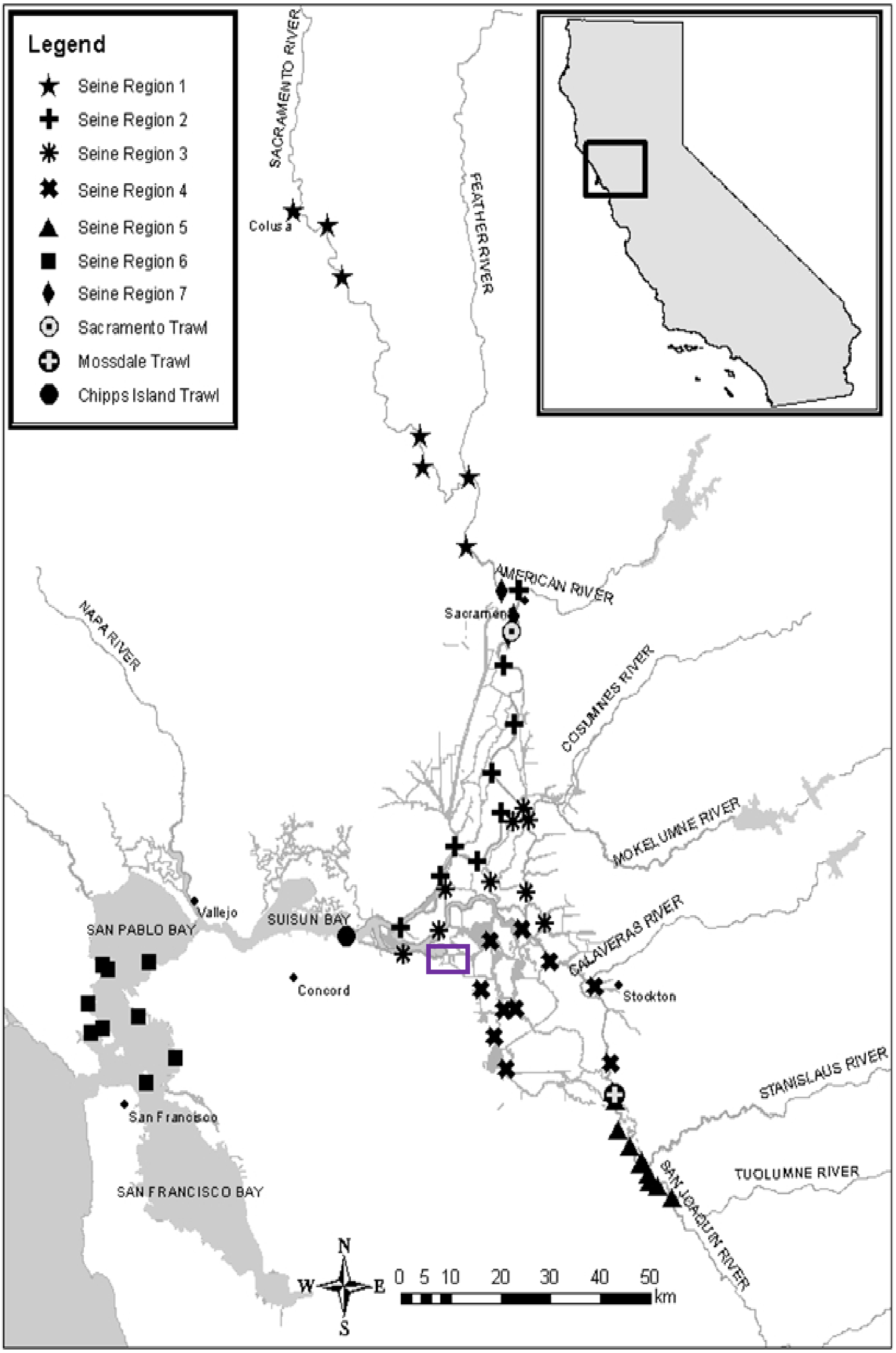


Figure 10. . Locations of USFWS sampling sites in the Delta.

Purple box surrounds Dutch Slough.

Figure 11. . Artificial sampling substrate modules of precast concrete to permit seine sampling within Dutch Slough in a manner similar to that performed by the Juvenile Fish Monitoring Programs of the USFWS. These materials are usually used to construct boat ramps

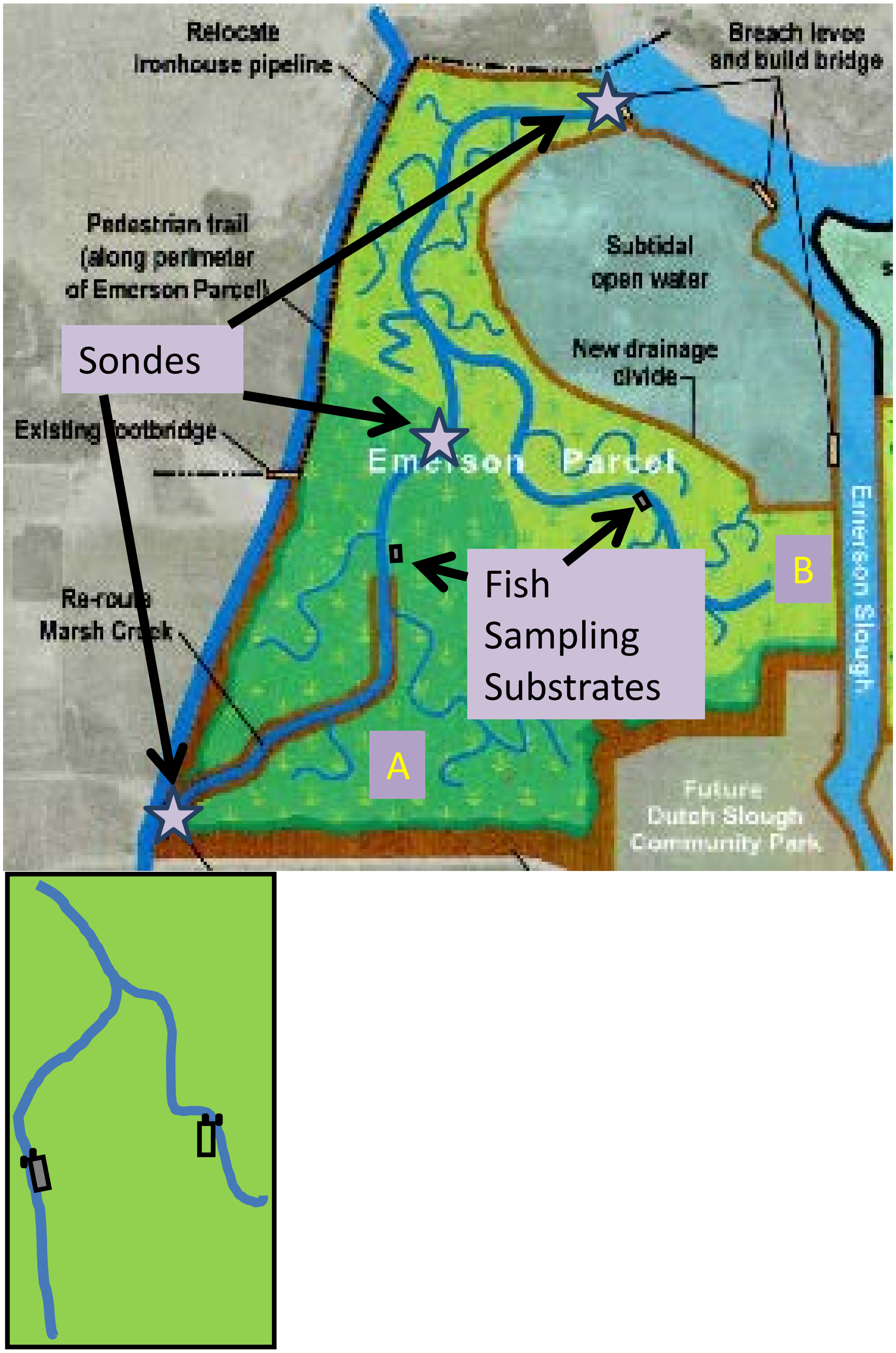


Figure 12. Locations of water quality sampling gear (Sondes) and artificial fish sampling substrates and facilities on Emerson Parcel. Cartoon on right shows placement of 2 pilings for fyke net anchoring in relation to the artificial substrates.

Not all the fishes or environmental conditions of interest will be adequately sampled by the 15 m beach seines. In particular, the adaptive management questions about Dutch Slough require sampling fish at different stages of the tide and in different habitats. Thus, smaller, more flexibly applied gears are needed. However, no such gears are broadly used in the Delta. Conclusions from such novel gears are limited to data gathered in different areas or at different times within Dutch Slough, or in comparison to data gathered in other restored sites using such gear.

Mobile sampling gear. Sampling in the flats and edges of marshes in Louisiana is often done with throw traps (Rojas and Minello 1997). Despite high inter-operator variance, cast nets yield data very similar to throw traps and can be deployed in some types of topography where throw traps are less suitable (Stein et al. 2014). Studies underway by UC Davis have shown the usefulness of cast nets, in sampling small, shallow waters. Field tests by DFW members of the TWM-PWT are ongoing and may soon provide guidance of the advantages and disadvantages of each gear. Lacking such guidance, we will use both for the first three years and then assess effectiveness.

Cast net sampling has been used sporadically by the North Delta Arc study and a tidal marsh study occurring in Petaluma River and Napa-Sonoma marshes, but has otherwise been little used in the San Francisco Estuary. UC Davis follows the protocol developed by Edo (2008). The cast net will have a 1.5 - 1.8m diameter mouth opening with 4.7mm mesh size. The net is deployed in shallow areas, where the net is allowed to sink to the bottom and then retrieved. Experience and collaboration with the standardized protocols being developed by the TWM-PWT may suggest changes in the gear and methods. One survey in each experimental unit each month from February through May with a minimum of 6 successful fish samples per survey. Sampling will occur in stratified random sampling locations within each experimental unit where depths < 2.5 m.

Fyke net sampling. Fyke nets are funnel shaped nets, secured to each side of a channel and deployed so that flow carries fish into the net (see Figure 13). In streams they are often deployed for up to 24 hours, but in a tidal system like Dutch Slough deployment should extend only over one falling tide, no more than 4 hours.

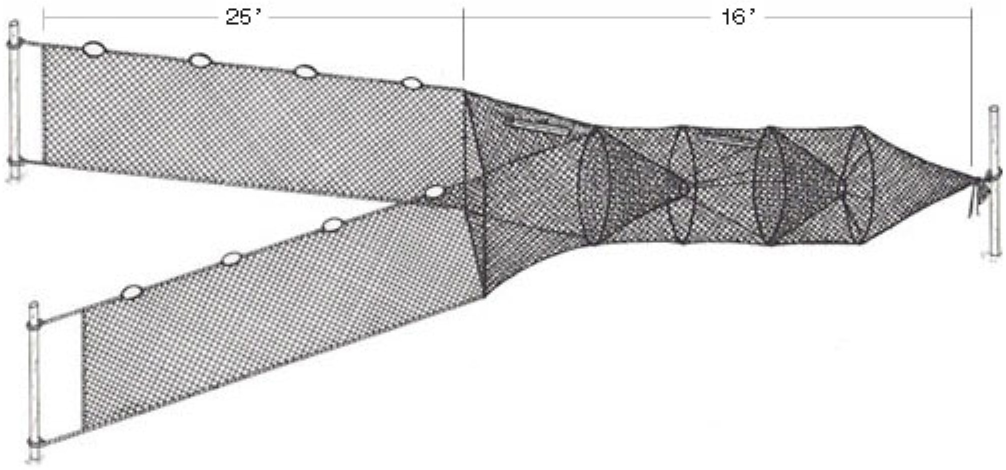


Figure 13. Typical, commercially available Fyke net.

Fyke nets are currently used occasionally by UC Davis in various areas in Suisun Marsh and the West Delta Arc (DeCarion 2014). The gear is deployed in high tide depths < 0.9m with the following gear dimensions: 1.8m x 1.2m net composed of ¼" stretch mesh (#126 knotless delta). This gear is typically deployed in shallow channel areas, but can also be placed along shallows. UC Davis follow the protocol developed by Lake (2013). They perform one survey per month in each experimental unit from February through May with a maximum of 4 samples per survey. Sampling will occur in channels < 5m in width and depths < 1.0 m during high tide.

Fyke net stations are at the bayward end of each artificial seining substrate to permit standardized sampling of young fish in the channel. Comparison will be made of relative catch and condition of fish in the Fyke nets and the mobile sampling gear to assess sample replicablility and effectiveness of each.

Adult predator sampling. In cooperation with DFW, electrofishing of Dutch Slough and Little Dutch Slough will be done every May to assess predator abundance as measured by catch per standardized sampling time unit. Such sampling can give the relative predator abundance and size distribution in different parts of the site and in different years. Stomach contents can be compared with data from small fish surveys to determine predator selectivity and overlap of the fish community on site with that in the predator stomachs. In addition or instead, creel censuses and public involvement in a local 'black bass' fishing derby each May could be used to acquire specimens and build public involvement in the project. Data gathered will include relative abundance, length and weight, and external health indicators. These data could identify the need for targeted studies on predation, fish health, or contaminants.

If predator distributions and abundance are not adequately assessed by the above measures, a small electrofishing boat should conduct a single pass for each identified habitat section during daylight. Depending on specific conductance, voltage will be set to 50-500 volts operating with 3-6 amps (following Brown and Michniuk 2007). Field crews will adjust electrofishing settings to stun the fish and avoid injury and mortality. A crew of three people (one boat operator and two crew members) will sample each section and collect all fish by a dip net (5mm mesh) and place them in a live well. The crew will measure and weigh up to thirty fish of any single species (fork length), and count any additional fish. They will also record the habitat type (beach, SAV, EAV, mudflat, etc) in which they collected the fish.

Productivity Sampling in May and October

Due to the Mediterranean climate of California, there is a marked disconnect in the seasonality of primary productivity and the seasonality of use of the site by target species. Most primary productivity occurs in the late spring when nutrients, temperatures, and insolation are high. By October, productivity declines with decreasing temperature and insolation. Thus, the lower trophic dynamics will be captured by samples in just two months, May and October. Data collected in May give a measure of the maximum productivity the site and samples in October estimate the productivity accumulated and retained on the site when the growing season is coming to a close. Measures of primary production in these months also provide estimates of primary production that is available to nearby pelagic foodwebs during the summer and fall seasons when Delta Smelt might be expected to benefit from such trophic augmentation.

Nutrient concentrations, if nutrients are limiting, can control primary productivity. Even if concentrations are not limiting, changes in the concentrations of nitrogen and phosphorus reflect nutrient uptake and loss through emigration from the site or by sequestration in the benthos. Since most primary productivity happens in the warmer, sunnier months nutrient impacts are less expected during the months of high salmon abundance and splittail growth. Therefore, water samples will be taken In May and October from three random sites within each of the eight test units (1-6 and A & B) and from three random sites in Little Dutch Slough, Dutch Slough and the tidal reach of Marsh Creek (that will shift in geographic location with creek outflow). In the field pH and Dissolved oxygen concentrations will be measured at the time and location of water samples. Standard EPA lab tests will be run within 24 hours for dissolved organic nitrogen, nitrate, ammonium, total phosphorus, soluble reactive phosphorus, total organic carbon, dissolved organic carbon and particulate organic carbon. Measurement of chlorophyll a and phaeophytins in these grab samples can be used to calibrate the mounted sondes. If nutrient effects are observed, later monitoring may need to extend nutrient sampling into more months.

Benthic sampling**.**

Benthic invertebrate sampling has been done in the San Francisco Estuary as part of the Environmental Monitoring Program since 1968. Ponar dredges used in the broad expanses of Suisun Bay and the Delta are heavy (24 Kg empty and 34 kg full), requiring boat deployments and winches. Dutch Slough channels are much smaller and shallower so other sampling tools are needed. Simple coring devices made from Polyvinylchloride (pvc) pipes, that may be twisted or hammered into the substrate can be used to good effect (See Figure 14). A suitable corer is at least 10cm in diameter and capable of penetrating the substrate to a depth of 20 cm. A simple corer may be constructed out of commercially-available PVC pipe (Fig. 5)

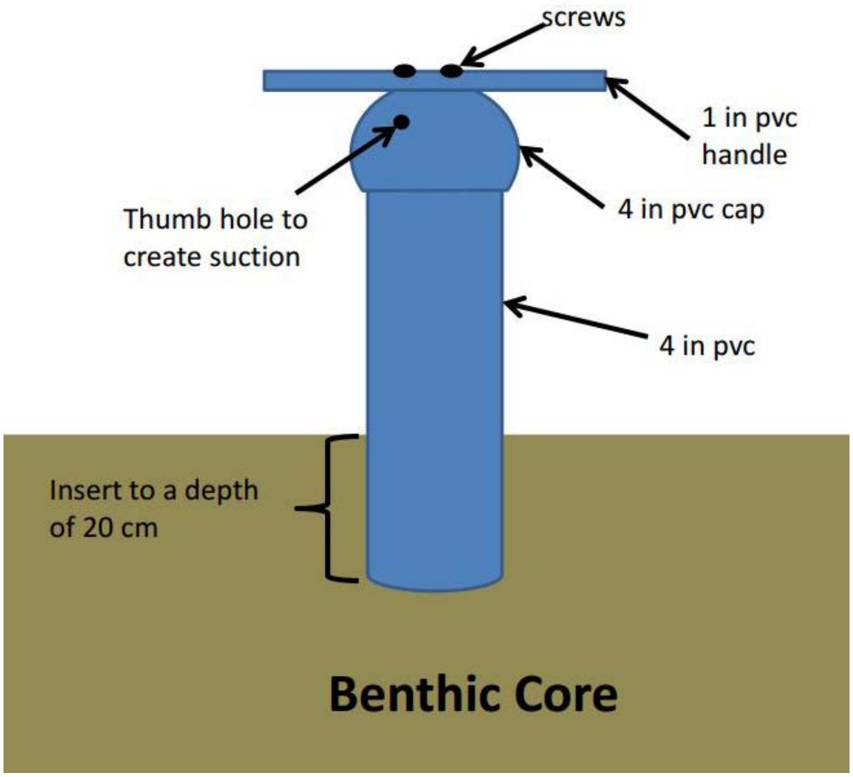


Figure 14. Four inch (10cm) benthic core easily constructed out of PVC pipe.

Epibenthic and Epiphytic sampling
Leaf packs consisting of standardized quart size bags of tule leaves may be used to sample benthic algal growth, identify benthic invertebrate community composition and assess relative benthic invertebrate abundance. Simple plastic dish scrubbers have been shown to have comparable collection results. Hester-Dendy artificial substrates (see Figure 15) can be used. Initially both should be deployed and results compared before settling on the final approach. Pilot efforts by DFW indicate that there may be a longer set time required before organisms settle on the plates of the Hester-Dendy sampler and that the species composition and relative abundances are more variable, but it has a broad history of use elsewhere. Neither gear has been deployed to much extent in the Delta. Either type of sampler should be placed in an area in which low water mark will not expose it. Sampling arrays will include one channel and two vegetated littoral habitats in each experimental unit, with sites selected randomly for each survey. The sampling array will be anchored to a brick and deployed so that it will remain submerged the entire duration of deployment (4 weeks). By placing a 500-micron net under the sampler, any organisms that may detach during removal will be captured.

To quantify bias in invertebrate community composition, sweep net samples will be collected in the same area as the artificial substrates. Sweep nets provide a less quantitative sample, but frequently have the highest species richness of any epiphytic sampling method (Blocksom and Flotemersch 2005). We will sweep a standard D-frame kick net through the water 5cm from the bottom for 3 sweeps of a distance of one-meter each. These samples will characterize community diversity in conjunction with the biomass estimate from the artificial substrate.

Figure 15. Figure 15. Hester Dendy artificial substrates for sampling benthic algae and invertebrates. The plates are three inches in diameter and easily fit into a quart Mason jar for ease of preservation for later analysis in the lab.

Zooplankton sampling

Planktivorous fishes such as Delta Smelt, Longfin Smelt, and juvenile striped bass rely on zooplankton for a large percentage of their diet (Feyrer, Herbold et al. 2003). However, they preferentially consume large meso- and macro-zooplankton such as calanoid copepods and mysid shrimp. Introduction of several non-native zooplankton to the region such as *Limnoithona tetraspina* (a small cyclopoid copepod with low nutritional value that now dominates the low salinity zone of the estuary) may be competing with larger zooplankters (Gould and Kimmerer 2010). Other introduced species, including the Asian calanoid copepod *Pseudodiaptomus forbesi*, now constitute the most important food source for adult smelt (Lott 1998). Declines in zooplankton were implicated in the Pelagic Organism Decline (POD) (Baxter, Breuer et al. 2010). Thus, for assessing the contribution of Dutch Slough to the trophic web of nearby channels (Restoration Hypothesis 4), it is essential to quantify both the quantity and quality of zooplankton both in the wetland and exported to the surrounding sloughs.

Zooplankton nets have been used extensively for measuring zooplankton community composition and biomass throughout the estuary, so our data will be comparable to similar efforts conducted by the DFW's long term monitoring programs, USFWS's Liberty Island monitoring, and UC Davis' Suisun Marsh monitoring program. To ensure comparability with other sampling for zooplankton done in the channels and on other restoration sites the gear and protocols under development by the TWM-PWT will be followed to the extent possible. Field crews will sample zooplankton during daylight from the levee breach connecting Dutch Slough to the nearby channel. On Emerson deployment can be done from the bridge; on Gilbert and Burroughs deployment will need to be done from the levee or from a boat anchored in the breach. Three deployments will be made on an incoming tide and another three 6 hours later on an outgoing tide. The nets will be cast in the direction of tidal flow and retrieved against any tidal current.

Crews will use one Clark-Bumpus net 1m long x 0.127m mouth diameter (153µm mesh) to measure mesozoplankton. The net used for measuring larval fish will also sample macrozooplankton (a 2m long net with500µm nylon mesh with a 0.2m mouth diameter attached to a metal O-ring frame and a flowmeter attached in the center). After retrieval, the crew will rinse the net from the outside to wash down sample into the cod end. All content collected in the cod end will be preserved in 10% buffered formalin and rose bengal. Crew will remove any fish that are visible in the sample before preserving. Laboratory personnel will identify a minimum of 6% of the sample to the lowest possible taxon in the lab either using a microscope or by photographing samples and using automated image recognition software (i.e. ZooImage, http://cran.r-project.org/web/packages/zooimage/index.html, as cited in Gislason and Silva 2009).

Because PM 4 is concerned with export of food to fish in the nearby channels, monitoring of small zooplankton and of phytoplankton is not included here. If future conditions warrant such sampling of non-food items protocols are in available from DFW.

Benthic and Littoral invertebrate sampling

The target species for on-site benefits of Dutch Slough restoration are salmon and splittail, that feed benthically or littorally, so channel sampling with a trawl is inappropriate to measure their food abundances and difficult in the shallow habitats of Dutch Slough. Pumped samples can help evaluate the actual prey populations available for salmon. Pumped samples are used in the channels of the delta to sample microzooplankton, but can be used with the same 0.505mm mesh size as used in trawl sampling for macrozooplankton.

At each randomly selected site the crew will lower the intake near to the bottom, turn on the pump, and raise the intake and move it slowly into the fringing vegetation. They will discharge this water into a plankton net (35 µm) to concentrate the larger volume sample. They will stop pumping when approximately 19.8 gallons have been collected and record the sample volume from the flow meter. After pumping, they will rinse the outside of the plankton net into the cod-end jar. They will preserve the contents of the cod-end in 10% formalin and rose bengal (from IEP zooplankton methods, Hennessy 2009). Laboratory personnel will identify a minimum of 6% of the sample to the lowest possible taxon in the lab either using a microscope or by photographing samples and using automated image recognition software (i.e. ZooImage, http://cran.r-project.org/web/packages/zooimage/index.html, as cited in Gislason and Silva 2009).

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Water quality monitoring and Wetland Habitat Assessment

Assessing basic water quality entering and leaving each parcel is essential either as direct information for most hypotheses, or as context for all others. Sondes such as the Yellow Springs Instrument **YSI 6600 V2-2** (see Figure 16) measure conductivity, temperature, dissolved oxygen, pH, turbidity and chlorophyll a. They can be either connected to cell phones for data transmission or used with data loggers that can be periodically downloaded.

For adequate characterization of Dutch Slough nine sondes are needed (Figure 8). However, since the parcels will be restored over the course of several years, only three are needed at first for Emerson. Sondes at each of the five levee breaches where water will enter the parcels (3 on Emerson, 1 on Gilbert, and 2 on Burroughs) will provide essential information for knowing the external conditions controlling many chemical and biological processes and how restoration alters those conditions. The additional four sondes within Gilbert will allow comparison of water quality conditions affected by the different sizes and elevations of the experimental arrangements of the restoration plan. On both the broad parcel level and the smaller test unit level, the effects of restoration on conductivity, temperature and turbidity of the water that enters and leaves on each tidal cycle are likely to be different.

Such sondes are subject to vandalism. However, the configuration of Dutch Slough allows minimization of such risk. Public access is largely restricted to Emerson parcel where there are bridges at each of the locations where water enters the parcel. Such bridges afford permanent attachments in securable and hidden spaces, reducing the risk of vandalism. Gilbert and Burroughs parcels exchange their water through three breaches into Little Dutch Slough. Little Dutch Slough is far from public access and supports dense vegetation that deters recreational boaters, thus reducing the risk of vandalism.

Figure 16. Six parameter YSI Sonde 6600 v2-2, 50 cm in length and 9 cm in diameter.

Physical nature of sites and Wetland Habitat quality

Channel evolution, vegetative growth, and substrate composition, when combined with water quality conditions, largely define the suitability of the site as habitat for the target fish species and for the secondary productivity on which they depend. Processes that change the physical nature of the site are generally slow enough in action to allow adequate characterization on an annual basis. Also, much of the relevant information is already gathered by others or is easily gathered as part of other sampling.

Site evolution will affect wetland productivity (e.g., via changing water residence times; Sommer et al. 2004) and suitability of fish habitat (e.g., turbidity; Hasenbein et al. 2013). Thus it is important to track changes as the site evolves, not only to assess the changing performance of the project, but also to inform adaptive management and design of future restoration projects.

Standardized Assessment of Wetland Structure

State and federal interest in the importance of wetlands has led to adoption of some quick and comparable evaluations of the 'health' of wetlands. The California Rapid Assessment Method CRAM (www.cramwetlands.org) is a standard wetland monitoring and assessment tool that is used throughout the state as a validated "level 2" evaluation in the EPA three-tier wetland monitoring framework. This rapid assessment combines considerations of site ecocline position with measures of hydrology, physical structure, and biotic structure and facilitates placing the development, or lack thereof, of a restoring wetland into a landscape context. CRAM results may also be used as factors in statistical models to characterize aquatic community composition. CRAM requires a team of 2-3 trained practitioners less than 3 hours to assess a wetland area and is needed at most once a year.

Aquatic vegetation mapping

The spatial extent and species composition of aquatic macrophytes and channel-edge emergent vegetation will be estimated when possible during the course of fish sampling in order to improve our ability to predict fish assemblages based on habitat. Characterization of the overall available habitat structure provided by aquatic vegetation will utilize existing data, augmented as needed with site-specific surveys. Our protocols will be consistent with methodology being developed for vegetation surveys by DWR's Aquatic Ecology section, and may include vegetation transects, digitization of true color 1-foot resolution aerial photography (CDFW VegCAMP), and/or analysis of hyperspectral imagery (e.g., Hestir et al. 2008). In the near future use of video drone overflights would allow even higher resolution aerial surveys at low cost, but such technology is not yet authorized for use here.

Substrate and Sediment dynamics

Substrate composition is important both as a broad indicator of the nature and evolution of the site and as a small scale indicator of the conditions associated with fish capture and vegetation growth; thus periodic measures of substrate composition can and should be combined with fish sampling efforts. Rough visual assessment of substrate should be taken at each sampling site, to correlate with data on fish, infauna, and benthic algae. But annual measurements using a series of gradated filters are necessary to determine the slower and less obvious changes of site evolution. Sediment samples will be assessed by weighing the contents of each filter for the percentages of each of four size categories, as identified in Table 2.

Table 2. Substrate Sizes (modified from Dunne and Leopold 1978)

Particle Size	Substrate Type
<0.06	Silt
0.06-5mm	sand
6-50 mm	gravel
51-300 mm	cobble

Data quality, management, and dissemination

Data management is a rapidly evolving subject of interest in the Delta. The State and Federal Water Contractors Association is working closely with the State Water Resources Control Board and with the Interagency Ecological Program Data Management Project Work Team to establish a transparent and standard data management process. For restored wetlands there is at present no standardized data management system. The IEP's Tidal Wetland Monitoring Project Workteam is developing procedures that will encompass the needs of the upcoming restoration projects, including Dutch Slough. We expect that by the time data begin to be gathered at Dutch Slough standardized data management procedures will be available.
We will develop a full quality assurance / quality control plan before monitoring begins. We will develop standard operating procedures (SOPs) for all field sampling, laboratory processing, and data entry activities. Where possible, Dutch Slough SOPs will match those of long-term regional monitoring programs to maximize data comparability. We will collect metadata at all stages of data collection and processing, and store metadata in standard formats along with the data. We will use a server-based relational database to organize all project-related data and metadata, and to cross-check all manually entered data for transcription errors. We will identify spurious data points using raw data scatter and box-and-whisker plots, and we will deal with any outliers identified by this method on a case-by-case basis, with full records of any changes. We will summarize monitoring data in annual reports, along with any analyses completed to-date. Data, data summaries, analyses, and/or reports will be shared with other researchers and the public via one or more wetland inventories or hubs (e.g. EcoAtlas, Estuarine Portal).

Data analysis

Once gear types and sampling regimes have been finalized, we will relate monitoring metrics to expected restoration outcomes/hypotheses.

In general, the Performance Measurement Hypotheses will be evaluated each year by comparing the data gathered on site compared to the data from the USFWS juvenile fishes monitoring program and the data gathered on site through time. If, as expected, food production and feeding opportunities are better for fish on site than in nearby waters, median fish length and biomass should be greater on site than in the nearby channels. If the restoration site provides refuge from predatory fish than nearby channels, the size distribution of fish on site should be more normal than in nearby channels where predation rates are greatest on smaller individuals.

Comparisons will primarily be with fish data from USFWS beach seine data from the Delta and with data from Dutch Slough as it changes through time. The TWP PWT is currently developing strategies to permit comparisons across different restoration sites and with the data from existing remnant tidal wetlands in the Delta. Results of the PWT's work will be incorporated into analysis of data from Dutch Slough.

In general, the Adaptive Management Hyoptheses apply the same approach as the Performance measures, but compare across the different cell elevations and sizes instead of with fish collected from nearby channels. In addition, measured differences in food production and availability (as primary production, pelagic, epibenthic, and littoral) will help identify how the restoration design and foodweb affect the target fish species.

Gear types will undergo efficiency evaluations, particularly efficiency calibrations similar to the USFWS's recent beach seine efficiency evaluation.

Deliverables

Annual reports on the work performed, data and data summaries and lessons learned will be provided to the TWM-PWT, funding agencies, and other interested parties.

As data are analyzed, conclusions will be shared first with the Tidal Wetlands Monitoring Project Work Team and other FESSRO monitoring efforts, so that spatial and temporal contexts can be developed. Such context will allow the identification of results from Dutch Slough that are part of large-scale variations from year to year and which are ascribable to conditions and restoration work on Dutch Slough.

As adequate data are collected to allow statistical tests of the various hypotheses, these conclusions will be published in peer-reviewed journals. How long it will take to acquire adequate data will be determined by the number of fish collected, the variability from year to year in weather conditions, and the how well the measurements on site and in nearby channels accurately portray the differences in median size and size distribution. Data from Yolo Bypass are encouraging that the impact of wetland access can be readily measured in a short time frame, but no such work has been done yet in the Delta for Tidal wetland effects.

Adaptive Management

The purposes of this project, in terms of this monitoring plan, are to benefit the target species and to contribute to adaptive management. Management works in short medium and long-term time scales and this project should help guide management at all three scales.

Short term adaptive management

Some preliminary conclusions from the inundation of Emerson in 2016 can be expected to provide guidance for other projects in the Delta and perhaps some final design aspects of Gilbert and Burroughs:

1. Did areas with tule plugs inserted before breaching show better vegetation coverage, especially at lower tidal elevation, than nearby areas without plugs? This design element was much discussed and had large effects on the implementation of restoration on Emerson. Future restoration efforts should be greatly influenced by early results here.
2. Did the channel design on Emerson reduce the coverage of floating aquatic vegetation, compared to nearby unaltered areas, such as Little Dutch Slough? Was the deep pond at the north end sufficiently deep to reduce *Egeria* abundance relative to nearby channels, such as along Dutch Slough and in Big Break? Considerable concern and skepticism remain about the ability of tidal wetlands to function without chemical or mechanical control of aquatic weeds.
3. Did target fish access the restored site and get captured by seines in densities similar to nearby stations of the USFWS juvenile monitoring program ? A fundamental assumption behind management's direction toward tidal wetland restoration is that target fish species will find and use restored wetlands. Dutch Slough is near the migratory pathways of all salmonids of the Central valley and the area of overlap of both strains of splittail, occurrence of target fish on site is a basic step in validating this assumption of wetland value.
4. Are target avian, mammalian and reptilian species observed on the site? Dutch Slough has been designed to meet a variety of conservation needs – as with fish, the presence of target species is an essential validation of the approach.

More years of data from Emerson, and following the inundation of Gilbert, should allow vegetative and geomorphic data from Emerson to guide more precise designs of future restoration sites.

Medium term adaptive management

Some aspects of restoration response are inherently slower and/or more dependent on year to year fluctuations in weather conditions and offsite fish abundance. However, some important management issues can be expected to be addressed at Dutch Slough while much of California EcoRestore is still underway.

1. Are constructed channel networks stable through time? Do they promote headcutting of smaller channels into more dendritic networks? Do they fill in excessively with

sediment or vegetation? Are constructed elevations stable through time or is there differential sediment capture causing the higher elevations to erode and deposit into the lower marsh? A lot of engineering effort was invested in designing Dutch Slough before opening it to tidal action. Was that effort successful? Was it worthwhile?

2. Does the greater public access to Emerson affect its wildlife value? Balancing human and wildlife uses of the restoration site was crucial to developing the strong local support it now enjoys. Were the negotiated results still supportive of the restoration goals?
3. A crucial restoration question is the degree to which restoration benefits occur of-site. This is captured in Performance Hypothesis 4 -- Dutch Slough will export, on every ebbing tide, substantial biomass of zooplankton of suitable size and food value for pelagic fish, including Delta Smelt. Flow in and out of the tidal wetlands will be primarily driven by tidal flows which are similar in all years. (Although some influence of flows from Marsh Creek may affect some food transport off-site). With sondes in place at the outflow points of Emerson, and with measurements of food produced within Emerson, the flux of material off-site can be estimated and used to address this question. A better answer to this question than we currently have would greatly benefit the design and location of future restoration projects.

Long-term adaptive management

From the experience of wetland restoration in South San Francisco Bay (Hobbs et al. 2012) and the Chesapeake (Abel et al. 2004), restored tidal wetlands can rapidly develop suitable vegetation and be occupied by most vertebrate species suited to the habitat. However, data to evaluate most of the performance and adaptive management hypotheses are heavily affected by the unpredictable factors of California weather and by the year-to-year fluctuations in the abundance of young salmon and splittail in the watershed. Statistical assessments of these hypotheses probably require at least five years of data. Thus, at five year intervals data will be analyzed for assessment of the project hypotheses. Recommendations will be made to modify hypotheses or monitoring methods based on results.

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Appendices

1. Attendees of the January 2015 meeting to assess species of interest and hypotheses
2. Chinook Salmon
3. Splittail
4. Delta Smelt
5. Tidal Wetland Overview
6. Foodweb
7. Aquatic Vegetation
8. Clam Effects
9. Contaminant Effects
10. Wetland Evolution
11. Transport

Appendix 1. Attendees of Jan 28, 2015 meeting of Dutch Slough Working Group

Darcy Austin	Delta Stewardship Council
Dave Contreras*	Dept Fish & Wildlife
Judy Drexler	USGeological Survey
Patty Finfrock*	Dept Water Resources
Rosemary Hartman*	Dept Fish & Wildlife
Lauren Hastings	Delta Stewardship Council
Mark Hester	University of Louisiana
Josh Israel	US Bureau of Reclamation
Alice Low*	Dept Fish & Wildlife
Ron Melcer	Dept Water Resources
Michelle Orr*	ESA-Associates
Anitra Pawley	Dept Water Resources
Stacy Sherman*	Dept Fish & Wildlife
Rebecca Sloan	ICF International
Hildie Spautz*	Dept Water Resources
Ramona Swenson	ESA-Associates
Dave Zezulak	Dept Fish & Wildlife

*Reviewers of revised plan

Also involved as reviewers:

Karen Kayfetz	Delta Science Program
Nadav Nur	Point Blue
Stuart Siegel	Stuart Siegel and Associates

Interagency Ecological Program
Tidal Wetlands Restoration Monitoring Project Work Team

Chinook Salmon Tidal Wetland Conceptual Model

Pascale Goertler[1], Kris Jones[1], Joseph Kirsch[2], Louise Conrad[1], Ted Sommer[1]
[1] CA Department of Water Resources
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Based on the Interagency Ecological Program's Management, Analysis and Synthesis Team's "An updated conceptual model for Delta Smelt: our evolving understanding of an estuarine fish":
http://www.water.ca.gov/iep/docs/Delta_Smelt_MAST_Synthesis_Report_January%202015.pdf

Estuarine wetlands are important nursery habitat for juvenile Chinook Salmon. Marshes and riparian wetlands are characterized by high insect production, refuge from predation, shade, and are particularly important for anadromous species, which require transitional habitat during their osmoregulatory shift into the marine environment. In addition, estuarine wetlands may provide a critical contribution to salmon habitat complexity and life history diversity. Therefore, tidal wetlands are a key watershed feature for targeted salmon habitat management and restoration. This conceptual model was designed to be used by several tidal restoration programs in the Bay-Delta Estuary to develop a set of hypotheses for tidal wetland restoration projects and formulate monitoring actions based on each project's goals. This conceptual model represents our best scientific understanding of Central Valley juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) and their use of tidal wetlands for rearing and migration. The Chinook Salmon tidal wetland model will inform monitoring plans for wetland restoration in the Sacramento-San Joaquin River Delta mandated by the Fish Restoration Program Agreement (FRPA) and other tidal restoration projects pursuing native fish habitat. There are many possible attributes to consider, however, we have first focused on the attributes and drivers that are considered to be the highest priority for monitoring juvenile salmon tidal wetland restoration projects in the Delta. Because salmon occur across a landscape, which can have cumulative contributions to measurable salmon responses, it can be difficult to relate population level management goals to site specific assessments. Therefore we have designed this conceptual model per recommendations from Simenstad and Cordell 2000, which advocate for a monitoring plan based on a habitat's capacity, opportunity and realized function.

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Introduction

This conceptual model represents our best scientific understanding of Central Valley juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) and their use of tidal wetlands for rearing and migration. The Chinook Salmon tidal wetland model will inform monitoring plans for wetland restoration in the Sacramento-San Joaquin River Delta mandated by the Fish Restoration Program Agreement (FRPA) and other restoration programs which provide tidal wetland habitat for fish. This conceptual model was designed to be used by the Fish Restoration Program and others to develop a set of hypotheses for tidal wetland restoration projects and formulate monitoring actions based on each project's goals. There are many possible attributes to consider, however, we have first focused on the attributes and drivers that are considered to be the highest priority for monitoring juvenile salmon tidal wetland restoration projects in the Delta.

Historically, the Central Valley supported approximately 1-2 million Chinook Salmon spawners annually (Yoshiyama 1998). Lindley *et al.* (2004) estimated that there were eighteen independent populations of Chinook Salmon within a single seasonal adult run. However, the overall abundance of Central Valley Chinook Salmon has decreased to less than 75% of estimates from the 1950s (Yoshiyama 2000), and in the past ten years adult escapement estimates have not reached 50% of the more conservative historical estimates (Figure 1). The major causes recognized for these declines are overfishing, destruction of habitat by hydraulic mining and construction of dams and water-diversion projects, dredge mining, railroad construction and logging (Yoshiyama 1998). Only a small fraction of the wetland rearing habitat is still available: 5% of historic wetlands remain in the Sacramento-San Joaquin River Delta (the Delta). Since the middle of the 19th century and the onset of the California Gold Rush, the Delta has been diked, dredged, diverted and contaminated with waste materials and non-native plants and animals to support agriculture and human population growth. In addition to substantial wetland loss, Chinook Salmon are extirpated from approximately 75% of the upstream spawning and rearing habitats (Van Cleve 1945). Approximately half of the major tributaries have lost at least one seasonal run of salmon, while others have been dammed without providing for fish passage, creating a permanent obstruction to salmon migration (Yohiyama 2000; Williams 2012). Fisheries managers concerned with these dramatic declines in habitat and salmon production have focused on tidal wetland restoration projects as a method for rehabilitating native marsh habitat and Chinook Salmon populations.

Estuarine wetlands are important nursery habitat for juvenile Chinook Salmon (Healey 1982; Simenstad et al. 1982). The percentage of estuary in pristine condition has a significant positive relationship with juvenile Chinook Salmon survival (Magnusson and Hilborn 2003), suggesting that improved fitness during estuary rearing and migration can have population-level effects on survival. Marshes and riparian wetlands are characterized by high insect production, refuge from predation (Beck et al. 2001), shade, and are particularly important for anadromous species, which require transitional habitat during their osmoregulatory shift into the marine environment (Simenstad et al. 1982; Thorpe 1994). In addition, estuarine wetlands may provide a critical contribution to salmon habitat complexity; connecting the watershed's streams, rivers and floodplains to the sea through a mosaic of freshwater tidal wetlands, brackish marshes and low salt marsh. This has been shown through habitat restoration in the Salmon River estuary, where the removal of dykes and the restoration of historic tidal marsh resulted in the expression of a previously depressed juvenile life history type of Chinook Salmon (Bottom et al. 2005). This study showed that the reestablishment of previously unavailable estuarine habitat can expand life history variation. Therefore, estuarine wetlands contribute to life history variation, survival (Magnusson and Hilborn 2003; Rechisky et al. 2012) and growth (Healey 1982; Neilson et al. 1985; Sommer et al. 2001; Maier and Simenstad 2009) among out-migrating juveniles, and may improve the productivity and resilience of the adult population (Reimers 1971).

Many salmon habitat restoration projects have concentrated on opportunistic removal of barriers to increase connectivity to historically available habitats (Tanner et al. 2002; Bottom et al. 2005; Roegner et al. 2010). These projects reduce habitat fragmentation and bottlenecks to recovery, but many are limited primarily by scale and the ability to match ecological priorities with practical goals and land allocation. Although the realities of implementation are an inevitable constraint, restoration projects must be designed at the appropriate scale and include sufficient planning to address their management goals (Simenstad et al. 2006). Therefore we have built a conceptual model to guide the title monitoring plan (citation), and clarify key assumptions and knowledge gaps. Our goals for this conceptual model and the subsequent monitoring plan is to provide a relevant and complete depiction of those tidal wetland features important to juvenile Central Valley Chinook Salmon use of and enhancement on Delta restoration sites. In addition, we hope to provide useful descriptions and applicable examples for each tidal wetland feature which we have included.

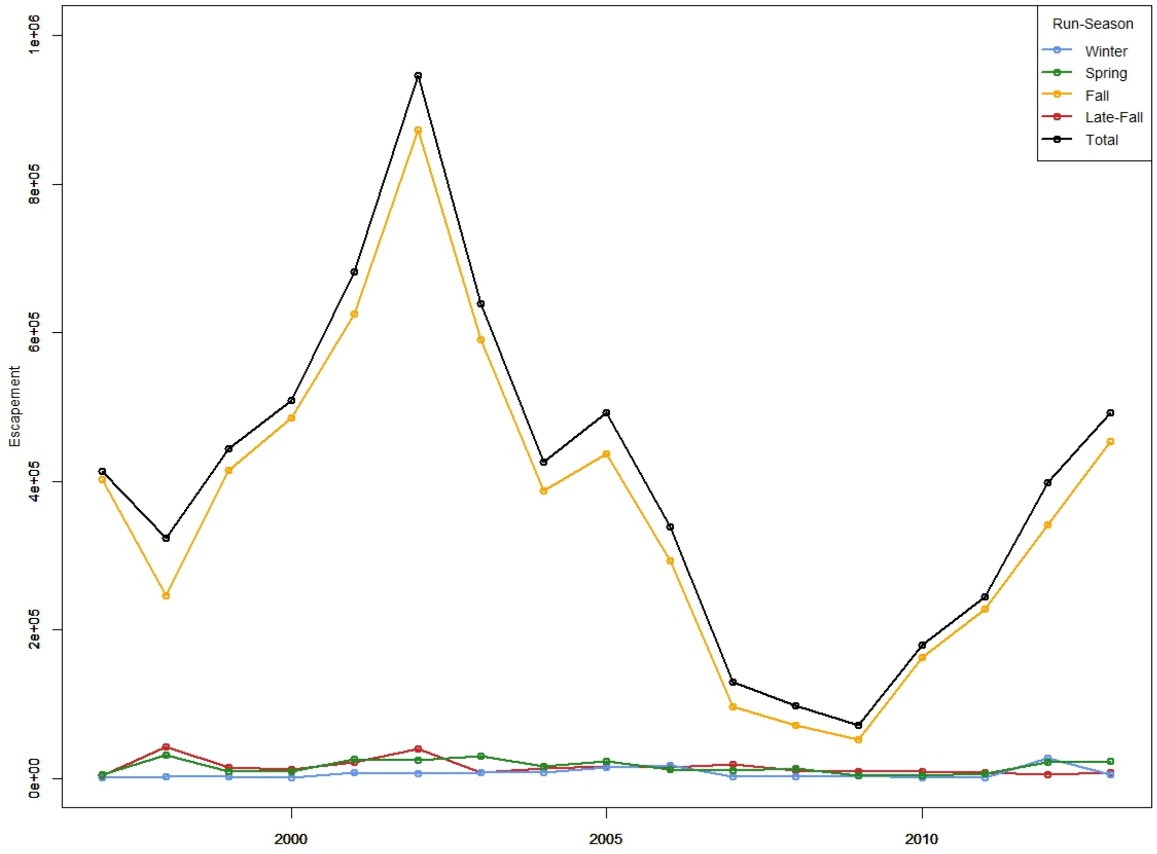


Figure 1: Adult escapement estimates for Central Valley Chinook Salmon, by return to river season (or run), for 1997-2013. These data have been synthesized from GrandTab 2014.04.22 California Central Valley Chinook Population Report (Azat 2014).

Central Valley Chinook Salmon

Pacific salmon are anadromous species and may complete their life cycles through a variety of alternate pathways. Pacific salmon externally fertilize their eggs in freshwater gravel nests, which hatch and develop into alevin, and rear as juveniles in the ocean, estuary, or stream. Eventually they migrate to the ocean as smolts and after a time at sea a series of physiological processes lead them to migrate back to the freshwater system from which they emerged, spawn and die (Quinn 2005). The size, age and habitat in which these life phases are completed are specific to each life history type and species of Pacific salmon.

Central Valley Chinook Salmon are commonly described by four runs, which signify the season in which adults return to the freshwater system from which they emerged, to spawn: winter, spring, fall and late fall (Yoshiyama et al. 1998). The Central Valley is the only system to have Chinook Salmon spawning year-round and maintain four distinct adult runs. In Pacific salmon (*Oncorhynchus* spp.), population diversity is an important mechanism for resilience. Several studies analyzing forty years of Alaskan sockeye (*O. nerka*) catches emphasize the significance of the biocomplexity of fish stocks in ensuring collective population productivity despite major environmental change (Hilborn 2003, Schindler 2010). In these studies resilience is dependent on the maintenance of all the diverse life history types and geographic locations that comprise the stock (Hilborn 2003). Despite their diversity in return to river timing, Central Valley Chinook Salmon are at critically low population levels; the winter run are listed as endangered and spring run are listed as threatened under the Endangered Species Act (ESA) (Good 2005). Differences in listing among Central Valley Chinook Salmon populations has resulted in permitting policies to delineate juveniles of each run-type by a length at date policy. The “length at date” categorization, although the best available method at the time, assumes constant growth and no overlap among the size distributions by run-type. Recent advances in genetic technology have shown that these assumptions are not accurate and nearly half of the genetically identified juveniles were measured outside their expected length ranges (Harvey et al. 2014). It remains difficult to distinguish between runs of Central Valley Chinook Salmon as juveniles and calls into question how to use run-type specific data as a measure of diversity or resilience. Studies of adults show some biocomplexity remains within Central Valley Chinook Salmon populations, but has been severely eroded; limiting its ability provide stabilization and leaving the population vulnerable (Carlson and Satterhwaite 2011).

Throughout the western United States major declines in Pacific salmon have given rise to artificial spawning and rearing of juveniles in hatcheries to improve growth and survival in the first year of life. Millions of hatchery-reared Central Valley Chinook are released each year (Figure 2) to support the California and Oregon salmon fisheries. To counter act harvest declines and ameliorate rearing habitat loss, hatcheries target fast growth and large sized juveniles. Although growth has been shown to be an important indicator for juvenile salmon survival (Duffy and Beauchamp 2011; Scheuerell and Williams 2005; Zabel et al. 2006), there is no one optimum phenotype. Therefore there is ongoing debate regarding the value of hatchery produced juvenile salmon; in addition to their negative effects on wild fish (Levin et al. 2001; Araki et al. 2007; Araki and Schmid 2010).

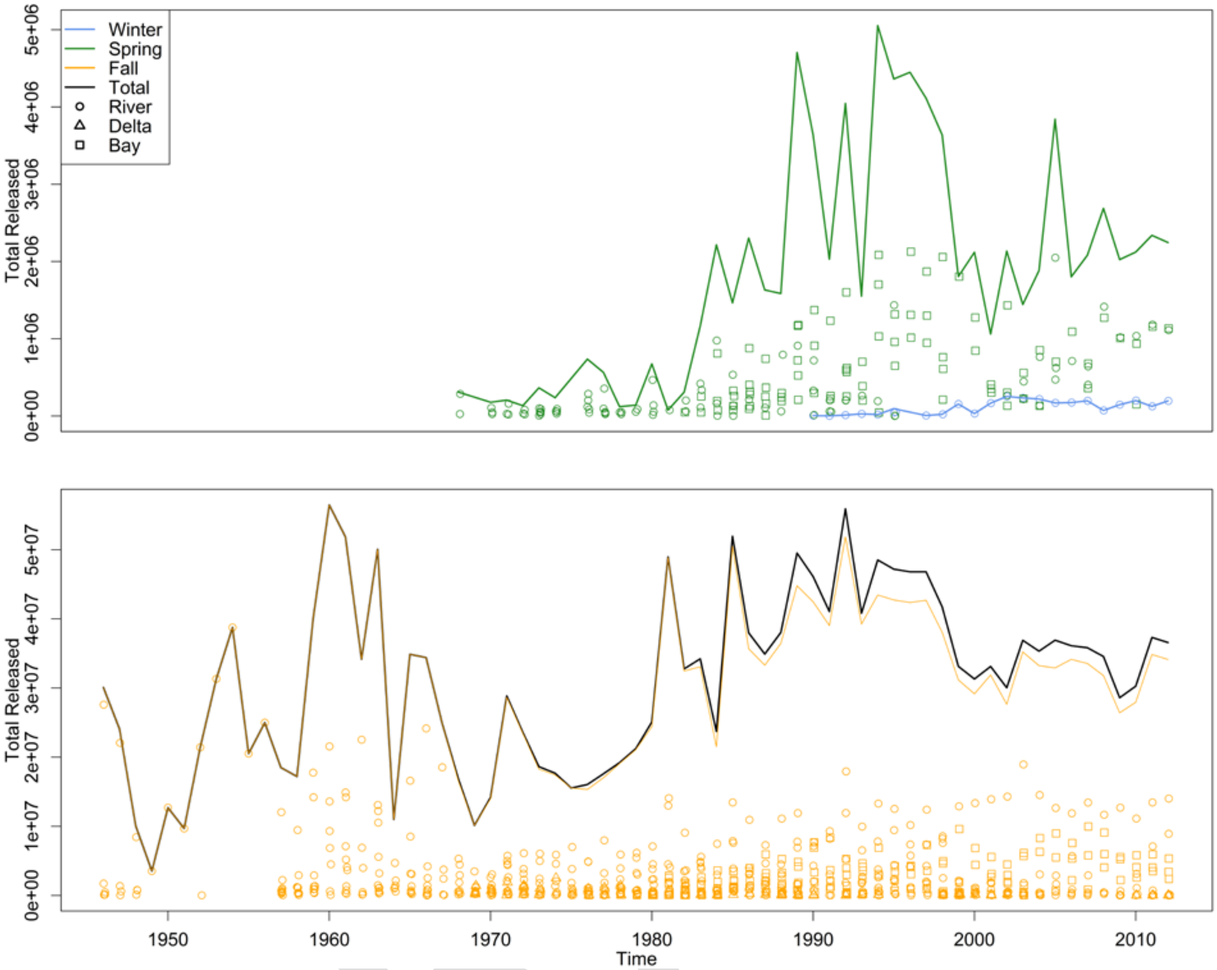


Figure 2: Total estimated hatchery releases by both federal and state operated hatchery facilities over time (Huber and Carlson 2015, Anna Kastner unpublished, Kevin Offill unpublished). Colors represent adult return to river timing population. The lines show yearly totals and the points depict where the juvenile hatchery salmon were released (in river, Delta or in the Bay) as well as across how many months within a year releases occurred. Note the difference in y-axes between the top and bottom figure.

Increasingly, fisheries managers are focusing on providing for the expression of life history diversity in salmon to incorporate resilience and long-term sustainability of commercially harvested species. In the Central Valley there is evidence for diverse juvenile migratory phenotypes contributing to the adult population (Miller et al. 2010). Juvenile life history types include different temporal and spatial patterns of habitat use, and are generally defined by their size relative to the age of migration or location of their rearing habitat in their natal watershed. Juvenile Chinook life history diversity in the Central Valley is described by Williams (2006) and (2012) and depicted in Figure 3. Central Valley Chinook Salmon can: (1) migrate directly to the brackish estuary, bays or ocean after emergence, (2) migrate to the ocean after less than one year, in which they (a) migrate directly to the Delta after emergence, (b) rear in low-gradient streams, (c) rear on or near the spawning gravel-beds, or (d) migrate to low-gradient streams after emergence and rear in the Delta after weeks to months in river, and (3) rear on or near the spawning gravel-beds for an entire year before migrating to sea (Figure 3; Williams 2012). Efforts to support naturally derived life history variation could buffer the seasonal risks of migrating through the Bay-Delta system and annual variability in ocean conditions. This conceptual model aims to incorporate tidal wetland habitat features and the seasonal variability used by all juvenile life history types of Central

Valley Chinook Salmon. This conceptual model supports a monitoring framework which integrates structural complexity, connectivity and a broad range of seasonal tidal wetland habitat use, for both rearing and migration by juvenile salmon.

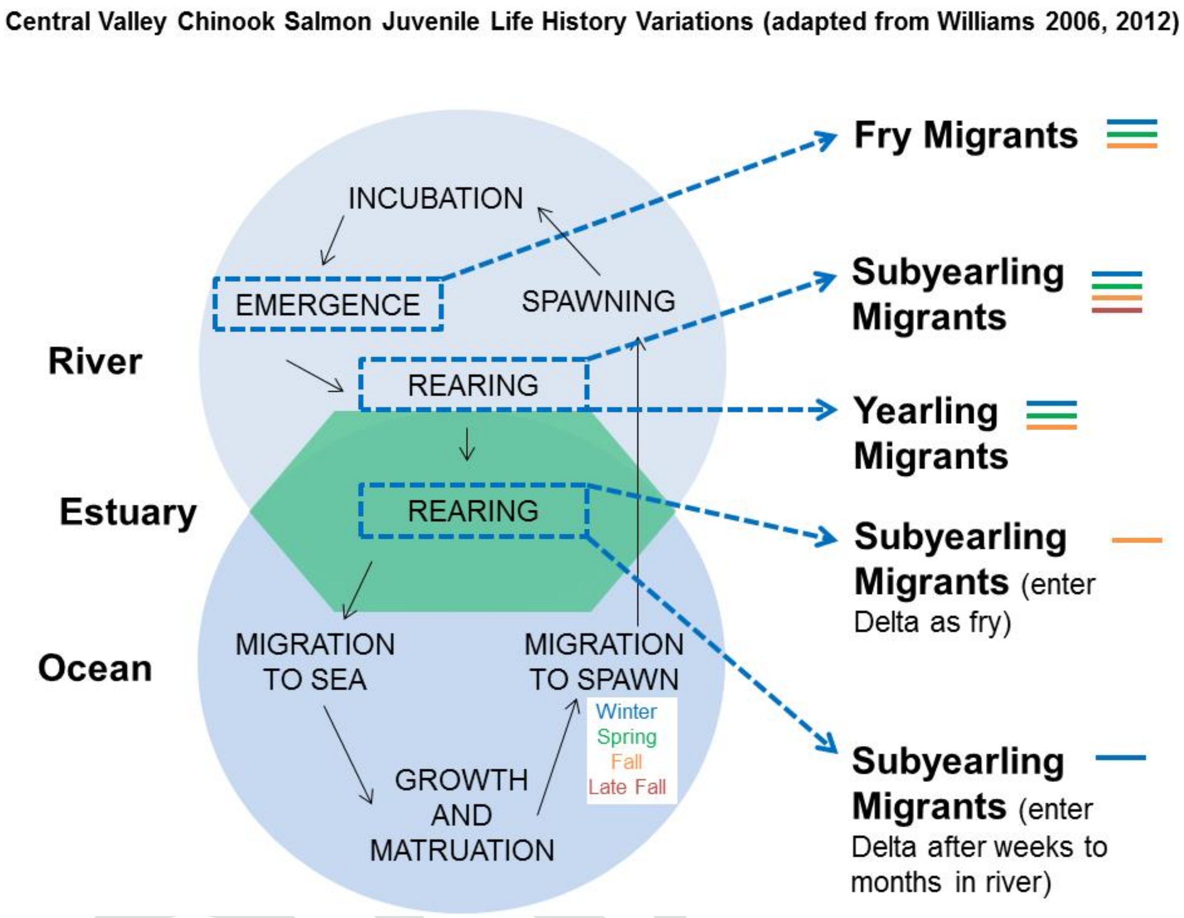

Central Valley Chinook Salmon Juvenile Life History Variations (adapted from Williams 2006, 2012)

Figure 3: An illustration of the juvenile Central Valley Chinook Salmon life history variations described by Williams (2006) and Williams (2012). The entire salmon life cycle is represented however differences among the timing of juvenile phases (fluvial rearing, tidal rearing and migration to sea) are emphasized. Dark blue dashed lines represent the life history phase which preceded migration to sea and the life history variant names denote the age/size at which migration to sea occurs. Colored bars correspond to the adult return to river timing. It is important to note that life history variants are meant to represent general patterns not discrete types and fluvial rearing includes both rearing in low gradient streams and rearing on or near the spawning gravel-beds, which are treated as separate in Williams 2012. Additionally, this form of visualizing juvenile life history variation was crafted follow those in Bottom *et al.* (2009).

Model Structure

The model structure and choice of tier names is directly taken from the Interagency Ecological Program, Management, Analysis and Synthesis Team's recently updated conceptual model (CM) for Delta Smelt (Baxter et al. 2015). Similar to the Delta Smelt CM, this model includes stationary ecosystem components and dynamic environmental drivers, habitat attributes, and target species responses. The Delta Smelt and salmon CMs have tiers, which build upon each other. Biotic and abiotic habitat elements (tier 3) drive salmon responses (tier 4), while natural and anthropogenic ecosystem drivers associated with the estuarine environment (tier 2) influence habitat (tier 3), and geographically fixed attributes of the estuarine ecosystem within the terrestrial, aquatic and marine landscape (tier 1) impact environment drivers (tier 2). The arrows between attributes and drivers are mechanistic linkages to juvenile Chinook Salmon responses. The linkages are directional, but the colors and sizes of arrows do not infer any importance or the significance of relationships. It is particularly important to note that arrows link specific terms from tier to tier as well as groups of terms taken together. For example, every term in the habitat attributes tier is considered linked to realized function in the juvenile salmon responses tier, because varying environmental regimes, locations, and/or management actions may enhance or mute the effect of a restorations site's habitat attributes on juvenile Chinook Salmon. Terms in blue underlined text are links to companion models which explain those terms in greater detail.

Unlike the Delta Smelt CM we did not include a "life stage season" (tier 5). We chose to include all juvenile salmon life stages between the upper estuary and ocean outmigration because wild juvenile Chinook life stage transitions are a continuum of sizes, growth rates, and timing of habitat occupancy and movement (Reimers 1973; Bottom et al. 2005; Anderson 2006) rather than discrete life history types. We expect juvenile Chinook to be present in the estuary nearly year round (Williams 2012); therefore an annual not seasonal approach is being advocated with this conceptual model. We have intentionally and explicitly included potential mechanisms that support resilience for Central Valley Chinook Salmon, such as landscape scale and site specific habitat complexity and life history diversity.

Additionally, tiers three and four reference principles that specifically address the complications of assessing the functions of recently restored estuarine habitat for transient species such as salmon, which temporarily occupy any one location (Simenstad and Cordell 2000). Because salmon occur across a landscape, which can have cumulative contributions to measurable salmon responses, it can be difficult to relate population level management goals to site specific assessments. Therefore we advocate for a monitoring plan based on a habitat's capacity, opportunity and realized function (Simenstad and Cordell 2000). 'Capacity' reflects habitat attributes that support juvenile salmon foraging, growth and decreased mortality. For example, capacity monitoring may include the availability and quantity of salmon prey, the physicochemical conditions that maintain prey communities and temperatures that promote high metabolic efficiency (Simenstad and Cordell 2000). 'Opportunity' addresses the ability of a juvenile salmon to access high capacity habitat. Flooding (tidal or fluvial), geomorphic features important to connectivity, proximity to disturbance, refugia and the strength of cues may all influence a salmon's opportunity to access a restored marsh (Simenstad and Cordell 2000). Finally, 'Realized Function' measures direct physiological or behavior responses to the restored marsh that supports fitness and survival (Simenstad and Cordell 2000). Simenstad and Cordell (2000) describe realized function in the context of survival, residence time, foraging success and growth. In the Chinook Salmon tidal wetland CM we have also included life history diversity and migration timing to incorporate potential mechanisms that support resilience.

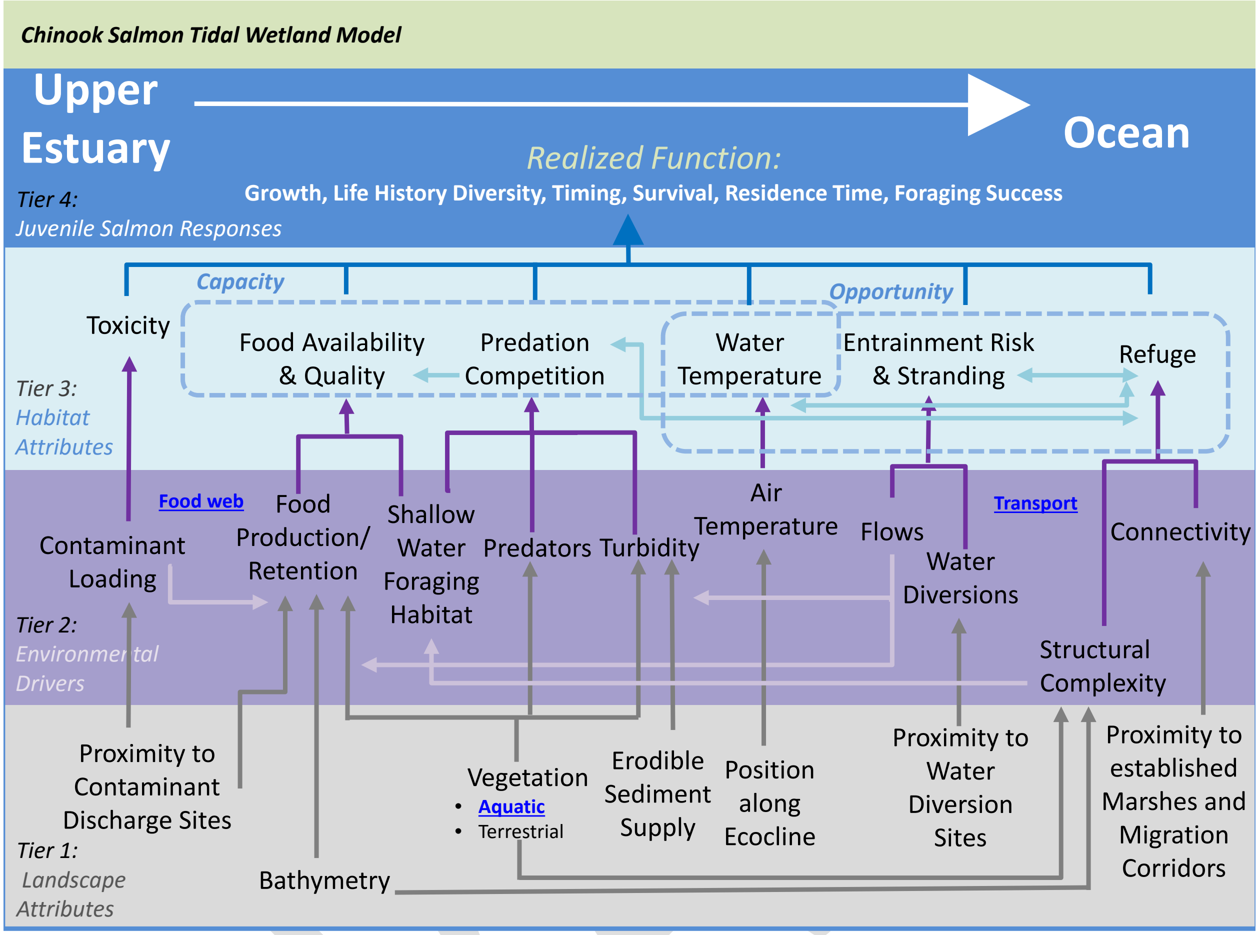


Figure 4: The Chinook Salmon Tidal Wetland Conceptual Model.

Below is a brief review of each of the terms listed within each of the tiers, and the linkages between terms within and among tiers.

1. Landscape Attributes

Proximity to Contaminant Discharge Sites

Link to Contaminants Model

Proximity to contaminant discharge sites includes the distance to agricultural, industrial, urban and mining sources. The location of a restored tidal wetland relative to contaminant discharge sites will greatly influence the potential for contaminant exposure for juvenile salmonids using the area.

Bathymetry

Bathymetry refers to water depth and affects structural complexity and food production and retention. Water depth will influence residence time of water and phytoplankton, which will have bottom-up effects on food production and retention for juvenile salmon. Additionally, the construction or natural formation of channels, ponds and floodplains will affect juvenile salmon's capability to access the habitat's capacity. The penetration of tidal channels and water depth will determine the duration and frequency of access to restored sites.

Vegetation

Vegetation includes, floating aquatic vegetation, submerged aquatic vegetation, emergent aquatic vegetation and riparian vegetation. Vegetation effects food production and retention, predators, turbidity and structural complexity. Juvenile salmon feed on both aquatic and terrestrial invertebrates and salmon habitat is structured by the upland and terrestrial landscape and the composition of a restored wetland's aquatic vegetation. Depending on the density of aquatic and emergent vegetation, and the size of a fish, vegetation could limit access by juvenile salmon and/or provide refuge from larger piscivorous aquatic or avian predators. However, vegetation may also provide holding areas for some non-native fish predators, such as Largemouth Bass (Brown 2003). Riparian vegetation provides allochthonous inputs, shade and refuge from avian predators and during flooding events could slow water and expand available habitat for juvenile salmon. Both aquatic and terrestrial vegetation can also increase the residency time of water, and reduce turbidity.

Link to Aquatic Vegetation Model

Erodible Sediment Supply

The erodible sediments supply controls the input of suspended sediments into the Delta and is a key feature contributing to turbidity. In combination with bathymetry and hydrology, the erodible sediment supply also effects wetland evolution.

Link to Wetland Evolution Model

Position along Ecocline

A restoration site's position along the ecocline refers to its location within the Bay-Delta landscape. For example, this would include the site's distance from the ocean, various freshwater inputs, tidal forcing and surrounding terrestrial setting.

Proximity to Water Diversion Sites

The proximity to large scale water diversions (e.g. Delta Cross Channel, Central Valley Project and State Water Project pumps, etc.) and the cumulative effect of small water control structures (e.g. agricultural siphons and pumps, weirs, etc.) can affect juvenile Chinook Salmon within tidal wetland restoration sites by altering hydrodynamics (Kimmerer 2002; Kimmerer 2004), productivity (Jassby et al. 2002) and/or causing entrainment or stranding at local and regional scales (Kimmerer 2008; Perry et al. 2010; Nagrodski et al. 2012; Perry et al. 2013).

Proximity to established Marshes and Migration Corridors

This term is meant to represent the complexity and connectivity of restoration site within the surrounding landscape. Particularly for salmonids, which are migratory species, the proximity of a restoration sites to established marshes and migration corridors may affect juvenile salmon access to the wetland and the strength of cues that might attract them to the restored wetland (i.e. opportunity). Additionally, their available paths to the ocean by way of migration corridors will affect survival and the suite of life history types and migration timing available.

Link to Transport Model

2. Environmental Drivers

Contaminant loading

Contaminants are considered one of the key factors contributing to the decline of numerous Delta species (Werner and Anderson 2008). Previous work has demonstrated that exposure to contaminants can negatively influence the fitness of fish directly through sublethal physiological, behavioral, or immunological effects, or indirectly through the food web. There is a general awareness that exposure to contaminants can impair the health of Chinook Salmon. A number of studies have documented adverse effects, but little is known regarding the thresholds at which most contaminants would be toxic to or otherwise adversely affect Chinook Salmon (or their prey).

Sources of contaminants within the San Francisco Estuary and its tributaries include wastewater treatment plants (Kolpin et al. 2002), urban and agricultural runoff (Kuivila and Hladik 2008;Weston and Lydy 2010), and metals from historic mining activities (Buck et al. 2007; Conway et al. 2007). While there are approximately nine wastewater treatment plants in the legal Delta, the Sacramento Regional Wastewater Treatment Plant (SRWTP) is one of the largest known sources of ammonia and ammonium entering the lower Sacramento River (Brooks et al. 2012). Urban runoff can also contribute contaminants, such as mercury, copper, silver, and other metals (Flegal et al. 2005). Agricultural applications of pesticides are also well documented in the San Francisco Estuary; however, the transport mechanisms for these chemicals into local waterways are poorly understood, and in most cases they are not monitored (Kuivila and Hladik 2008). The applications of urban pesticides are even less documented, making it difficult to quantify the loadings for these chemicals entering the Delta (Johnson et al. 2010). Finally, while mining activities in the mid-1800s contributed substantial amounts of metals to the system, (e.g. copper, mercury etc.; Domagalski et al. 2004), mines in the Sacramento River watershed have met remediation criteria since 2003 (USEPA 2008). For the purpose of this CM, contaminants include; mercury, selenium, pyrethroids, copper, nickel, organometallic compounds, PCBs, PAHs, ammonium, and endocrine disrupting chemicals (Werner and Anderson 2008).

Food production and retention

Link to Food Web Model

Our general expectation is that estuaries provide energetically rich foraging grounds and abundant prey for juvenile Chinook Salmon. Chinook Salmon estuarine rearing can be extensive, and they have been suggested to be the most estuarine dependent of the salmon species (Healey 1982). Adequate food supply is an obvious necessity for the survival and fitness of juvenile salmon. However, in practice quantifying sufficient food supply requires more information that is usually available, such as prey availability, threshold populations for successful feeding, prey composition and prey aggregation (Healey 1982). Food production and retention refer to the physicochemical conditions that maintain prey communities within a restored marsh, which may be more easily measured (Simenstad and Coredell 2000). Commonly in estuaries, juvenile salmon diets are supported by detritus based food webs (Healey 1979, Healey 1982; Maier and Simenstad 2009, Naiman and Sibert 1979, Sibert 1978). Therefore mechanisms for retaining organic inputs in estuaries may be very important for food production (Healey 1982). Retention of organic carbon could be enhanced by limited exchange with the open ocean and low bed load transport (Odum et al. 1979), such as detritus trapping vegetation, and braided, meandering channels (Healey 1982).

Productivity in the Delta is and has declined over recent decades due to invasive filter-feeding bivalves, decline in sediment influx, and changes in river inflow levels (Jassby et al. 2002). Indeed, limited food supply is thought to be a causative factor in recent pelagic fish declines (Sommer et al., 2007). While the Delta food web is generally thought to be pelagic-based, little attention has be devoted to understanding detrital pathways, which may be particularly important in wetland habitats (How and Simenstad 2011; Howe, in prep). As food web enhancement is one of the primary goals of wetland restoration, it will be critical to thoroughly monitor productivity of lower trophic organisms at restoration sites in order to evaluate project success and habitat capacity for juvenile salmonids.

Shallow water foraging habitat

Many studies have documented the use of shallow water habitat by juvenile salmon (Levy and Northcote 1982, Healey 1991, Miller and Simenstad 1997, Miller and Sadro 2003, Bottom 2005a), and shallow water habitat has historically supported salmon food webs (Bottom et al. 2012). Therefore, the extent of shallow water foraging habitat is expected to influence food availability and competition in this CM. In the salmon CM, shallow water foraging habitat is an important habitat type within the structural complexity of the marsh landscape and is shaped by bathymetry and vegetation. We expect shallow water habitat to be used by smaller individuals and are therefore defining shallow water foraging habitat as the optimal depth for subyearling Chinook: 0.1-2.0 m deep (Bottom et al. 2012). Bottom *et al*. (2012) found that small size classes favored shallow water habitats, and that the smallest individuals reared in shallow peripheral channels of the wetland regardless of the vegetation type (emergent, scrub-shrub, forested and mixed). These smaller fish also had the largest residency times (2-3 months), and gradually moved offshore and toward the estuary mouth as they fed and grew (Bottom et al. 2012). Shallow water marsh habitats may support Delta rearing fry migrants; in the Columbia River Estuary 32-45% of juvenile salmon captured in shallow water habitats had entered the estuary soon after emergence (Bottom et al. 2012). For Central Valley Chinook Salmon, fry migrants are estimated to make up 20% of the adults whom successfully return to spawn, and subyearling migrants accounted for 48% (Miller et al. 2010). Therefore, supporting these smaller individuals could have population-level consequences for Central Valley Chinook productivity.

We also expect naturally produced subyearling salmon to dominate shallow wetland channels (Bottom et al. 2012) and most directly benefit from restoration of wetland habitats. Subyearling migrants have shown extensive sallow water habitat use; PIT tagged results showed some juveniles returning repeatedly to the same site on the tidal cycle and others visited the same channel intermittently for weeks (Bottom et al. 2012). Shallow water foraging habitat is the dominant habitat type we expect rearing juvenile Chinook Salmon to use within restored tidal wetlands. Currently, spring flooding creates the majority of the shallow water habitat available to juvenile Chinook Salmon in the Delta (Bennett and Moyle 1996; Sommer et al. 2005).

Predators

Generally, estuaries are thought to reduce predation risk for juvenile fish, and emergent and submerged vegetation can provide cover for small fish from larger predators (Kilgore et al. 1989). However, *Egeria densa* may invade subtidal areas and harbor predators, thereby increase the probability of predation on native fishes by invasive fish, such as Largemouth Bass (Brown 2003). Therefore vegetation can have a major influence on the predator community and its spatial overlap with juvenile salmon. Additionally, artificial structures can create habitat bottlenecks and predation hotspots in human modified landscapes (Grossman et al. 2013; Sabal 2014). An increase of more natural tidal habitat and vegetation may provide protection from predatory fishes (Lott 2004). The major predators (and is some cases competitors) on juvenile salmon in the Delta (and those of primary concern in the conceptual model) are Striped Bass (*Morone saxatilis*), Largemouth Bass (*Micropterus salmoides*), Smallmouth Bass (*Micropterus dolomieu*), White Catfish (*Ictalurus catus*), Channel Catfish (*Ictalurus punctatus*), Sacramento Pikeminnow (*Ptychocheilus grandis*), and juvenile Chinook Salmon (Grossman et al. 2013).

Turbidity

Turbidity is the loss of transparency of water, caused by suspended particles scattering light. The amount of suspended particles in the water depends on the erodible sediment supply and hydrodynamic energy, which is affected by runoff, tidal forcing, bathymetry, and wind waves. Turbidity can be reduced by vegetation and water diversions, which can both reduce hydrodynamic energy. In the context of this conceptual model, as in the Delta Smelt conceptual model, turbidity is a habitat attribute, which references the visibility of forage fish by predators and is therefore a modifier of predation risk (Baxter et al., 2015). Turbidity can limit predation; a study on predation by piscivorous fish on migrating juvenile salmon showed a reduction in both predator encounter and consumption (Gregory and Levings

1998).Turbidity has also been found to reduce the impact of predation risk; low visibility reduces the distance at which predator-prey interactions occur and reduces the predator avoidance behaviors which can negatively affect growth rates of prey (Abrahams and Kattenfeld 1997). The Bay-Delta is considered most turbid in the Suisun region, Liberty Island and Cache Slough complex, moderately turbid in the lower channels of the Sacramento and San Joaquin River and least turbid in the South Delta (Baxter et al., 2015).

Air temperature

Air temperature is driven by its regional climate; the warming or cooling of the surrounding terrestrial landscape, proximity to the ocean and Pacific-related teleconnections, such as PDO (Pacific Decadal Oscillation) and PNA (Pacific North American Pattern) (Prowse et al. 2007). Water temperature can be predicted by air temperature with a linear regression for temperatures above 0°C, with the exception of the highest and lowest temperatures (Mohseni et al. 1998). This non-linearity at the temperature extremes can be caused by high evaporative cooling during the warmest periods and heat storage effects (Mohseni et al. 1998). Mohseni *et al*. (1998) suggest using a logistic equation to predict weekly maximum and minimum water temperatures as a reliable way to estimate weekly stream temperatures from air temperature. In the Delta, water temperature is commonly estimated by air temperature and the prior day's water temperature (Wagner et al. 2012).

Flows

The flow regime within the San Francisco Estuary is highly influenced by water year type, tidal forces, and water operations management (Kimmerer 2004). Many studies have suggested that wetter water years or periods of flooding, especially floodplain inundation, create better habitat for rearing and migratory fish. Example benefits include: higher biotic diversity (Junk et al. 1989), increased production (Bayley 1991; Gladden and Smock 1990; Halyk and Balon 1983), increased available habitat (Junk et al. 1989; Sommer et al. 2001; 2005), inputs of terrestrial material into the aquatic food web (Sommer et al. 2001; Winemiller and Jepsen 1998) and increased connectivity during migration (Bennett and Moyle 1996). However, high water events can also be very disruptive for many small fishes. Therefore, refuge habitat, such as vegetated marshes are necessary to buffer against flushing (Pearson et al. 2011; Simenstad et al. 2000). Further, considerable fluctuations in flow and increased water diversion can elevate the risk of juvenile Chinook Salmon stranding in peripheral habitats (Nagrodski et al. 2012 and references therein) and entrainment from water diversions (Kimmerer 2008). For example, Sommer et al. (2005) found that weir structures tended to increase stranding rates following flow events. Therefore, flows can have a very important and highly variable impact on juvenile Chinook Salmon habitat availability, productivity and risk.

Water diversions

The San Francisco Estuary and its watershed have been substantially altered by water control structures (e.g. levees, temporary rock dams, weirs, dams, etc.) to minimize flooding and facilitate out-of-basin water exports (Nichols et al. 1986; Kimmerer 2004). The Delta is managed and engineered to supply freshwater for export via the Central Valley Project and State Water Project pumping plants (Kimmerer 2004). To maintain water quality standards (e.g. low salinity) for out-of-basin export, water is often conveyed from the Sacramento River into the central Delta via the Delta Cross Channel (Kimmerer 2004). Large scale water diversions (e.g. export and conveyance) can affect juvenile Chinook Salmon and their habitat within or near tidal wetland restoration sites by altering water quality (Kimmerer 2004; Monsen et al. 2007), productivity (Jassby et al. 2002; Kimmerer 2002), migration routes and timing (Perry et al. 2010; Perry et al. 2013), and causing entrainment (Brown et al. 2008; Kimmerer 2008; Zueg and Cavallo 2014). For example, numerous studies have demonstrated that large scale water operations can greatly alter the flow regime (e.g. Delta outflow) and water quality (e.g. salinity) at both regional and local scales (Jassby et al. 2002; Kimmerer 2002; Kimmerer 2004). In addition, there are over 2,000 small agricultural siphons and pump diversions within the Delta used for local irrigation and over 300 small floodgate diversions within Suisun marsh for wetland management (Herren and Kawasaki 2001). Although the cumulative effect of these diversions on juvenile Chinook Salmon is not well understood, small diversions can cause entrainment and influence the hydrogeomorphology of a site by influencing depth, shoreline structure, and water velocity (Kimmerer 2004; Moyle and Israel 2005; Mussen et al. 2013).

Structural complexity

Estuaries are complexes of interconnected wetlands along a salinity gradient, and many interactive landscape elements can influence habitat use by juvenile salmon. Structural complexity is meant to include both the structure of the marsh and the principal processes that maintain that structure. In our conceptual model bathymetry and vegetation are underlying landscape attributes that sustain structural complexity, which affects both the capacity of the marsh (shallow water foraging habitat) and juvenile salmon access (refuge). Bathymetry dictates structural complexity with channel morphology (order, size, density and sinuousity) and maintains it through hydraulic energy, such as the tidal prism and freshwater inflow (Simenstad et al. 2000). While the structure and heterogeneity of riparian and aquatic vegetation both responds to bathymetry and can encourage marsh evolution, such as sedimentation. Vegetation also creates the marsh-edge, which is the ecotone most used by juvenile salmon for foraging and refuge from predation (Simenstad et al. 2000).

Structural complexity can support habitat capacity by providing a variety of marsh microhabitats, such as vegetated and unvegetated marsh plains, salt ponds and pans, and tidal channels of different order, size and depth (Simenstad et al. 2000). Each of these microhabitats may affect the production or availability of specific groups of prey resources for juvenile salmon (Simenstad et al. 2000). In tidal wetlands detritivores are the primary consumer that serve as important fish prey (Healey 1979, Healey 1982; Maier and Simenstad 2009, Naiman and Sibert 1979, Sibert 1978). Therefore, the production of detritivore prey species depends upon processes of detritus accumulation and decomposition (Healey 1982; Simenstad et al. 2000), such as limited exchange with the open ocean and low bed load transport (Odum et al. 1979), detritus trapping vegetation, and braided, meandering channels (Healey 1982). Marsh topography can be important for production and retention of juvenile salmon prey as well as concentrate terrestrial drift invertebrates (Hood 2002; Simenstad et al. 2000). In one study, residence time of drift invertebrates was directly related to the distance from the up-channel mouth of sloughs in the tidal floodplain of the Chehalis River (Simenstad et al. 2000).

In addition to enhanced capacity, structural complexity can also provide refuge for juvenile salmon. Extensive channel and slough systems can provide refuge during high current flows from ocean storms or rain events. Hydraulic complexity has been shown to retain proportionately more fish after disturbance events (lower disturbance intensity) and have higher fish diversity than simpler reaches (Pearson et al. 2011). For example, Coho, which rear primarily in riverine reaches, have been shown to overwinter in tidal-freshwater floodplain marshes during high river discharge periods (Simenstad et al. 2000). During storm flows, complex stream morphology and developed vegetation can maintain high transient storage fractions, important for retention (Kaufmann and Faustini 2012). Bathymetric heterogeneity has also been shown to increase residence time of juvenile Chinook Salmon in the Salmon River estuary by providing refuge throughout the tidal cycle (Hering et al. 2010). The structure provided by vegetation can also encourage increased residency; one study showed that a positive relationship between cover complexity and the residence time of Coho salmon (*Oncorhynchus kisutch*) (McMahon and Hartman 1989). Marsh structure dictates both the capacity of tidal rearing habitats to support juvenile salmon and opportunity for juvenile salmon to access that capacity. Simenstad *et al.* (2000) lists four suggestions for incorporating landscape structure in tidal marsh restoration:

> "1. Use natural landscape templates that are specific to the estuary and local region to guide restoration;
> 2. Emphasize corridors and other linkages among marshes and other tidal landscape elements that facilitate physiological, foraging and refuge requirements of different fish species and life history stages;
> 3. Incorporate landscape elements and a mosaic that maintain a natural diversity of primary producers and detritus sources; and,
> 4. Promote landscape structure that accommodates fish responses to climatic variability and natural disturbance regimes."

A common measure of habitat complexity is to compute a Shannon-Wiener diversity index of various habitat descriptors, such as depth, current velocity, substrate (Gorman and Karr 1978; Schlosser 1982; Angermeirer and Schlosser 1989), channel thalweg depth profiles, and woody debris tallies (Kaufmann and Faustini 2012). In addition to the above habitat descriptors, Coats *et al.* (1995) suggested measuring metrics specific to the marsh landscape: patch size and shape, connectivity and edge:patch area. Furthermore, Simenstad *et al.* (2000) proposes additional landscape metrics, such as heterogeneity of topography, vegetation patch structure, channel system order, the number of channels, average sinuous length of channels, length of channel edge, drainage density, and the occurrence, distribution and size of pans on the marsh plain (Simenstad et al. 2000). It has also been shown that bifurcation ratios can indicate opportunities for foraging interactions between prey being transported off the marsh and fish in larger channels (Coats et al. 1995; Simenstad et al. 2000). In addition to bifurcation ratios, juvenile fish access to dendritic tidal channels, residence in tidal freshwater sloughs, and the allometric relationships among estuarine sloughs (Simenstad et al. 2000) can be important habitat descriptors specific to juvenile salmon.

For more information see the Wetland Evolution Conceptual Model.

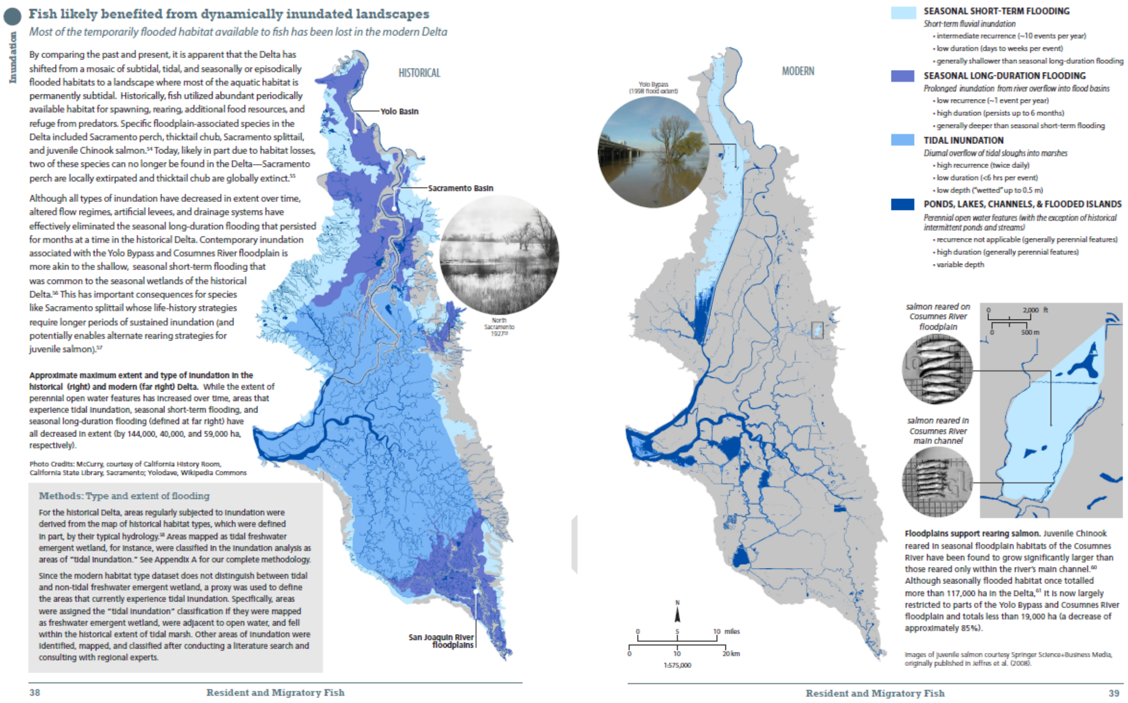
Connectivity

In the salmon CM connectivity describes the strength of the connection between the restoration site and established marshes and migration corridors. Established marshes and migration corridors act as source populations for vegetation, detritus, nekton and invertebrates for the restoration site, which will influence marsh evolution, habitat function and access to the restoration site by juvenile salmon. Connectivity between marshes also provides refuge for juvenile salmon (Simenstad et al. 2000; Hering et al. 2010; Hansen et al. 2012). Marsh corridors that fringe distributary channels and sloughs in tidal floodplains and deltas provide shallow water habitat for foraging and refuge from predation during migration (Simenstad et al. 2000). For salmon, connectivity is not simply flowing water;, although necessary there are many factors that make habitat suitable for salmon that must be considered to provide juvenile salmon with the opportunity to access a tidal wetland restoration site and promote fitness.

Marsh corridors and shallow water habitat that fringe channels may have a relatively small contribution on the total marsh habitat in an estuary, but they may have a large effect on outmigrating salmon. For example, a study by Hansen *et al*. (2012) found that backwater channel habitat use by juvenile Chinook Salmon mitigated for energy declines incurred during migration in the Columbia River freshwater tidal estuary, suggesting that these fish were foraging at a greater rate than fish only migrating through the mainstem channel. Beamer *et al.* (2005) found that mainstem edge, backwaters and off-channel habitat were essential for foraging and refuge of Chinook Salmon during their primarily density dependent migration through the Skagit River system. In the shallow water foraging habitat section we discussed the use of shallow water habitat primarily by small fish, and while fish size has been related to fidelity of shallow-water tidal habitat (Bottom et al. 2012), it is not the only indicator for how a juvenile salmon may use marsh habitat (Simenstad et al. 2000). For example, Chum (*Oncorhynchus keta*), Pink (*Oncorhynchus gorbuscha*) and Chinook Salmon fry occur at the same size and time in tidal channels, but Pink salmon will use the habitat primarily for migration, while chum will rear for up to two weeks and Chinook may be present for several months (Reimers 1973, Levy and Northcote 1981, 1982, Schreffler et al. 1990, Simenstad et al. 2000, Bottom et al. 2005, Herring et al. 2010). Therefore, juvenile salmon use estuarine mosaics for migration and growth at the same time and that variability must be accounted for by providing connectivity whenever possible.

Connectivity will also impact migration route and timing for juvenile Chinook Salmon on their seaward migration. Route choice and timing may be influenced by parental migration timing, winter and spring water temperatures and flows, the outcome of intraspecific competition and interspecific interactions in freshwater, and physiological cues for juvenile life history transitions (Simenstad et al. 2000). All of which occur across a continuous mosaic of fluvial and tidal microhabitats. Unnatural obstructions along migration corridors have been shown to delay migrants, which can result in a mismatch of environmental cues and migration-timing adaptations (Caudill et al. 2007; Marshall et al. 2011; Schaller et al. 2014). These anthropogenic barriers to migration can have severe consequences for survival. In one study 76% of the juvenile Chinook Salmon died in the marine environment as a consequence of their obstructed outmigration and that delayed mortality increased with the number of dam passages and decreased with increased speed (Schaller et al. 2014). Comparisons of pre- and post-construction of the Ice Harbor Dam, showed that impoundments delayed migration time by close to twice as many days regardless of flow levels, post construction (Bentley and Raymond 1976). Dams are not the only form of habitat fragmentation; one study showed that several salmon species were negatively related to riparian fragmentation (Andrew and Wulder 2011). Another study tested the use of baffles in stream culverts to reduce fragmentation, and found that species richness increased with age of baffles, indicating that culverts changed fish populations at the whole-stream scale and restoration effects of reconnection with baffles can take decades (Favaro et al. 2014).

The Delta is an intensively human modified landscape; the channel network no longer operates across predictable gradients for native fish, resulting in a less coherent landscape to navigate (SFEI-ASC 2014). Specifically, connectivity between large channels for shipping and water conveyance through canals, meander cutoffs, cross-levees and dredged channels have homogenized environmental cues and produced unnatural routes of migration (SFEI-ASC 2014). Juvenile salmon migration must contend with 1770 km of levees, a channel network of large dredged channels connected by channel cuts and loop channels, unnatural flow paths and extensive loss of small marsh channel networks replaced by the flooding of subsided islands (SFEI-ASC 2014, Figure 5). Furthermore, these connectivity changes have had extensive consequences for the Delta ecosystem: two of the six interrelated drivers of change that are implicated in the loss of ecological function in the Delta in the SFEI-ASC 2014 report concern losses in connectivity (SFEI-ASC 2014). Delayed migration in the Delta has been shown by acoustic tagging studies; outmigration was fastest through the upper Sacramento River and slowest through the Delta (Michel et al 2012). Both longer travel times and lower survival have been documented for juvenile salmon migrating through the interior Delta (Baker and Morhardt 2001; Brandes and McLain 2001; Newman and Brandes 2010; Perry et al. 2010). In one study, survival probabilities were negatively associated with water exports, suggesting that water exports affect migration by way of limiting connectivity (Newman and Brandes 2010).



Fish likely benefited from dynamically inundated landscapes

Most of the temporarily flooded habitat available to fish has been lost in the modern Delta

By comparing the past and present, it is apparent that the Delta has shifted from a mosaic of subtidal, tidal, and seasonally or episodically flooded habitats to a landscape where most of the aquatic habitat is permanently subtidal. Historically, fish utilized abundant periodically available habitat for spawning, rearing, additional food resources, and refuge from predators. Specific floodplain-associated species in the Delta included Sacramento perch, thicktail chub, Sacramento splittail, and juvenile Chinook salmon.[54] Today, likely in part due to habitat losses, two of these species can no longer be found in the Delta—Sacramento perch are locally extirpated and thicktail chub are globally extinct.[55]

Although all types of inundation have decreased in extent over time, altered flow regimes, artificial levees, and drainage systems have effectively eliminated the seasonal long-duration flooding that persisted for months at a time in the historical Delta. Contemporary inundation associated with the Yolo Bypass and Cosumnes River floodplain is more akin to the shallow, seasonal short-term flooding that was common to the seasonal wetlands of the historical Delta.[56] This has important consequences for species like Sacramento splittail whose life-history strategies require longer periods of sustained inundation (and potentially enables alternate rearing strategies for juvenile salmon).[57]

Approximate maximum extent and type of inundation in the historical (right) and modern (far right) Delta. While the extent of perennial open water features has increased over time, areas that experience tidal inundation, seasonal short-term flooding, and seasonal long-duration flooding (defined at far right) have all decreased in extent (by 144,000, 40,000, and 59,000 ha, respectively).

Photo Credits: McCurry, courtesy of California History Room, California State Library, Sacramento; Yolodave, Wikipedia Commons

Methods: Type and extent of flooding

For the historical Delta, areas regularly subjected to inundation were derived from the map of historical habitat types, which were defined in part, by their typical hydrology.[58] Areas mapped as tidal freshwater emergent wetland, for instance, were classified in the inundation analysis as areas of "tidal inundation." See Appendix A for our complete methodology.

Since the modern habitat type dataset does not distinguish between tidal and non-tidal freshwater emergent wetland, a proxy was used to define the areas that currently experience tidal inundation. Specifically, areas were assigned the "tidal inundation" classification if they were mapped as freshwater emergent wetland, were adjacent to open water, and fell within the historical extent of tidal marsh. Other areas of inundation were identified, mapped, and classified after conducting a literature search and consulting with regional experts.

38 Resident and Migratory Fish

SEASONAL SHORT-TERM FLOODING
Short-term fluvial inundation
- intermediate recurrence (~10 events per year)
- low duration (days to weeks per event)
- generally shallower than seasonal long-duration flooding

SEASONAL LONG-DURATION FLOODING
Prolonged inundation from river overflow into flood basins
- low recurrence (~1 event per year)
- high duration (persists up to 6 months)
- generally deeper than seasonal short-term flooding

TIDAL INUNDATION
Diurnal overflow of tidal sloughs into marshes
- high recurrence (twice daily)
- low duration (<6 hrs per event)
- low depth ("wetted" up to 0.5 m)

PONDS, LAKES, CHANNELS, & FLOODED ISLANDS
Perennial open water features (with the exception of historical intermittent ponds and streams)
- recurrence not applicable (generally perennial features)
- high duration (generally perennial features)
- variable depth

Floodplains support rearing salmon. Juvenile Chinook reared in seasonal floodplain habitats of the Cosumnes River have been found to grow significantly larger than those reared only within the river's main channel.[60] Although seasonally flooded habitat once totalled more than 117,000 ha in the Delta,[61] it is now largely restricted to parts of the Yolo Bypass and Cosumnes River floodplain and totals less than 19,000 ha (a decrease of approximately 85%).

Images of juvenile salmon courtesy Springer Science+Business Media, originally published in Jeffres et al. (2008).

Resident and Migratory Fish 39

Figure 5: Approximate maximum extent and type of inundation in the historical (left) and modern (right) Delta, which represents the comparitive loss of connectivity for resident and migratiory fish (SFEI-ASC 2014).

Counteracting habitat fragmentation with the removal of unnatural obstacles to migration and establishment of continuous marsh fringes along migration corridors may ameliorate these negative consequences of delayed migration and promote growth, survival and the expression of juvenile life history diversity (Tanner et al. 2002; Bottom et al. 2005; Roegner et al. 2010). Considering both connectivity and structural complexity when evaluating restoration projects requires a landscape approach. However, urbanized estuaries can be constrained by the industries they support. For this reason site selection must be taken in context with the landscape, such as the influence of contaminants, invasions by non-native species, alterations to flow (proximity to water diversions) and proximity to donor populations (established marshes and migration corridors) (Simenstad et al. 2000). A monitoring effort may, for example, monitor the establishment and progression of marsh fringes along migration corridors connecting the restoration site to established marshes and the ocean, and the extent of shallow water habitat surrounding the restoration site throughout the tidal cycle. Marsh fringes are defined as those habitats flooded for half of most flood tides (Simenstad et al. 2000), also known as lateral channel migration (Beamer et al. 2005). It may also be advisable to mathematically model the likely paths for juvenile fish to and from the restoration site, and identify those paths' fringe marshes, shallow water habitat, temperature and risks (e.g. water diversions, obstructions, predation bottlenecks and contaminant discharge sites).

For more information see the Transport Conceptual Model.

3. Habitat Attributes

Toxicity

Contaminant exposure has been shown to adversely affect fish reproduction, influencing fecundity, egg survival, spawning behavior, and time to reproductive maturity (Spromberg and Meador 2005). Previous work has also shown that maternally transferred contaminants in eggs (e.g. PCBs) can negatively affect the hatching success in Chinook Salmon (Ankley et al. 1991). Other environmental pollutants can also influence hormonal activity and cause disruptions of the reproductive endocrine system or directly affect gamete viability (Kime 1999). There is also increased support, which suggests that Chinook Salmon exposed to environmental concentrations of contaminants experience immune suppression and increased susceptibility to disease (Arkoosh et al. 1994). In fact, a number of studies have indicated that fish exposed to chemical contaminants even had a higher predisposition to infection months after their removal from contaminated areas (Arkoosh et al. 1998; Arkoosh et al. 2001). It is still unclear whether exposure of juvenile salmon to contaminants has any long term effects on immunity; however, a lab-based study by Milston *et al.* (2003) observed humoral immunosuppression in Chinook one year after their embryos were briefly exposed to sub-lethal concentrations of dichlorodiphenyldichloroethylene (DDE).

A number of contaminants such as metals and organics also inhibit somatic growth in fish (Roch and McCarter 1984; Finlayson and Verrue 1985; Glubokov 1990; Stein et al. 1995; Hansen et al. 2002). For example, exposure to environmentally relevant concentrations of certain metals (e.g. copper, cadmium, and zinc) has a negative effect on the growth of Chinook Salmon fry (Roch and McCarter 1984). Since size is linked to age-specific survival rates, age at reproductive maturity, fecundity, spawning success (Bagenal 1969; Roff 1984; Sigler and Sigler 1987; Wootton 1992; Koztowski 1996), alterations in somatic growth could have both short and long term fitness effects.

A number of studies have also demonstrated that exposures to contaminants (e.g. organophosphate and pyrethroid insecticides) had effects on olfactory response in salmon (Scholz et al., 2000; Sandahl et al., 2004; Moore and Waring, 2001). Behavioral changes in fishes, such as decreased ability to detect prey and avoid predation, have been associated with exposure to very low concentrations of contaminants (Linbo et al. 2006, Lurling et al. 2007, Sandahl et al. 2004, Sandahl et al. 2007). Exposure to environmentally relevant concentrations of copper eliminated juvenile salmon avoidance response to predator cues (I. Werner, unpublished data).

While contaminants can directly influence the fitness of Chinook Salmon, they also have the potential to indirectly influence Chinook by influencing the quality and quantity of their food (such as amphipods, Werner et al. 2010). Lab based studies suggest that contaminants can negatively affect the feeding rate of primary consumers (Day and Kaushik 1987), which may influence their growth and nutritional value. Food limitation for Chinook may also result from shifts in community structure, as contaminant exposure more adversely impacts larger species of primary consumers (Hurlbert et al. 1972; Papst and Boyer 1980; Kaushik et al. 1985; Day and Kaushik 1987; Yasuno et al. 1988; Hanazato and Yasuno 1990b; Hanazato 1991a). Other work has shown that contaminant exposure can reduce population growth rates in zooplankton (Hanazato and Dodson 1992), which may result in contaminant-mediated food limitation for Chinook Salmon.

For the purposes of the salmon conceptual model, there are two pathways to toxicity: (1) proximity to contaminant discharge site → contaminant loading → toxicity: this pathway illustrates that the level of toxicity is dependent on proximity to the contaminant discharge site as well as the concentrations of contaminants entering the system; and (2) proximity to contaminant discharge site → contaminant loading → food production → food availability and quality: similar to the pathway leading to toxicity of fish (described above), this pathway illustrates that food production and availability is dependent on proximity to the contaminant discharge site as well as the concentrations of contaminants entering the system.

Monitoring for contaminants in the San Francisco estuary has been somewhat limited, as many contaminants are difficult to detect and expensive to monitor (Scholz et al. 2012). As a result, contaminant data for the Delta region are particularly scarce, as are data regarding toxicity in Chinook and other fishes. Given the transient nature of species such as Chinook Salmon, it may be difficult to tease apart toxicity effects resulting from contaminant exposure within the wetland relative to other areas (pre- and post-restoration). Nevertheless, rather than monitoring toxicity levels in fish, monitoring efforts could focus on measuring several key contaminants within the restoration site and an adjacent reference site (e.g. pre- and post- restoration) to demonstrate differences in exposure levels. For example, monitoring efforts could focus on legacy pesticides (DDT and their families), currently used pesticides with high binding affinity (e.g. chlorpyrifos-AChE inhibitor, pyrethroids, firponil, and diuron-herbicide), and copper (known olfactory inhibitor for salmonids). Additional monitoring could examine food web effects, by examining whether there is a significant interaction between restoration and contaminant levels on the relative abundance of lower trophic levels.

Food availability and quality

Wetland resources can be important for Chinook Salmon at the population-level. For example, Reimers (1971) found that juvenile Chinook Salmon which entered the estuary in early summer and remained for a period of improved growth in the estuary represented ~90% of the returning spawners, despite representing a disproportionally small fraction of the total juvenile out-migrants. Estuarine rearing also contributes to life history variation (Bottom et al. 2005), survival (Magnusson and Hilborn 2003; Rechisky et al. 2012) and growth (Healey 1982; Maier and Simenstad 2009) among juvenile salmon. To be valuable to juvenile Chinook Salmon a wetland restoration site must have the capacity to promote juvenile salmon production (Simenstad and Cordell 2000). Food availability and quality is a very important and obvious necessity for juvenile salmon growth, survival, residency time within a tidal wetland and foraging success. For the purposes of the salmon CM there are two pathways to food availability and quality: (1) bathymetry - vegetation → structural complexity → shallow water foraging habitat → food availability and quality; and (2) proximity to contaminant discharge sites – bathymetry - vegetation → food productions/retention → food availability and quality. Flow, contaminant loading and competition also interact with food availability and quality within tier two and three. Flows will affect retention of prey as well as the detritus they feed upon. As mentioned in the previous section, contaminant loading could affect the prey communities for juvenile Chinook Salmon. Finally, competition will affect food availability if salmon and other forage fish densities are high in the tidal marsh restoration site, which will be discussed in further detail below. Food availability and quality is one of the most interconnected terms in the salmon CM, therefore we will be primarily be focusing on its relationship with food production and retention and shallow water foraging habitat, which act together to provide food availability and quality for juvenile Chinook Salmon.

Wetlands are generally characterized by highly productive foraging habitats, dominated by detritus-based food-webs (Healey 1982; Limm and Marchetti 2009; Maier and Simenstad 2009). Detritivores are the primary consumers that constitute juvenile salmon prey (such as chironomids), which depend upon detritus accumulation and decomposition, related to landscape structure (Simenstad et al. 2000). Shallow water foraging habitat provides a location for foraging, as well as promotes production and retention of organic matter and invertebrates. Food production is an obvious necessity for food availability. However other aspects of tidal wetland production can be important to food availability for juvenile Chinook Salmon. For example, one study showed that an abundance of nonnative prey production limited food availability and growth in juvenile salmon (Tiffan et al. 2014). Comparing both prey abundance and juvenile salmon diets can be useful to confirm that the prey available is also consumed and of high caloric value. A table listing the energy densities (kJ/g wet mass) of common estuarine prey for juvenile salmon can be found in the supplemental material of David et al. 2014 (Appendix Table 1), but insects are generally considered high quality prey for juvenile Chinook Salmon (Duffy et al. 2010). Finally, increased food availability (by way of retention and production) allows juvenile salmon the opportunity to exploit habitats that might be otherwise bioenergetically unsuitable, such as higher velocity microhabitat (Rosenfeld et al. 2005) and stressful water temperatures (Beauchamp et al. 2009). Studies have found that growth may be more limited by prey availability and quality than by temperature depending on size (Beauchamp et al. 2009). This tradeoff between prey quality and temperature thresholds is one example of why we advocate for monitoring a habitat function, such as capacity. The effect of food availability and quality on juvenile salmon will be dependent upon a salmon's response to several habitat attributes as well as competition.

The primary groups we will be focusing on as juvenile Chinook prey are surface epifauna, sedentary infauna, epibenthic plankters, pelagic zooplankton, neustonic and drift invertebrates, and motile fish (Simenstad et al. 1991). Prey will vary depending upon the restoration site's position along the Bay-Delta ecocline, however certain prey taxa are notable prey resources for juvenile salmon: fly larvae and adult insects, gammarid amphipods, and harpacticoid copepods (Simenstad and Cordell 2000). Simenstad *et al.* (1991), "Estuarine Habitat Assessment Protocol" recommends monitoring species occurrence, density and standing stock (wet weight biomass) for all food groups, as well as settled volume or percent cover for mobile and sessile organisms, respectively. However, monitoring prey resources is not without its challenges; prey populations can vary over space and time, and may involve substantial sampling to be quantified (Simenstad and Cordell 2000). Additionally, juvenile salmon eat a wide variety of prey that occupy an assortment of microhabitats. Due to the variety of organisms eaten by juvenile Chinook Salmon there are an assortment of methods for collecting food availability and quality samples: benthic cores, excavation with benthic quadrats, epibenthic suction pumps, paired nets towed at a fixed depth, neuston netting and purse seining (Simenstad et al. 1991). Additionally, prey quality is a more dynamic metric, which may be more effectively measured through the caloric value, number and weight of items within diet samples.

Predation and competition

Predation and competition within the juvenile Chinook Salmon tidal wetland conceptual model are linked to many factors. Predation and competition represent the population of piscivorous fish in the Bay-Delta and their interactions with juvenile Chinook Salmon. This includes Striped Bass, Largemouth Bass, Smallmouth Bass, White Catfish, Channel Catfish, Sacramento Pikeminnow, and juvenile Chinook Salmon (Grossman et al. 2013). The gape limitation and size of any particular fish will determine if the relationship is one of predator-prey or competitor. Predation and competition are affected by turbidity, the piscivore population, and the extent of shallow water foraging habitat and refuge. These factors are further contextualized within the landscape by bathymetry, vegetation, the erodible sediment supply and the structural complexity and connectivity within the marsh. These many influences have historically made predation and competition difficult to directly measure in relation to tidal wetland restoration. Additionally predation and competition may fluctuate seasonally or annually and therefor may have timing-specific influences on particular life histories and sizes of juvenile salmon (Grossman et al. 2013; Osterback et al. 2014). In the context of the population, it is uncertain if predation on smaller life stages is compensated by increased growth or survival by those fish that may thrive in tidal wetlands (Brown 2003). Currently, juvenile Chinook Salmon survival to ocean entry ranges from 3-16% for all runs in studies using acoustic tags (Grossman et al. 2013), and therefore predation and competition are a major concern to fisheries managers.

In the salmon CM there are three pathways to predation and competition, one of which is: (1) vegetation - bathymetry → structural complexity →shallow water foraging habitat → predation/competition: this represents how the structure of the tidal marsh habitat can provide refuge from predation or create predation hotspots, and affect density dependent competition. Generally, estuaries are thought to reduce predation risk for juvenile fish, and emergent and submerged vegetation can provide cover for small fish from larger predators (Kilgore et al. 1989). However, in the Delta results from a tethering study suggest that predation rates are high regardless of vegetation (Grimaldo et al. 2000). Additionally, artificial structures can create habitat bottlenecks in human modified landscapes. These artificial structures, such as forebays, dams and water intakes have been shown to enable and intensify predation on salmon (Grossman et al. 2013). One study showed a 10-29% decrease in migrating juvenile salmon survival due to striped bass predation below an irrigation dam on the lower Mokelumne River (Sabal 2014). Therefore, vegetation and habitat features within tidal wetlands can provide refuge from or enhance predation risk (Kilgore et al. 1989; Grimaldo et al. 2000).

Tidal wetland habitat may be a limited resource, as only 5% of the historical Delta is currently accessible to juvenile salmon. Theoretically, if a restoration site provides highly productive shallow water foraging habitat for juvenile salmon production it will alleviate density dependent competition for estuarine rearing habitat. Population density has been shown to negatively affect both food consumption and individual growth rates and is largely influenced by intraspecific competition for limited resources (Amundsen et al. 2007). However, we are unaware of an estimates of the density of juvenile salmon in the Delta's currently accessible tidal wetlands or if those density estimates can be related to intraspecific competition. Additionally, juvenile salmon may compete for prey resources with other forage fish species that are not part of the predator population, such as Delta Smelt (*Hypomesus transpacificus*), Splittail (*Pogonichthys macrolepidotus*), Shad (*Dorosoma petenense* and *Alosa sapidissima*), Bluegill (*Lepomis macrochirus*), Silversides (*Menidia beryllina*), juvenile Hitch (*Lavinia exilicauda*) and Tule Perch (*Hysterocarpus traskii*). Niche overlap with and competitive exclusion could limit both the capacity of a restoration site to support juvenile salmon production and the opportunity for a juvenile salmon to access a tidal wetland (Gilpin 1974).

The second pathway to predation and competition in the salmon CM is: (2) vegetation - erodible sediment supply → turbidity → predation/competition: which represents the visibility of forage fish by predators and is therefore a modifier of predation risk (Baxter, 2015). Turbidity affects predation rates and perceived risk of predation, as explained above in "Turbidity". Finally, a clear impact on predation risk and competition in the salmon CM is the distribution, abundance and species of piscivores within and surrounding the restoration site: (3) vegetation → predators → predation/competition. Unfortunately, there is limited information on the predator population and its association with habitats used by juvenile Chinook Salmon, such as shallow water foraging habitat. Additionally, there is little population-level piscivore data or population-level effects on juvenile chinook (Grossman et al. 2013). However, Grossman *et al.* (2013) did predict that there are likely enough predators to consume all the available salmon, which makes habitat specific predation risk a primary focus for resource managers.

In the Delta, the decline in the abundance of native prey species over the last several decades has coincided with increased invasions of centrarchids (Nobriga and Chotkowski 2000; Lindley and Mohr 2003). Evidence for direct predation effects is rare, but some piscivores may have a higher potential to effect salmon populations. For example, Largemouth Bass become predominantly piscivorous at smaller sizes than the native predator, the Sacramento Pikeminnow (Nobriga 2007). Additionally, native predators may have a larger behavior effect, but lower predation rate. A study examining juvenile Chinook Salmon anti-predator responses to native Pikeminnow and nonnative Smallmouth Bass showed a strong behavioral response to native predators and no response to chemical cues of Smallmouth Bass (Kuehne and Olden 2012). This suggests that non-native predators could inflict proportionally higher mortality on prey with limited coexistence. This strong behavioral response to native predators (Sacramento pikeminnow) also affects habitat use of their prey (Brown and Moyle 1991, Brown and Brasher 1995, Brown and Moyle 1997). Preliminary results from two studies also indicate that non-native piscivores may act as prey subsidies; supporting large predator populations by providing an alternative prey resource for the Delta predator community (Hayes unpublished; Schreier unpublished).

Although predation risk may be crippling in the Delta, fisheries managers and restoration practitioners can aid juvenile salmon in negotiating those risks. Site selection and restoration design may lessen the invasion of non-native predators; it has been show that within restoration and reference sites native fish are more often found in intertidal habitat and invasive fish densities were highest in deep, lake-like breached islands (Simenstad et al. 2000). Additionally, a manipulative study tested the effects of increased flow and predator removal, and showed that both can be useful management applications for increased survival of fall-run hatchery juvenile chinook Salmon (86-121 FL) (Cavallo et al. 2013). Limiting sublethal stressors may make juvenile salmon less susceptible to predation. For example, monitoring maximum daily temperatures in advisable, as temperatures exceeding 18°C may decrease prey survival by increasing the appetite of predators (Anderson 2003, Williams 2006). Contaminants can also affect predation risk; exposure to copper was shown to completely eliminate the avoidance response of juvenile salmon to a predator cue (I. Werner, unpublished data). In addition to sublethal stressors, climate change may also affect the distribution overlap of juvenile salmon and their predators. One study showed that climate warming would increase the spatial overlap between Chinook Salmon and nonnative Smallmouth Bass, because Bass would no longer be restricted by cooler water habitats, limiting in-stream rearing for Chinook (Lawrence et al. 2014), and potentially increasing the importance of low predation risk estuarine rearing in the future. Predation and competition are very complex issues that will be difficult to monitor and are likely patchily distributed and seasonally variable. Therefore, special studies may be necessary to examine site-specific hypotheses concerning predation and competition risks for juvenile salmon.

Water temperature

The conceptual model for Delta Smelt includes an in-depth introduction to water temperature, its relationship with air temperature, and its historic range in the Delta, which is summarized here. Water temperature can affect the physical and chemical properties of water, such as dissolved oxygen, primary productivity and cycling of nutrients and other chemicals, as well as the metabolic rates and life histories of aquatic organisms, including activity, digestion, growth and reproductive development. In the Delta, temperatures range between monthly averages of nine to twenty-two degrees Celsius, seasonally. The hottest months are July and August, and water temperatures are usually highest in the south Delta (maximum 28 °C) and lowest in Suisun and San Pablo Bays (maximum 24°C).

In the Chinook Salmon conceptual model we assume that air temperature is the major driver of water temperature (Wagner et al. 2012). However heterogeneity in water temperature can also be affected by vegetation, bathymetry and position within the landscape. For example, a site highly connected to a cool water source, heavily shaded or closer to the ocean within the Bay-Delta landscape may vary within a cooler range of temperatures. In tier three, water temperature is included within habitat opportunity and habitat capacity. Moderate to high water temperatures are thought to be a bioenergetic limitation for juvenile Chinook Salmon and temperatures above 19 °C are associated with shallow wetland habitat exclusion (Bottom et al. 2011).

Traditional techniques (measuring chronic exposure to preselected water temperatures) have estimated the upper lethal temperature limit for Chinook Salmon in the Sacramento-San Joaquin River system at approximately 25.8 °C (Orsi 1971). Additionally studies have found decreased growth at around 20 °C with similar techniques (Geist et al. 2014, Marine and Cech 2004). These temperature thresholds are likely due to a series of physiological responses to stressful water temperatures. A study inflicting acute temperature stress observed a switch to anaerobic metabolism and cardiac arrhythmias at 25 °C, which suggested the oxygen supply to systemic and cardiac tissues was compromised at temperatures exceeding 25 °C (Clark et al. 2008). The acute temperature stress tested in the above experiments is possible; however, we expect the water temperature experienced by wild juvenile Chinook to fluctuate gradually with daily solar warming. A study that accounts for this natural fluctuation showed that juvenile Chinook

Salmon growth began declining at daily maximum water temperatures exceeding 23 °C (Geist et al. 2014). It may be important to consider lethal temperature thresholds as well as daily maximum water temperature that could inhibit growth or access.

To further complicate the establishment of temperature thresholds, thermal tolerances and optimal temperatures for growth shift with increasing body size and daily ration sizes (Beauchamp 2009). Thermal tolerance for juvenile salmon is further convoluted by the maternal influence on offspring; it has been shown that females with larger eggs have more thermally tolerant offspring (Muniz et al. 2014). Additionally, for smaller, younger life stages, simulations suggest that growth is limited by feeding rate and prey quality more frequently than by temperature (Beauchamp 2009). Therefore, if more food is available smaller fish may be able to withstand and benefit from higher temperatures than if less prey items are available. Parental genes, size and the prior life stage's energetic performance both complicate the study of and diversify salmon phenotypes. It may be advisable to monitor daily maximum temperatures at several depths and a variety of locations within the restoration area (e.g. associated with vegetation, bathymetry and proximity to source water inflow).

Entrainment and stranding risk

Water diversions and fluctuations in flow can result in the impingement, entrainment, and/or stranding of juvenile Chinook Salmon across a range of spatial and temporal scales. At water diversion sites (e.g. pumps), juvenile Chinook Salmon can be impinged on gates or barriers, and entrained within water control structures, which can cause injury, stress, or mortality (Moyle and Israel 2005; Castillo et al. 2012). In general, the risk of impingement on and entrainment within water control structures has been shown to increase with prolonged exposure time (Mussen et al. 2014), the amount of water being diverted (Carlson et al. 2007; Kimmerer 2008; Mussen et al. 2013), proximity to the diversion site (Zueg and Cavallo 2014), turbidity (Nobriga et al. 2004; Mussen et al 2013), and increased river flow (Grimaldo et al. 2009; Cavallo et al. 2015). Although fish screens are often installed on water diversions to limit entrainment risk, the effectiveness of most fish screens is largely unknown (Moyle and Israel 2005) and approximately 98% of water diversions within the San Francisco Estuary are reported to not be adequately screened (Herren and Kawasaki 2001).

In the salmon CM there are three pathways to entrainment and stranding risk as well as an interaction between refuge and entrainment and stranding risk: (1) proximity to water diversion sites → water diversions → entrainment risk and stranding: this refers to risks associated with water conveyance and exports through the Delta Cross Channel, as well as smaller scale pumping from wetland management and irrigation; (2) flows → entrainment risk and stranding: this includes risks connected to both natural and anthropogenic flow fluctuations; and (3) entrainment risk and stranding ← → refuge: which pertains to those instances when juvenile Chinook Salmon are restricted to poor or suboptimal habitat as a consequence of water diversions and fluctuations in flow; reducing or eliminating their access to refuge.

At a larger scale, large water diversions can influence the distribution of juvenile Chinook Salmon throughout the San Francisco Estuary. Increases in water conveyance at the Delta Cross Channel or increased water exports at the CVP and SWP pumping facilities can influence local and regional hydrodynamic processes (Kimmerer 2004) and thereby juvenile salmon migration patterns (Kimmerer 2008; Perry et al. 2010). For example, numerous studies have demonstrated that tagged juvenile Chinook Salmon from the Sacramento River can be advected into the central Delta through the Delta Cross Channel during periods of water conveyance (Perry et al. 2010; Steel et al. 2012; Perry et al. 2013). In general, the survival of these juvenile Chinook Salmon is often decreased as they migrate through these alternative routes, presumably based on longer migration times through suboptimal habitats that can contain poor physiochemical conditions coupled with higher densities of non-native piscivorous fish (Gingras 1997; Kimmerer 2008; Grimaldo et al. 2009; Newman and Brandes 2010; Castillo et al 2012;

Perry et al. 2013). As a result, water diversions can greatly influence juvenile Chinook Salmon access to tidal wetland restoration sites.

Juvenile Chinook Salmon stranding has been attributed to both natural and anthropogenic flow fluctuations and often occurs more in engineered environments that have poor drainage (Bradford 1997; Sommer et al 2005). Nagrodski et al. (2012) defines stranding as an event where fish are restricted to poor or suboptimal habitat as a consequence of physical separation from a main body of water. The effects of stranding can be direct or indirect and depends on fish residency time (Nagrodski et al. 2012). For example, stranding can affect the survival of juvenile salmon through site desiccation, suffocation, or increased predation risk from birds, small mammals, and piscivorous fish (Quin and Buck 2001; Sargent and Galat 2002; Nagrodski et al. 2012). Alternatively, stranded fish can be subjected to increased temperatures, ammonium, and hypoxia that can influence metabolic activities and thereby the growth and locomotion of juvenile salmon (Sargent and Galat 2002). In addition, limited resources coupled with higher fish densities can cause increased inter- and intra-specific competition during stranding events (Magoulick and Kobza 2003). Many studies have demonstrated that dewatering from water diversions (e.g. floodplain or wetland management, irrigation, etc.), ship wakes, floods, droughts, or even tidal exchange can result in the stranding of juvenile Chinook Salmon (Sargent and Galat 2002; Fresh et al. 2003; Magoulick and Kobza 2003; Jones and Stuart 2008; Pearson and Skalski 2011). In addition, because juvenile Chinook Salmon fry are more likely to occur in low velocities and shallow depths relative to smolts (Everest and Chapman 1972; Roper et al. 1994; Bradford and Higgins 2001; Fresh et al. 2003; Sommer et al. 2005), smaller juveniles also may be more susceptible to stranding than larger individuals that have undergone smoltification. Therefore, tidal wetland restoration projects can reduce the likelihood of a stranding event by minimizing sudden reductions in flow and water levels during periods of salmon occupancy, and maintaining connectivity with main channels during changing water levels.

Refuge

Wetlands are dynamic systems and juvenile Chinook Salmon refuge is likely a moving target as its location and extent will change seasonal, by tidal cycle and with varying river discharges. However, the presence and proximity to refuge, whether it is refuge from extreme temperatures, predation, stranding or perceived refugia, may determine if a salmon enters a restored marsh complex, and influence residency time and growth. For example, it has been shown that behavioral thermoregulation of thermal heterogeneity in juvenile Coho salmon (*Oncorhynchus kisutch*) substantially increased growth rates (Armstrong et al. 2013). Thermal heterogeneity enabled these fish to mitigate the trade-offs between trophic and thermal resources for faster growth rates. Therefore, a diverse mosaic of foraging and refuge habitats can improve growth rates for juvenile salmon, maintaining high assimilation efficiencies without limiting food availability.

In the conceptual model there are two pathways to refuge as well as interactions between refuge and water temperature, predation, and entrainment risk: (1) vegetation - bathymetry → structural complexity → refuge: this pathway represents micro-habitat within the marsh, and will change as the vegetation community evolves, but in some cases will also be restricted by the existing bathymetry; and (2) proximity to established marshes and migration corridors → connectivity → refuge: this pathway reflects the opportunity for salmon to get to the restoration site and mitigate the risks that may exist within the marsh. This type of refuge may include the edges of the restoration site which make it accessible to salmon as well as the habitat the salmon may occupy on their way in and out of the marsh, which for salmon may be critical for residence time in the marsh. Several mark-recapture studies have shown that juvenile Chinook Salmon will move into flooded marsh channels during high tides (Congleton et al. 1981; Levy and Northcote 1982; Shreffler et al. 1990). In one study, individuals entered and left a single marsh channel over multiple successive tidal cycles (Hering et al. 2010). Therefore, the refuge adjacent to a marsh or at the marsh entrance may dictate whether a juvenile salmon can or will return after low tides.

The design and monitoring of such a dynamic and interconnected habitat function may be difficult but, could be pursued with a habitat mapping framework. For example, refuge can be measured by defining a combination of thresholds or habitat preferences for salmon. Stressful features of the environment can be defined with respect to warm temperature, water diversion sites, and habitats in which they would be vulnerable to predators or where predators have been found. The mapping could include geomorphic features, such as total edge or penetration of tidal channels and structural habitat features such as marsh vegetation height, overhanging vegetation and the variation in the depth of watered habitats (Simenstad and Cordell 2000). If some of these factors do not already exist in a GIS database, part of the monitoring plan may be to collect these data in the field and record their location.

4. Juvenile Salmon Responses

Realized function

Realized function is a concept recommended by Simenstad and Cordell (2000), and represents a direct measure of physiological or behavioral responses to a particular habitat that promotes fitness and survival. This includes, habitat-specific residence time, foraging success and growth (Simenstad and Cordell 2000), as well as life history diversity (measured by the variation in individual size and timing when occupying a restored wetland) and the seasonal timing of juvenile salmon occupancy in the restored wetland compared to other sites within the Delta. It will also be valuable to use these salmon responses to validate the relationships between capacity, opportunity and realized function within and among restoration sites (Simenstad and Cordell 2000).

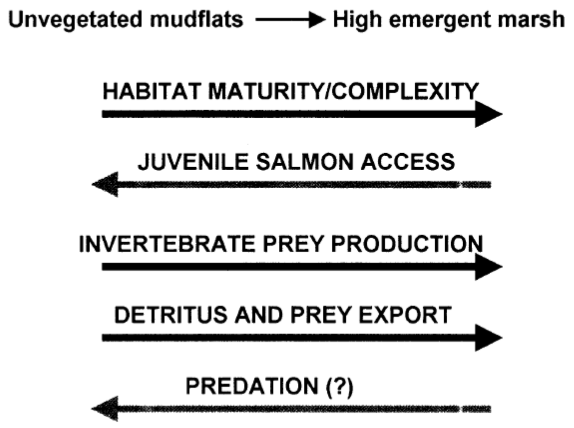
Although improvements in individual juvenile Chinook survival and population-level productivity may be of the most interest to resource managers, direct measurements will be challenging. Simenstad and Cordell (2000) recommend “self-monitoring”, by way of testing if the occupation of fish in a restored habitat equates to improvement in surrogate metrics for survival, such as residence time, foraging success or growth equivalent to that in a reference habitat. For example, wetland restoration in the Nisqually River delta was evaluated by a combination of sampling fish assemblages, water temperature, diet composition, consumption rates and bioenergetics modeling (David et al. 2014). In addition, we recommend an evaluation of diversity in growth rates and in the fish size, timing and duration of rearing habitat use in relation to adult returns, which may aid in incorporating long-term population-level productivity and resilience. For example, a change in estuarine residency and migration patterns was documented after the removal of dikes in the Salmon River estuary with mark-recapture and abundance studies (Bottom et al. 2005). In addition to the expansion in life history diversity, the mark-recapture study revealed diversity within individual salmon behavior and residency of restored marsh habitat (Hering et al. 2010). Otoliths can also be used to track changes in juvenile life history diversity at a broader ecotype scale (Miller et al. 2010).

Although growth is thought to be a more comprehensive and robust metric for evaluating habitat (Simenstad and Cordell 2000), it is influenced by many factors including a fish’s condition in a previous stage or habitat (Beauchamp 2009), and that variability can complicate its interpretation (Appendix Table 2). This is shown in Appendix Table 2, which summarizes the variability in primarily otolith-derived growth rates for juvenile Chinook Salmon from estuaries across the west coast. Every metric will have some disadvantages; the recovery of marked fish can be difficult in open systems and depending on the tag can be restricted to only the largest sized fish, and otoliths studies can be subjective (Campana 1985). It is likely that several metrics will need to be use in tandem to account for these shortcomings and aid in a clearer interpretation of the relationship between juvenile salmon fitness and tidal wetland restoration. Many of these monitoring challenges arise from essential knowledge gaps in our understanding of salmon habitat restoration (Simenstad and Cordell 2000). Namely, (1) how the processes, patterns and rates of restored habitat evolution are linked to habitat functions essential to juvenile salmon; (2) how to measure refuge (physiological requirements and from predation); and, (3) how to account for anadromous species' integration of the landscape instead of individual sites (Simenstad and Cordell 2000). For this reason, a series of site, species and life history-specific special studies may be necessary to monitor realized function. It would also be useful for this study to not only monitor change, but also contribute to the body of knowledge on tidal restoration and those processes that govern its rehabilitation.

Conclusion

Although difficult to monitor, wetland restoration efforts have been shown to positively affect juvenile Chinook Salmon populations (Roegner et al. 2010, Bottom et al. 2005, Hering et al. 2010, Greene and Beamer 2011). Often different metrics will evaluate different factors, many of which must be taken together to understand how or if a habitat supports fish production (Simenstad and Cordell 2000). For example, restoration in the Skagit River tidal delta did improve connectivity and capacity, which was demonstrated through increases in local rearing densities for juvenile Chinook Salmon, and system-wide density shifts (Greene and Beamer 2011). By examining tidal restoration efforts such as the Skagit River delta, this document provides guidance for those actions mandated by FRPA. FRPA's restoration actions specific to Chinook Salmon pertain primarily to floodplain rearing habitat in the lower Sacramento River basin (e.g. Liberty Island and Lower Cache Slough) (FRPA 2012). These mandates are responding to water project operations (i.e. flood management and storage) which limit the availability of tidal freshwater floodplain habitats to juvenile salmon by reducing the frequency and duration of seasonal over-bank flows (FRPA 2012). More specifically FRPA plans to increase the year-to-year inundation frequency of existing floodplains, restore rearing habitat attributes to suitable areas, and create side channels and re-created floodplain terrace areas (FRPA 2012). The plan also includes "additional potential anadromous actions", such as improvements in adult fish passage through those restored and rehabilitated areas (FRPA 2012).

This conceptual model does not predict discrete outcomes or define specific expectations for juvenile salmon from tidal marsh restoration. Instead we describe the primary processes that will affect juvenile salmon responses to tidal wetland restoration, and advocate for a monitoring effort focused on habitat function (Figure 6). Within the salmon CM many of the properties in tier one are fixed or severely constrained and therefore a restoration site must be chosen within those existing conditions. Tier two highlights the major drivers of change within the restoration site, which will be dictated by the landscape, while tier three pertains to our understanding of how juvenile salmon perceive their environment and what may dictate the capacity and opportunity therein. The single arrow connecting tier three to tier four represents the challenges in assessing which of the many influences determines juvenile salmon responses to tidal wetland restoration, and the knowledge gaps associated with that relationship. Realized function is dependent on both the capacity



and opportunity available within the restoration site. Therefore, tier four lists the metrics of interest, but is also meant to emphasize the many ways to evaluate individual and population fitness, and should be evaluated together with capacity and opportunity metrics.

Figure 6: Change in habitat function with increasing successional development of estuarine wetlands (Simenstad and Cordell 2000).

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Appendix

Table 1: Energy densities compiled from the literature for prey items found in the stomachs of juvenile Chinook Salmon in the Nisqually River delta (David et al. 2014). A more complete table of prey items from several estuaries across the west coast of North America is in review (David et al. In review).

Prey type	Taxonomic resolution for analysis	Energy density (kJ/g wm)	Source	Source taxa and notes
Acari	Arachnida	5.320	Gray 2005	Araneae
Araneae	Arachnida	5.320	Gray 2005	Araneae
Pseudoscorpion	Arachnida	5.320	Gray 2005	Araneae
Hypogastruridae	Collembola	5.621	Gray 2005	Collembola
Onychiuridae	Collembola	5.621	Gray 2005	Collembola
Americorophium salmonis	Corophiidae	3.065	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Americorophium sp	Corophiidae	3.065	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Americorophium spinicorne	Corophiidae	3.065	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Amphipoda	Amphipoda, other	2.970	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Ampithoe sp	Amphipoda, other	2.970	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Ampithoe valida	Amphipoda, other	2.970	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Anisogammarus pugettensis	Amphipoda, other	2.875	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Brachyura megalopa	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Brachyura zoea	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Calanoida	Calanoida	4.620	Higgs et al. 1995	Mean of 8 values
Caridea megalopa	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Caridea zoea	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Cirripedia cipris	Cirripedia larva	2.160	Lucas et al. 1979	Cirripedia cipris
Cirripedia nauplii	Cirripedia larva	3.280	Lucas et al. 1979	Cirripedia nauplii
Corophiidae	Corophiidae	3.065	Cordell et al. 2011; Gray	Mean of values from both studies

			2005	
Crangon sp	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Cumacea	Cumacea	3.370	Bieber 2005	Other Crustacea
Cumella vulgaris	Cumacea	3.370	Bieber 2005	Other Crustacea
Decapoda megalopa	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Decapoda zoea	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Eogammarus confervicolus	Eogammarus confervicolus	2.875	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Epilabidocera longipedata	Calanoida	4.620	Higgs et al. 1995	Mean of 8 values
Gnorimosphaeroma oregonense	Isopoda	2.960	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Grandidierella japonica	Amphipoda, other	2.970	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Harpacticoida	Harpacticoida	4.620	Higgs et al. 1995	Mean of 8 values
Harpacticus uniremis group	Harpacticoida	4.620	Higgs et al. 1995	Mean of 8 values
Hyperiidae	Amphipoda, other	2.464	Davis et al. 1998	Hyperiid amphipods
Leptochelia dubia	Tanaidacea	3.370	Bieber 2005	Other Crustacea
Limnoria lignorum	Isopoda	2.960	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Monocorophium acherusicum	Corophiidae	3.065	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Monocorophium insidiosum	Corophiidae	3.065	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Monocorophium sp	Corophiidae	3.065	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Mysida	Mysida	3.550	Gray 2005	Mysida
Neomysis mercedis	Mysida	3.550	Gray 2005	Mysida
Nippoleucon hinumensis	Cumacea	3.370	Bieber 2005	Other Crustacea
Paguridae megalopa	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Pinnotheridae megalopa	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Pinnotheridae zoea	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea

Porcellanidae zoea	Decapoda larva	3.360	Higgs et al. 1995	Crab zoea
Sinelobus stanfordi	Tanaidacea	3.370	Bieber 2005	Other Crustacea
Sphaeromatidae	Isopoda	2.960	Cordell et al. 2011; Gray 2005	Mean of values from both studies
Talitridae	Amphipoda, other	3.040	Gray 2005	Talitridae
Tanaidacea	Tanaidacea	3.370	Bieber 2005	Other Crustacea
Unidentifiable crustacea	Crustacea, other	3.370	Bieber 2005	Other Crustacea
Aphididae	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Auchenorrhyncha	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Baetidae nymph	Ephemeroptera	3.664	Pizzul et al. 2009	Ephemeroptera
Brachycera	Diptera, other	8.920	Gray 2005	Other Diptera
Brachycera adult	Diptera, other	8.920	Gray 2005	Other Diptera
Brachycera pupa	Diptera, other	8.920	Gray 2005	Other Diptera
Cantharidae	Coleoptera	7.940	Gray 2005	Cantharidae adult
Carabidae	Coleoptera	10.085	Chen et al. 2004	Carabidae
Ceratopogonidae	Ceratopogonidae	3.830	Gray 2005	Chironomidae adult
Ceratopogonidae adult	Ceratopogonidae	3.830	Gray 2005	Chironomidae adult
Ceratopogonidae larva	Ceratopogonidae	2.580	Gray 2005	Diptera larva
Ceratopogonidae pupa	Ceratopogonidae	3.830	Gray 2005	Chironomidae adult
Cercopidae	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Chalcidoidea	Hymenoptera	12.670	Gray 2005	Hymenoptera
Chironomidae	Chironomidae	3.830	Gray 2005	Chironomidae adult
Chironomidae adult	Chironomidae	3.830	Gray 2005	Chironomidae adult
Chironomidae larva	Chironomidae	2.580	Gray 2005	Diptera larva
Chironomidae pupa	Chironomidae	3.830	Gray 2005	Chironomidae adult
Chloropidae adult	Diptera, other	8.920	Gray 2005	Other Diptera
Cicadellidae	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Coleoptera	Coleoptera	7.970	Gray 2005	Coleoptera adult
Coleoptera larva	Coleoptera	2.405	Gray 2005	Coleoptera larva
Delphacidae	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Diptera larva	Diptera, other	2.580	Gray 2005	Diptera larva
Diptera pupa	Diptera, other	8.920	Gray 2005	Other Diptera
Dolichopodidae	Dolichopodidae	6.200	Gray 2005	Dolichopodidae adult
Dolichopodidae adult	Dolichopodidae	6.200	Gray 2005	Dolichopodidae adult
Dolichopodidae larva	Dolichopodidae	2.580	Gray 2005	Diptera larva
Dolichopodidae pupa	Dolichopodidae	2.580	Gray 2005	Diptera larva

Empididae adult	Diptera, other	8.984	Brodmann and Reyer 1999	Empididae
Ephemerellidae nymph	Ephemeroptera	3.664	Pizzul et al. 2009	Ephemeroptera
Ephemeroptera nymph	Ephemeroptera	3.664	Pizzul et al. 2009	Ephemeroptera
Ephydridae	Ephydridae	8.920	Gray 2005	Other Diptera
Ephydridae adult	Ephydridae	8.920	Gray 2005	Other Diptera
Ephydridae larva	Ephydridae	2.580	Gray 2005	Diptera larva
Ephydridae pupa	Ephydridae	8.920	Gray 2005	Other Diptera
Figitidae	Hymenoptera	12.670	Gray 2005	Hymenoptera
Formicidae	Hymenoptera	5.689	Brodmann and Reyer 1999	Formicidae
Hemiptera	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Hemiptera parts	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Heteroptera	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Hymenoptera	Hymenoptera	12.670	Gray 2005	Hymenoptera
Ichneumonoidea	Hymenoptera	12.670	Gray 2005	Hymenoptera
Immature Sternorrhyncha	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Insecta larva	Insecta, other	7.412	Bieber 2005	Other Insecta
Lepidoptera	Lepidoptera	8.500	Gray 2005	Lepidoptera (Adult and larval)
Lepidoptera larva	Lepidoptera	8.500	Gray 2005	Lepidoptera (Adult and larval)
Limnephilidae	Trichoptera	7.760	Gray 2005	Trichoptera adult
Miridae	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Muscidae adult	Diptera, other	9.293	Brodmann and Reyer 1999	Muscidae
Muscidae larva	Diptera, other	2.580	Gray 2005	Diptera larva
Mymaridae	Hymenoptera	12.670	Gray 2005	Hymenoptera
Nematocera	Diptera, other	3.830	Gray 2005	Chironomidae adult
Nematocera adult	Diptera, other	3.830	Gray 2005	Chironomidae adult
Nematocera pupa	Diptera, other	3.830	Gray 2005	Chironomidae adult
Phoridae adult	Diptera, other	8.920	Gray 2005	Other Diptera
Proctotrupoidea	Hymenoptera	12.670	Gray 2005	Hymenoptera
Psocoptera	Psocoptera	7.412	Bieber 2005	Other Insecta
Psychodidae adult	Diptera, other	3.830	Gray 2005	Chironomidae adult
Psyllidae	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)

Saldidae	Hemiptera	10.930	Gray 2005	Hemiptera (adult and immature)
Scatopsidae	Diptera, other	3.830	Gray 2005	Chironomidae adult
Sciaridae	Diptera, other	3.830	Gray 2005	Chironomidae adult
Sciaridae adult	Diptera, other	3.830	Gray 2005	Chironomidae adult
Staphylinidae	Coleoptera	7.970	Gray 2005	Coleoptera adult
Staphylinidae larva	Coleoptera	7.970	Gray 2005	Coleoptera adult
Tipulidae	Diptera, other	7.947	Brodmann and Reyer 1999	Tipulidae
Tipulidae adult	Diptera, other	7.947	Brodmann and Reyer 1999	Tipulidae
Tipulidae larva	Diptera, other	2.580	Gray 2005	Diptera larva
Unidentifiable insecta	Insecta, other	7.412	Bieber 2005	Other Insecta
Unknown Diptera	Diptera, other	8.920	Gray 2005	Other Diptera
Unknown Diptera adult	Diptera, other	8.920	Gray 2005	Other Diptera
Fish	Fish	6.830	Higgs et al. 1995	Mean of 20 values
Macoma siphons	Bivalvia	3.574	Ciancio et al. 2007	Mean of three values
Nematoda	Nematoda	3.000	Author's estimate	
Nereidae	Polychaeta	1.980	Gray 2005	Annelida
Polychaeta	Polychaeta	1.980	Gray 2005	Annelida
Plant material	Plant material	3.850	Gray 2005	Plant material

Table 2: A summary of the variability in growth rates for juvenile Chinook Salmon from estuaries across the west coast, with examples of when estuary, habitat, year, season and origin might affect growth rates. Estuaries are listed from north to south along western North America, and freshwater to saline when information was available.

Estuary	Habitat	Growth Rate	Season/Year	Reference
Campbell River estuary		0.46 mm day^{-1} 0.55 mm day^{-1}	1982 (wild only) 1983 (wild only)	Levings et al. 1986
Strait of Georgia	Ocean entry	1.06 mm day^{-1}	1995, 1996	Zhang and Beamish 2000
Nitinat estuary		0.62 mm day^{-1}		Fedorenko et al. 1979
Nanaimo River estuary		0.71 mm day^{-1} 1.32 mm day^{-1}	 1975, 1976 (Spring)	Sibert 1975 Healey 1980
Fraser River estuary		0.56 mm day^{-1} 0.39 mm day^{-1}	1978 1979	Levy and Northcote 1981

Columbia River estuary	Freshwater tidal	0.23 mm day^{-1}	0.11 to 0.43 mm day^{-1} in 2010-2012	Goertler et al. 2015
	Brackish/ lower	0.4 mm day^{-1}	0.11 to 0.67 mm day^{-1} in 2003, 2004, 2005	Campbell 2010
Columbia River plume	Estuary exit/ocean entry	2.75 ± 0.36 μm	“good ocean conditions”: 1999-2002, 2006-2008	Miller et al. 2013
		3.09 ± 0.30 μm	“poor ocean conditions”: 1998, 2003-2005	
		1.04 ± 0.51 mm day^{-1}	Natural origin	Claiborne et al. 2014
		1.01 ± 0.44 mm day^{-1}	Hatchery origin	
Salmon River estuary	Upper estuary	0.52 ± 0.22 mm day^{-1}	2001, 2002	Volk et al. 2010
	Lower estuary	0.51 ± 0.24 mm day^{-1}	2001-2002	
		0.54 ± 0.16 mm day^{-1}	2003	
Coos Bay		0.29 mm day^{-1}	1987	Fisher and Pearcy 1981
		0.54 mm day^{-1}		
Sixes River estuary		0.9 mm day^{-1}	April to June	Reimers 1973
		0.07 mm day^{-1}	June to August	
		0.5 mm day^{-1}	Sept. to November	
	Freshwater	<3 μm	1979-1981	Neilson et al. 1985
	Estuary	>4 μm		
Sacramento River	Off channel	6.07 ± 1.27 μm	2001, 2002	Limm and Marchetti 2009
	Main channel	4.28 ± 0.88 μm		
		0.33 mm day^{-1}	0.26 to 0.40 mm day^{-1} in 1981	Kjelson et al. 1982
		0.52 ± 0.02 mm day^{-1}	1998	Sommer et al. 2001
		0.43 ± 0.03 mm day^{-1}	1999	
Sacramento River Delta	Freshwater	0.86 mm day^{-1}	0.57 to1.23 mm day^{-1} in 1980	Kjelson et al. 1982
		0.53 mm day^{-1}	0.40 to 0.69 mm day^{-1} in 1981	
	Yolo Bypass	0.80 ± 0.06 mm day^{-1}	1998	Sommer et al. 2001
		0.55 ± 0.06 mm day^{-1}	1999	
San Francisco Bay	Brackish	1.01 mm day^{-1}	1980 (two weeks)	Kjelson et al. 1982
San Francisco Bay	Estuary exit	0.48 ± 0.10 mm day^{-1}	2000 and 2001	Woodson et al. 2013
		0.37 ± 0.06 mm day^{-1}	2005	
Gulf of Farallones	Ocean entry	0.85 ± 0.13 mm day^{-1}	1998 El Nino	Macfarlane et al. 2005
		0.55 ± 0.06 mm day^{-1}	1999 La Nina	

Sacramento-San Joaquin Delta
Regional Ecosystem Restoration Implementation Plan

Semi-Final Species Life History Conceptual Model

Sacramento Splittail
Pogonichthys macrolepidotus

Prepared by: Daniel Kratville, Department of Fish and Game, **dkratville@dfg.ca.gov**

Date of Model: September 3, 2008

Status of Peer Review: Model has not yet fully completed the peer review and collegial review process and final modifications may be required of the developers. Model may not be cited or circulated until that process is complete. It may be used in identifying and evaluating restoration actions with assistance from content experts. Model is appropriate for use by experienced evaluation team with input from content experts as necessary.

DO NOT CITE

For further inquiries on the DRERIP conceptual models, please contact Brad Burkholder at BBURKHOLDER@dfg.ca.gov, or Alison Willy at Alison_Willy@fws.gov.

PREFACE

This Conceptual Model is part of a suite of conceptual models which collectively articulate the current scientific understanding of important aspects of the Sacramento-San Joaquin River Delta ecosystem. The conceptual models are designed to aid in the identification and evaluation of ecosystem restoration actions in the Delta. These models are designed to structure scientific information such that it can be used to inform sound public policy.

The Delta Conceptual Models include both ecosystem element models (including process, habitat, and stressor models) and species life history models. The models were prepared by teams of experts using common guidance documents developed to promote consistency in the format and terminology of the models http://www.delta.dfg.ca.gov/erpdeltaplan/science_process.asp .

The Delta Conceptual Models are qualitative models which describe current understanding of how the system works. They are designed and intended to be used by experts to identify and evaluate potential restoration actions. They are not quantitative, numeric computer models that can be "run" to determine the effects of actions. Rather they are designed to facilitate informed discussions regarding expected outcomes resulting from restoration actions and the scientific basis for those expectations. The structure of many of the Delta Conceptual Models can serve as the basis for future development of quantitative models.

Each of the Delta Conceptual Models has been, or is currently being subject to a rigorous scientific peer review process. The peer review status of each model is indicated on the title page of the model.

The Delta Conceptual models will be updated and refined over time as new information is developed, and/or as the models are used and the need for further refinements or clarifications are identified.

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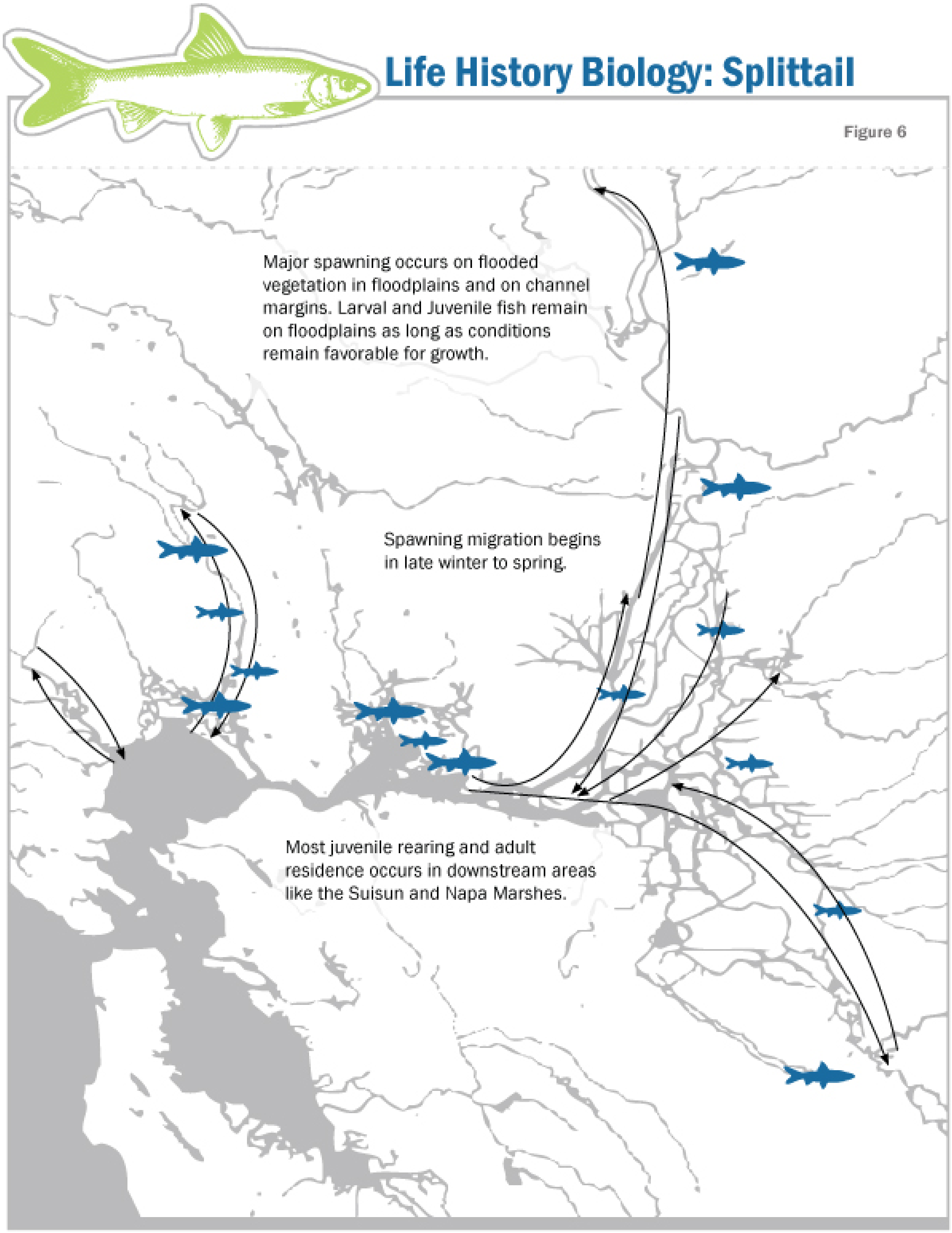
Introduction

The purpose of this conceptual model is to aid management decisions for the continued persistence and recovery of Sacramento splittail. This model, along with the suite of models developed through ERP as well as other information and tools, is intended to be used for evaluating proposed restoration actions for the Delta and throughout the species geographic range. It is a qualitative life-cycle conceptual model that attempts to bring together many sources of information to build a comprehensive document that represents the current state of knowledge for the species. It is not a quantitative model however. It will not provide quantitative limits on species take or population number outcomes. It will only provide an indication of anticipated population response (i.e. increase or decrease with a magnitude of x or y) based on the most current scientific knowledge. With this knowledge, proposed restoration actions can be grouped and a positive population response can be developed through an adaptive management framework. The model must be read and understood in its entirety and does not provide users/readers quick solutions or answers. The entire life-cycle of the animal must be understood by decision makers if they intend to make sound choices in the future. The conceptual model diagram and associated tables can be used to focus the reader on potential species and population limiting factors given the current scientific understanding under current conditions. The model has a specific emphasis on the Delta; however the entire species range must be considered when population level responses are of concern. For this reason the geographic scope of this document is beyond the legal boundaries of the "Delta". This is not intended to be the final version of this conceptual model. As new information becomes available through research and in the scientific literature, this model will be updated as time allows providing managers with the most current information available. This conceptual model relies heavily on the extensive work done by Peter Moyle and others in their 2004 white paper Biology and Population Dynamics of Sacramento Splittail (*Pogonichthys macrolepidotus*) in the San Francisco Estuary: A Review which was published in the San Francisco Estuary and Watershed Science online journal. Content in the model is taken from that review unless otherwise indicated. These sections will be introduced with a citation to Moyle et al. 2004. This model also takes into account new research on Sacramento splittail which has occurred subsequent to the white paper.

Biology (Figure 1, 3, Table 1)

The Sacramento splittail is a Cyprinid native to California and the only surviving member of its genus. It can live 7-9 years and has a high tolerance to a wide variety of water quality parameters including salinity, temperature, and dissolved oxygen. Adult splittail are found predominantly in the Suisun Marsh but are also found in other brackish water marshes in the San Francisco Estuary as well as the fresher Sacramento-San Joaquin Delta. While in these areas splittail feed on a wide variety of invertebrates and detritus. In the spring when California's Central Valley experiences large amounts of snow melt runoff from the Sierra Nevada Mountains, adult splittail will move onto inundated flood plains in the valley to spawn. After spawning the adult fish move back downstream. The eggs that were laid on submerged vegetation begin to hatch in a few days and the larval fish grow at an accelerated rate in the warm and food rich environment. Once they have grown a few centimeters these juvenile fish begin moving off of the floodplain and downstream into similar areas as the adults. In the marsh these juvenile become sexually mature in two to three years (Figure 1 and Figure 3).



Major spawning occurs on flooded vegetation in floodplains and on channel margins. Larval and Juvenile fish remain on floodplains as long as conditions remain favorable for growth.

Spawning migration begins in late winter to spring.

Most juvenile rearing and adult residence occurs in downstream areas like the Suisun and Napa Marshes.

Figure 1 Life History Biology

Table 1. Life stages by biological measures

	Life stage by Biological Measures	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**
	Habitat	**Dates**	**Age**	**Weight**	**Length**	**Acclimation Temperature (Celcisus)**	**Optimum temperatures (Celcius)**	**Maximun Critical Temperature**	**Common Salinity**	**Critical Salinity**	**Min DO Limits**	**Water Velocity**	**pH**	**Turbidity**	**Referen ces**
	(*life stage*)		**(Days post hatch)**		**(mm)**										
1	Floodplain/Channel Margin	Feb - May	-3 to -5	1.55-2.04 mg	1-1.6		UNK	UNK	UNK	UNK	UNK	UNK	UNK	Unknown presumably high tolerence	Moyle et al. 2004
	(*Egg/Embryo*)														
2	Floodplain/Channel Margin	Feb - May	0 - 5 dph		5.5-6.5 mm TL		UNK	UNK	UNK	UNK	UNK	UNK	UNK	Unknown presumably high tolerence	Moyle et al. 2004
	(*Larvae*)		5 - 7 dph		7 - 8 mm TL										
3		Feb - July	Age - 0	0.1-0.5 g	20-30 SL	17	22	30.8	0 ppt	22.4 ppt	9-18 torr Po$_2$ or 0.6-1.3 mg O$_2$/L	3-6 Body Lengths per Second	UNK	Unknown presumably high tolerence	Young and Cech 1996
	Floodplain/Channel Margin/Delta		Age - 0	0.1-0.5 g	20-30 SL	20	24	32.0							
	(Juveniles)		Age - 0	1.0-4.0 g	40-70 SL	17	21	30.0							
			Age - 0	1.0-4.0 g	40-70 SL	20	25	33.0							
4		Year Round	Age - 1	10-42 g	90-130 SL	12		20.5	0 ppt	23.7 ppt		3-6 Body Lengths per Second	Unknown presumably high tolerence to high pH	Unknown presumably high tolerence	Young and Cech 1996, Moyle 2002
	Brackish Marsh		Age - 1	10-42 g	90-130 SL	17	20	28.9							
	(Juveniles)		Immature Age - 2	80 - 200 g	180-230 SL	12		21.9	0 ppt	28.8 ppt			Unknown presumably high tolerence to high pH	Unknown presumably high tolerence	Young and Cech 1996, Moyle 2002
			Immature Age - 2	80 - 200 g	180-230 SL	17	20	29.0	0 ppt	27.4 ppt					
5	Brackish Marsh	Year Round	Mature Age 2 +		200+ SL	NA	Seasonally 5 - 24	29 - 33	2 - 18 ppt	29 ppt	< 1 mg O$_2$/L		Unknown presumably high tolerence to high pH	Unknown presumably high tolerence	Moyle et al. 2004, Moyle 2002
	(*Adult*)														
6	River	Late Nov. - Late Jan.	Age 3 - 10		250 - 400 SL	NA	Seasonally 5 - 24	29 - 33	0 ppt	29 ppt	< 1 mg O$_2$/L		Unknown presumably high tolerence to high pH	Unknown presumably high tolerence	Moyle et al. 2004, Moyle 2002
	(*pre-spawnAdult*)														
7	Floodpalin/River	Feb. - Apr.	Age 3 - 10		250 - 400 SL	NA	Seasonally 5 - 24	29 - 33	0 ppt	29 ppt	< 1 mg O$_2$/L		Unknown presumably high tolerence to high pH	Unknown presumably high tolerence	Moyle et al. 2004, Moyle 2002
	(*Spawner*)														
8	River/Brackish Marsh	Feb. - May	Age 3 - 10		250 - 400 SL	NA	Seasonally 5 - 24	29 - 33	2 - 18 ppt	29 ppt	< 1 mg O$_2$/L		Unknown presumably high tolerence to high pH	Unknown presumably high tolerence	Moyle et al. 2004, Moyle 2002
	(*Post-spawn*)														

Distribution (see Figure 2)

The Sacramento splittail is endemic to the San Francisco Estuary and its associated watershed. Its upper known extension is to the Mud Slough on the San Joaquin at river kilometer (rkm) 201 (an additional 17rkm into Mud Slough) and to rkm 391 on the Sacramento where the Red Bluff Diversion dam exists (Feyrer et al. 2005). There are also genetically distinct populations inhabiting the Napa and Petaluma Marshes and their respective rivers (Baerwald et al. 2005).

Within the Sacramento River tributaries splittail have been documented to the following locations:

- American River to rkm 19
- Feather River to rkm 94 and from just below the Thermolito outlet (95 rkm) (Bruce Oppenheim 2003 pers comm. to Randy Baxter)
- Butte Creek/ Sutter Bypass – to vicinity of Colusa State Park

Within the San Joaquin tributaries splittail have been documented to the following locations:

- Cosumnes River – just above the confluence with the Mokelumne River (Crain et al. 2004).
- Mokelumne River – observed above Woodbridge Diversion Dam to rkm 96.

- Stanislaus River – no confirmed sightings, but based on observations from other tributaries, splittail probably inhabit low gradient portions of the lower river.
- Toulumne River – rkm 27.4 (Legion Park, Modesto, Tim Ford pers comm. to Randy Baxter) and several annually at rkm 8 1999-2002 (Tim Heyne, pers comm. to Randy Baxter).
- Merced River – rkm 20.9 several annually 1999-2001 (1.6 km upstream of Hagaman Park, Tim Heyne pers. comm. to Randy Baxter).
- A second, less studies population exist in the Napa and Petaluma Rivers and marshes (Sommer et al. 2007, Moyle et al.2004):
- Napa River – River Mile 32
- Petaluma River – River Mile 28

Splittail have extended their range in to south and central San Francisco Bay when high freshwater outflow from the Delta allow them to utilize the edges of the bay to move in water with salinities below that of sea water and have been found in Coyote Creek (Moyle et al. 2004). Historically when the Tulare Lake basin was connected to the San Joaquin River during exceptionally high outflow years, splittail were able to access this extremely productive habitat. Splittail bones have been found in Native American middens on the edges of the now extinct lake (Gobalet et al. 1993, 2004 Moyle 2002, Moyle et al. 2004).

As summarized in Moyle et al. 2004:

> In a CDFG angler survey conducted at Garcia Bend (Sacramento rkm 80.5) and upstream, 1202 adult splittail were caught during 1991-1994 and 1999-2000; 94% of these were collected during the January-March migration period (R. Baxter, unpublished data). In 1998-99, four migration pulses were indicated by peaks in catch rate in an experimental fishery at Meader's Beach (Sacramento rkm 39) that occurred in mid-December, late January, and early February, and late February (Garman and Baxter 1999). …
>
> Evidence of splittail spawning on floodplains has been found for both the San Joaquin and Sacramento rivers. In the San Joaquin drainage, spawning has apparently taken place in wet years in the region where the San Joaquin is joined by the Tuolumne and Merced rivers (T. Ford, pers. comm.; F. Ligon, pers. comm.[Sommer et al. 2007]). Larvae and small juveniles have been found in Mud and Salt sloughs within Kesterson and San Luis national wildlife refuges (USFWS, unpublished data). Presumably, spawning took place in the flooded grasslands surrounding these sloughs. Spawning has also been documented on flooded areas along the lower Cosumnes River (Crain and others 2004). Spawning may take place elsewhere in the Delta (e.g., on mid-channel islands) but it has not been documented. In the Sacramento drainage, the most important spawning areas appear to be the Yolo and Sutter bypasses, which are extensively flooded during wet years (Sommer and others 1997, 2001a [Feyrer et al. 2006]). However, some spawning takes place almost every year along the river edges and backwaters created by small increases in flow. Based on larval and beach seine sampling, splittail spawn in the Colusa to Knights Landing region of the Sacramento River in most years (Baxter 1999a; R. Baxter, unpublished data). Occasionally spawning can occur as far upstream as Hamilton City, as evidenced by sporadic collection of adult and YOY fish at a screw trap near the Glenn- Colusa Irrigation District intake (rkm 331). They apparently spawn in riparian vegetation adjacent to flooded rice fields in the lower 12 km of Sutter Bypass and in Butte Slough, based on the presence of numerous early-stage larvae during 1996, 1998, and 1999 (Baxter and Garman 1999; R. Baxter, unpublished data). Splittail may also spawn in the lower reaches of the American River when parts of the American River Parkway flood (R. Baxter, unpublished data). In the

> eastern Delta, the floodplain along the lower Cosumnes River appears to be most important as spawning habitat. …
>
> Likewise, in state and federal fish salvage facilities in the south Delta, adults are captured most frequently in January through April when they are presumably engaged in migration to and from the spawning areas. …
>
> In 2000, they were present in permanent sloughs adjacent to the Cosumnes River floodplain for only about two weeks after leaving the floodplain and were present in large numbers at the mouth on the Mokelumne River about 2 weeks later (P. Moyle and USFWS, unpublished data). This pattern has been seen elsewhere in the system. For example, large numbers of YOY splittail are typically captured in screw traps (set at the base of floodplains) in the Sutter and Yolo bypasses in May, with diminishing numbers in June (Sommer and others 2004; CDFG unpublished data). YOY splittail are typically captured in large numbers at the SWP and CVP fish salvage operations in the south Delta in late May through mid-July, suggesting a seasonal downstream movement. By June and July, YOY splittail are present in marshes along Suisun Bay and in Suisun Marsh (Daniels and Moyle 1983; P. Moyle, unpublished data; C. Kitting, unpublished data). The downstream dispersal of YOY splittail is now well documented and particularly evident after a wet spring. A less well studied aspect of splittail life history is the small fraction of YOY spawned in the Sacramento River and Butte Creek that remain upstream through their first growing season or first year (Baxter 1999a). Age-1 splittail have been captured moving down the Sutter Bypass in spring after rearing in Butte Creek or the Sacramento River (Baxter 1999a; CDFG, unpublished data). Additional YOY have been collected in the Sacramento River beach seine survey in fall and winter (Baxter 1999a; USFWS, unpublished data). There is little evidence for riverine rearing in the San Joaquin River. …

While the preceding paragraphs indicate known collection and distribution of Sacramento splittail, it is assumed that these fish are distributed much more widely in small creeks and marshes throughout the lower portions of the watershed. Splittail are a very dynamic fish that can take advantage of myriad opportunities where habitat is concerned. Having no known collection in an area that splittail could physically access does not mean they are not there. Based on their known distribution I would assume they occur in most waterways in the Sacramento-San Joaquin watershed below dams that are still hydraulically connected to the Estuary.

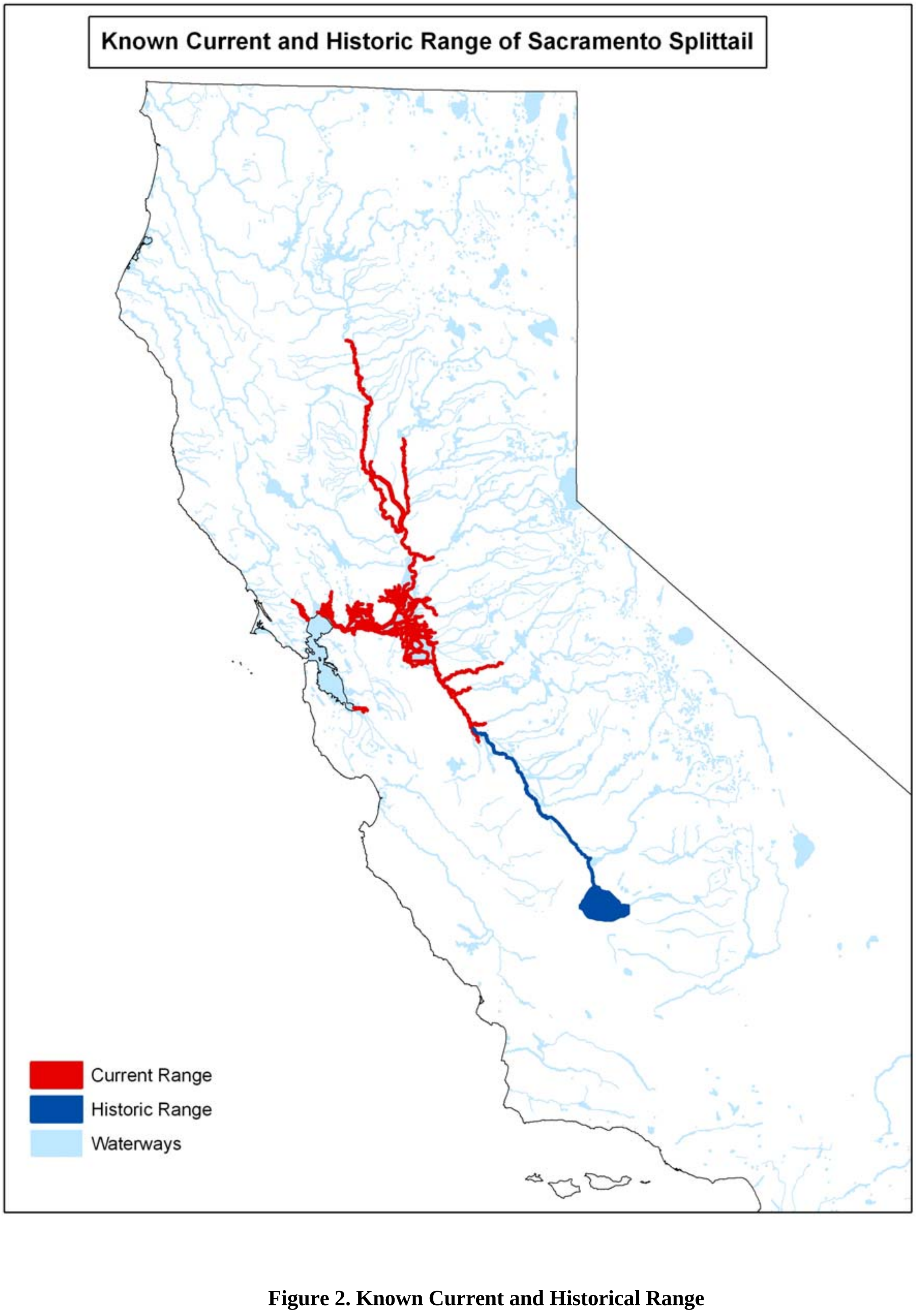


Figure 2. Known Current and Historical Range

Ecology (see *Figures 3, 4*)

Figure 3, found in this section, is a conceptual diagram meant to illustrate the dynamic life history and some of the environmental variables of splittail ecology. The left axis shows location of the various life stages and an estimated salinity found there. The top axis shows a range of temperatures which are illustrated in the figure with red shaded boxes. Temperature ranges are found within the boxes and the life history types pass through these to show how splittail interact with temperature. The dashed blue line arbitrarily delineates water year types. The Department of Water Resources classifies water years as Wet, Above Normal, Below Normal, Dry, and Critical in describing amount of precipitation for that water year. Sacramento splittail have different spawning strategies based on the amount of flood plain inundation in the watershed which is a response to the amount of precipitation and associated flood plain activation. Above the line I assume flood plain inundation and below the line I assume none. This is an arbitrary distinction. Water year type alone does not indicate the amount of flood plain inundation. The line is intended to show the distinction in spawning strategies associated with amounts of precipitation and flood plain activation. On the bottom axis there is a sliding date line. This line is not set for splittail migration, they do not posses calendars. The line is meant to give a general idea of when migration and spawning is occurring. The migration is timed more with changing photo period, increased water flows and lowering of temperatures.

It is likely that splittail are responsible for large transfers of energy from upstream floodplains into Suisun Marsh and Bay areas. These large migrations, both upstream adults and downstream juveniles are probably important in the seasonal transfer of energy within the estuary's foodweb (Feyrer et al. 2007). While this is unquantified, the sheer numbers of juvenile splittail that are found following large inundation events would indicate a significant positive impact to the downstream foodweb. It is also possible that splittail are responsible for the movement of contaminants around the system, both Selenium upstream into the Sacramento and Mercury downstream into the estuary during these migrations.

Sacramento splittail are secondary consumers throughout their life cycle and are benthic foragers most active during the day (Caywood 1974, Goals Project 2000). In Suisun Marsh adult splittail gut contents are predominantly detritus (60-79%), however, a shift has occurred since the invasion of Corbula amurensis in 1986 (Feyrer et al. 2003). After the invasion and establishment of the clam, mysid shrimp (Neomysis mercedis) populations collapsed (Kimmerer and Orsi 1996). Mysid shrimp which originally made up to 24% (average dietary importance) of splittail gut contents was reduced to 2% in the post clam period (Feyrer et al. 2003). The amount of detritus remained similar at 70% with the difference being made up with other invertebrates and Corbula (6%) itself (Feyrer et al. 2003). This shift in diet may expose splittail to higher levels of Selenium which has been implicated in reproductive stress in fishes in general and splittail (Feyrer et al. 2003, Teh et al. 2002, 2004). The nutritional importance of detritus in splittail diets is unknown. The fact that it makes up such a large portion of the gut contents indicates that it must have some nutritional value (Moyle et al. 2004). As adult splittail begin migration and movment onto inundated flood plains and channel margins, terrrestrial worms and insects become important food items (Moyle et al. 2004). Historically when splittail were more common in areas where Chinook salmon spawning was occurring splittail have been observed feeding on loose salmon eggs (Moyle et al. 2004). After yolk sac absorption the larvae begin feeding on small rotifers (Bailey 1994). Prey composition shifts as they increase in size to cladocerans and chironomid larvae (Kurth and Nobriga 2001). Larval splittail to 15mm feed heavily on zooplankton, primarily made up of cladocerans. Chironomid larvae begin to dominate after 15mm in length has been achieved (Feyrer et al. 2007). Splittail 50-100mm SL also feed primarily on detritus with calanoid and harpactocoid copepods being important as well (Moyle et al. 2004). On the Cosumnes River floodplain, the early larval period seems to coincide with large blooms of zooplankton, providing an abundant food supply (Crain et al. 2004). Splittail that rear on flood plain habitats as opposed to river margin habitats showed greater length and condition factor in the Cosumnes system (Ribeiro et al. 2004). On the Yolo bypass, seasonal inundation is accompanied by a large hatch of an endemic chironomid, Hydrobaenus saetheri Cranston and is more abundant on previously dry areas of the floodplain (Cranston et al. 2007, Benigno and Sommer 2008). This is probably an important food source for larval and juvenile splittail spawned on floodplains in the Central Valley. Age-0 splittail are strongly opportunistic feeders (Feyrer et al. 2007). Their diet changes based on habitat types occupied through ontogeny (Feyrer et al. 2007).

Larval and small juvenile splittail in flooded areas are preyed upon by an array of invertebrate predators, as well as by juveniles of both native and alien fishes that invade the areas during flood events (Moyle et al. 2004). If larval mortality rates are similar to those of other fishes, then it is likely that the vast majority of splittail die in their first few weeks of life at rates that are independent of densities of larvae but, dependent on densities of predators (Moyle et al. 2004). Stochastic environmental factors, such as sudden drops in water level that strand embryos and larvae are also important (Moyle et al. 2004). Water level may also affect predator density by expanding or contracting inundated habitat: expanded habitat should reduce predator (e.g. birds) density directly (Moyle et al. 2004). In addition, most adult alien predatory fishes such as largemouth bass (Micropterus salmoides) and bluegill (Lepomis macrochirus) seldom venture far from permanent water ways onto the flood plain (Moyle et al. 2004).

> Adult and juvenile splittail are preyed upon by piscivorous fishes and birds. Although splittail are uncommon in striped bass diets, their effectiveness as bait for striped bass has long been recognized by anglers, who fish for splittail to use them for bait. Presumably centrarchid basses, sunfish, and crappies (Pomoxis spp.) are important predators on juveniles as they leave floodplain rearing areas. Juvenile pikeminnow (Ptychocheilus grandis) and Chinook salmon are common on the floodplain and may prey on larvae and small juveniles, but this has yet to be documented. Bird predation appears limited until water recedes and floodplains begin to isolate from main channels at which point fish are exposed to wading birds. [Moyle 2004]

Hydrology is a major driver of splittail populations. Large scale spawning occurs only in years with significant inundation of flood plains in the Sacramento - San Joaquin watershed. While some small amount of spawning occurs in perennial marshes (Moyle et al. 2004) this is not enough to sustain the population in the face of its many stressors as indicated by the need to list the species during the extended drought of the 90’s and associated water management (Feyrer et al 2006). Splittail are considered to be obligate flood plain spawners (Moyle 2002). Seasonal inundation of areas with a mosaic of low velocity off channel habitat types would be beneficial to this and likely other native species. They must have seasonally flooded lands on which to spawn and for early rearing of larval and juvenile fish. The ability of California water managers to manipulate flows in this system makes the dynamics of flood plain inundation extremely important to policy makers. Splittail need water levels and inundation duration in ranges that were historically present (30 - 90 days). The risk of stranding eggs and larvae on the flood plain could result in the complete loss of any benefits accrued with the initial inundation. Proper connectivity (i.e. maintaining sufficient flows to allow time for ontogenetic movement downstream of juvenile fish) of the flooded area with the main channel of the river is as important as initial flood plain inundation. This requires sufficient and constant flows onto and off of the inundated areas for a length of time required for initial rearing to occur. The minimum length of inundation is required to achieve strong year classes when associated with large scale flood plain inundation as occurs on the Yolo Bypass. Longer inundation periods allow for extended and multiple spawning events as well as other food web associated benefits. Flood plains also do not allow for residency of non-native predators. These types of habitats dry in the early summer in California (unlike the Midwest and Eastern United States where hydrology is very different) and non-native fish are unable to utilize floodplain in California in the same way as their home ranges. Managers have a unique opportunity to improve populations of a native fish with a relatively simple management option, inundation of flood plains throughout the system.

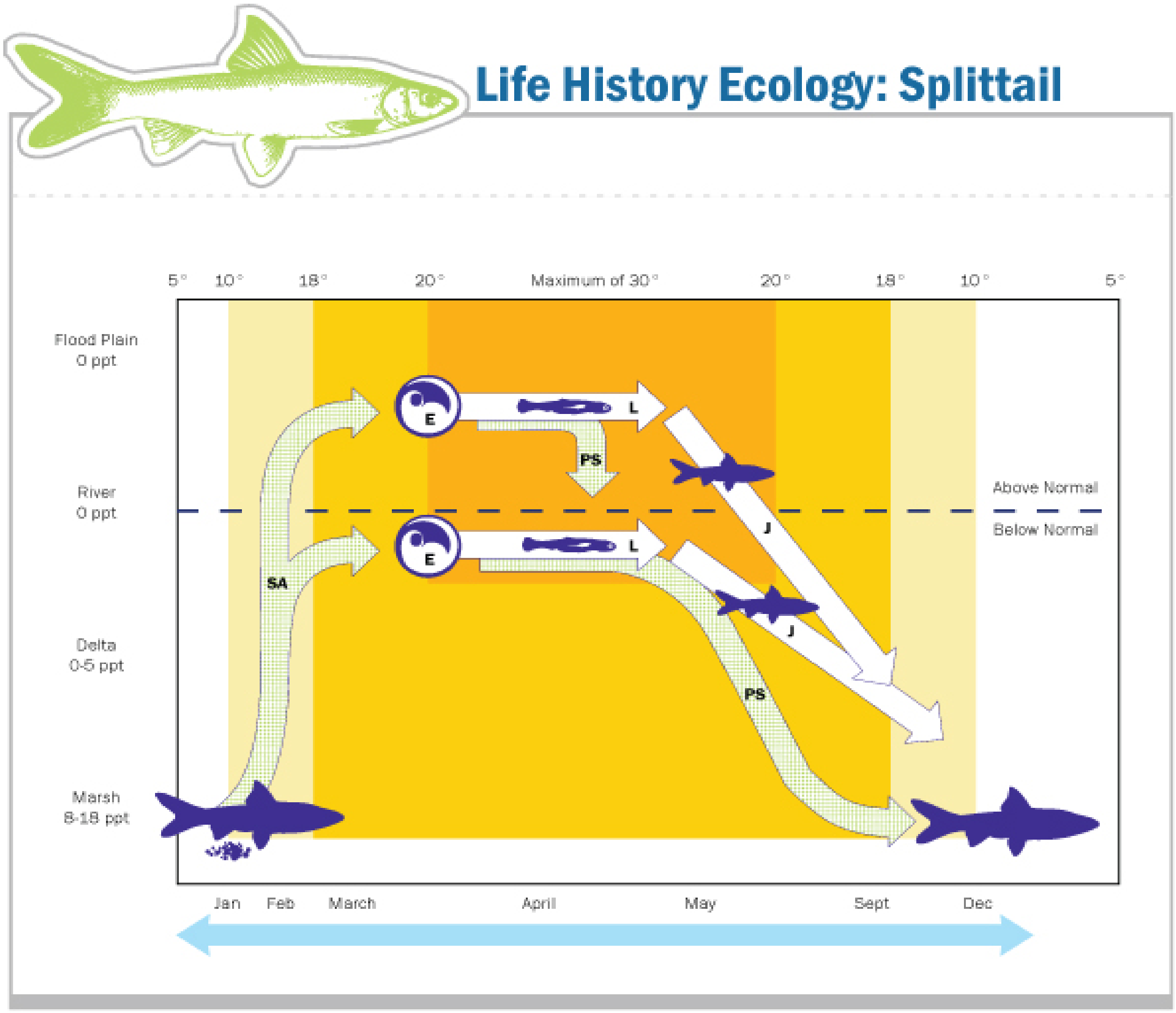


Figure 3. Life History Ecology Diagram Life history ecology diagram showing a hypothetical conceptual life history model for Sacramento splittail. Spawning Adults (SA), Post-Spawning Adults (PS), Eggs (E), Larvae (L), Juveniles (J). The blue dashed line indicates a change in spawning strategy that accompanies flooding associated with water year types. Adult fish are shaded in green.

An accurate abundance estimate of the splittail population does not exist. There are several sampling efforts that used to give an idea of the population size but, because they are not all designed to measure splittail abundance directly the results must be interpreted with caution. The gear used in some of these efforts is not intended for capture of splittail, a predominantly benthic fish found in association with marsh vegetation. Sommer et al. 2007 describes many of these efforts and gives some direction. The various sampling efforts indicate that splittail abundance varies widely for age-0 fish between years and water year types. In wet years (with large amounts of floodplain inundation) the abundance of age-0 fish can be extraordinarily high and in dry years worrying low. For adult fish (age-2+) the trends are different. The adult fish show much less variation between years, something expected from such a long lived fish. This is probably an evolutionary response to environmental stochasticity. These adult fish do show an increase in abundance two to three years after a strong year class of age-0 fish.

Adult Spawning migration **(see Figure 7)**

Spawning takes place on inundated floodplains (especially the Yolo Bypass, possibly the most important flood plain for splittail in the Sacramento system)) and riparian areas of rivers in submerged terrestrial vegetation (Moyle 2002). Sacramento splittail may be the most floodplain-dependent fish in the San Francisco estuary (Sommer et al. 2007). Adults begin a gradual upstream migration towards spawning areas sometime between late November and late January, but this is most evident in February thru April during high spring flow events (Moyle et al. 2004). Feyrer et al. (2006) showed that upstream migration and spawning activity are cued by changing photo period associated with the vernal equinox and or increases in flows associated with spring runoff. For cyprinids gonadal development is partially controlled by photo period (Feyrer 2006). This inundation of floodplains and riparian edges of areas upstream of Suisun Mash in the Sacramento-San Joaquin system provide feeding and spawning habitat (Moyle et al. 2004, Feyrer et. al 2006). Feeding in these areas by adults is probably important for both pre and post spawning success and survival. This migration may also occur in the Napa and Petaluma Rivers but is not monitored under the current programs. Early migrants tend to be larger fish (median FL = 293.5 mm) with later migrants being smaller (median FL = 273mm) (Moyle et al. 2004). This gives support to the hypothesis that older larger fish migrate upstream before younger smaller individuals. Mature, ripe splittail have been found in association with high turbidity, temperature below 15°C, and flooded terrestrial vegetation (Moyle et al. 2004). In late February and early March splittail begin moving onto inundated areas of the Cosumnes River floodplain. Fish stay through February and/or March to spawn on the submerged vegetation while sufficient depths and temperatures (<20°C) are maintained depending on the hydrograph (Crain et al. 2004). Flooding early in the season is not always required for adult fish to successfully spawn on inundated floodplain habitats. In 2003 the Cosumnes River floodplain was not inundated until April. Splittail were still able to move upstream to these flooded areas and spawn successfully with YOY collected in May (Moyle et al. 2004). According to Wang (1986, 1995) spawning can occur from late February to early July based on collection of larval fish. Spawning after early May is very unlikely, due to water levels and temperatures in the system. Their eggs are adhesive and stick to submerged vegetation and/or detritus until hatching (Moyle 2002, Moyle et al. 2004). When ripe splittail are ready to spawn, the males color darkens, fins develop a red tinge on their edges, and they develop white tubercles at the base of their fins and heads (Moyle 2002). In Suisun Marsh, splittail held in pens were gravid when salinities hit 1.2‰ and temperatures of 15°C (Bailey et al. 2000). Males stopped producing sperm after one month while females remained ready to spawn (Bailey et al. 2000). Bailey et al. (2000) also found that females held in indoor tanks would not mature when temperatures were held constant at temperatures of 18°C even with injections of hormones. This suggests that reproduction is partially controlled by a change in temperatures. Splittail spawning behavior has only been observed in captivity (Moyle et al 2004). Spawning in captivity occurred subsequent to routine cleaning of the holding tank that was accompanied by a drop in water level (Moyle et al. 2004). One or more males will swim along side and slightly behind a female and attempt to fertilize the released eggs as she swims through and above submerged vegetation. Channel margin riparian habitat maintains the population in dry years by providing small localized areas of inundated vegetation for spawning (Moyle et al. 2004, Feyrer et al. 2005). When floodplain inundation does not occur in the Yolo or Sutter bypasses, adult splittail must migrate farther upstream to find suitable habitat along channel margins or flood terraces that are inundated even in lower water year types; although spawning in such locations occurs in all water year types (Feyrer et al. 2005). Maintaining and increasing this type of habitat throughout the species range will help maintain genetic diversity during prolonged drought events and avoid a genetic “bottleneck”.

Early life stages - Egg/embryo **(see Figure 5)**

Fecundity for splittail varies considerably between studies which have been conducted over the past 30+ years (Moyle et al. 2004). For wild fish they range from 165 ova per mm SL with 100,800 per female, 600 ova per mm SL with 17,000 to 266,000, and 261 ova per mm SL with 150,000 per female (Moyle et al. 2004). Fish held in the laboratory for an extended period of time, showed fecundities that ranged from 24,753 to 72,314 and an average of 12.3% of body weight being egg mass (Moyle et al. 2004). This variability could be due to a wide range of environmental variables but is most likely from food variability as illustrated by the bioenergetics equation found in Moyle and Cech 2003. A recent concern for splittail is the partial shift in diet from mysid shrimp to bivalve prey and the possible bioaccumulation of selenium which is a known reproductive contaminant (Moyle et al. 2004). The most recent fecundity relationship equations are F = 0.0004 (SLmm)3.40 and F = 107.39(TWg)1.06 (Moyle et al. 2004).

This is summarized well in Moyle et al. 2004 and is reproduced below:

> Early Life History: Splittail eggs are 1.0-1.6 mm in diameter with a smooth transparent chorion (Wang 1986; Feyrer and Baxter 1998). Bailey and others (2000) found that eggs weighed an average of 1.55-2.04 mg wet weight and had an average diameter of 1.38 mm. The eggs are demersal and adhesive (Wang 1986; Bailey 1994), attaching to submerged vegetation or any other submerged substrate. At 18.5 °C they start to hatch within 3-5 days after spawning (Bailey 1994). Eggs laid in clumps hatch more quickly than individual eggs. Larvae are 5.5-6.5 mm total length (TL) when they hatch, have a yolk sac, a non-functional mouth and no eye pigment (Wang 1986, 1995; Bailey and others 2000). At 5-7 d post-hatch, they reach 7-8 mm TL, the yolk is absorbed, and feeding begins…
>
> They reach 10-11 mm in 15 days post-hatch under laboratory conditions (Bailey and others 2000). By the time they are 13-16 mm TL, they are recognizable as juveniles, with a swim bladder (Wang 1995). …

Larval downstream and Juvenile Stage **(see Figure 5)**

This is summarized well in Moyle et al. 2004 and is reproduced below:

> By the time they are 20-25 mm TL, they are easily recognizable as splittail and capable of fairly active swimming. Observations on small-scale floodplain wetlands indicate that the splittail are strongly associated with shallow edge habitat at a size of 20 mm, but gradually begin to use a variety of offshore habitats by 29 mm (Sommer and others 2002). These early life history stages also appear to show strong diel differences in behavior; at night, many young become completely benthic. They stay on the floodplain to feed and grow as long as conditions are suitable (i.e., cool, flowing water is present). On the Cosumnes River floodplain in 1998, a year in which flooding persisted well into June, juvenile splittail were common into May (K. Whitener, unpublished data). In 2000, most left abruptly over a short period in early May, when the floodplain was briefly reconnected with the river during two flow pulses produced by late rainstorms. Prior to the pulses, water had ceased flowing on to the floodplain and water temperatures had been steadily climbing. On the Cosumnes River juveniles have been observed leaving the floodplain at a size of 25-40 mm TL, when they dispersed rapidly downstream (P. Moyle, unpublished data).

Although some are swept off floodplains and downstream by flood currents (Baxter et al. 1996), many splittail larvae and juveniles remain in riparian or annual vegetation along shallow edges on floodplains as long as water temperatures remain cool (Sommer et al. 2002, Moyle et al. 2004). By about 29 mm splittail move to deeper habitats (Sommer et al. 2002). Warming water temperatures may be a cue to exit floodplains and begin downstream migration. Stranding of splittail in perennial ponds on the Yolo Bypass does not appear to be a problem (Feyrer et al. 2004). Such migrations often occur in late-April, May or even June of high flow years (Moyle et al. 2004). When these fish have reached 30-40 mm migration begins indicating a possible ontogenetic effect (Feyrer et al. 2006). These age 0 fish are 6.5 times more likely to be found in back water sloughs upstream and 3.5 times more likely to be found in inter-tidal habitats downstream than in main channel areas (Feyrer et al. 2005). Two early life history strategies occur in the Sacramento River system: the dominant one is characterized by juveniles migrating downstream in late spring and early summer to the Delta and Suisun Bay and Marsh; a less well studied strategy is to remain upstream through the summer into the next fall or spring and then migrate downstream (Baxter 1999a, Moyle et al. 2004). This latter strategy occurs in Butte Creek and the main stem Sacramento River. As the water recedes further, juveniles remaining in upstream riverine habitats congregate in large eddies for feeding (Randy Baxter unpublished data). The exact cost-benefit ratio of this particular life history strategy is unknown.

Juvenile stage to Adult (Figure 6)

Juveniles are most abundant in shallow (<2m), turbid water with a current, and are often found in small narrow sloughs lined with tules and other emergent plants (Moyle et al. 2004). Adults enter the same types of habitats and have been observed to move into shallow water < 1m with incoming tide to feed (Moyle et al 2004). Non-reproductive splittail are abundant in moderately shallow (< 4m) brackish and freshwater tidal sloughs and shoals such as found in Suisun Marsh and the margins of the lower Sacramento River (Feyrer et al. 2005; Moyle et al. 2004). Most late stage juveniles and adult splittail inhabit tidal fresh and brackish water in the Delta, Suisun Bay, Suisun Marsh, as well as the lower portions of the Napa and Petaluma rivers and their marshes (Moyle et al. 2004). Individuals of all sizes can be found in very shallow water (≤ 0.5m) foraging; particularly on a flooding tide (Randy Baxter unpublished data).

This is summarized well in Moyle et al. 2004 and is reproduced below. Figure 4 is taken from Moyle et al. 2004 (Figure 11 in original) to aid the reader of this document:

> Splittail, like other Central Valley cyprinids are relatively long-lived and reach fairly large sizes (for North American cyprinids). Analysis of scales indicates life spans of 5-7 years (Daniels and Moyle 1983) but analysis of other hard parts indicates that the largest fish may be 8-10 years old (L. Grimaldo, CDWR, unpublished data; R. Baxter, unpublished data). Both sexes reach about 110-120 mm SL in their first year, 140-160 mm in the second year, and 200- 215 mm SL in their third year, growing about 25-35 mm per year thereafter. They may reach over 400 mm SL but fish over 300 mm SL are uncommon. The largest and oldest fish are females. Growth rates, especially in the first year or two of life, may be strongly dependent on availability of high quality food, as suggested by changes in growth rate following the invasion of the overbite clam into the marsh in the 1980s. This invasion was followed by the collapse of Neomysis populations upon which splittail historically specialized (Feyrer and others 2003). When growth rates of three strong cohorts of immature fish (to year 2) in Suisun Marsh are visually compared, the growth rate of the 1980 cohort appears to be greater than that of the 1995 cohort, with the 1986 cohort showing an intermediate growth rate (Figure 11). Because splittail in Suisun Marsh grow very slowly during the cool months of October-March (e.g., change in YOY SL = 10 mm), data for these months were pooled and used in preliminary analyses comparing splittail lengths from

1979-1986 ("pre-clam") with those from 1986-1999 ("post-clam"). Pre-clam YOY (n = 2113) were significantly larger than the post-clam YOY (n = 906) and pre-clam 1+ (n = 1105) were significantly larger than the post-clam 1+ (n = 267) (T-test, 2 tailed; S. Matern and P. Moyle, unpublished data).

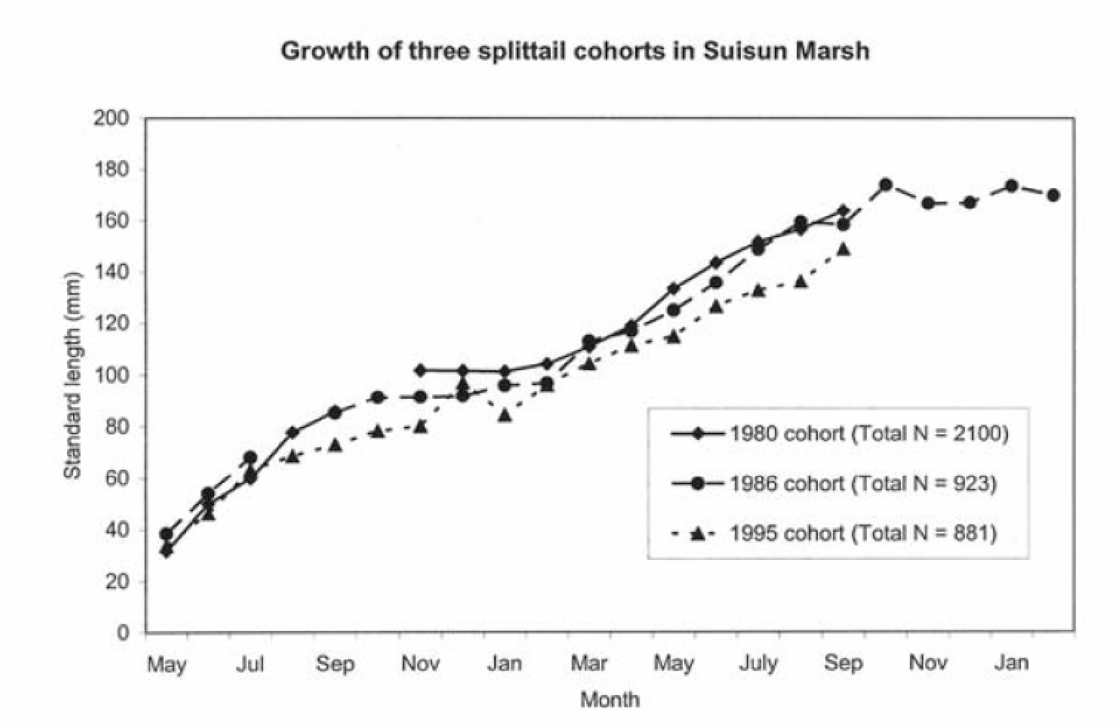


Figure 4. Growth of three cohorts of splittail for the first 22 months in Suisun Marsh, based on mean lengths from monthly samples from 1980,1985 and 1995. S Matern and P. Moyle unpublished data (Figure 11 in Moyle et al. 2004).

Stressors by life-history stage (Limiting Factors)

Modification of floodplain habitat and riverine edge/flood terrace (G): Splittail juveniles are believed to use increasing water temperatures to queue emigration; however, artificially constructed channels are often too deep to warm rapidly and splittail juveniles may get stranded by receding water. The substantial loss of floodplain from conversion to agriculture and urban areas and loss of river edge spawning habitat is probably the key limiting factor for splittail populations (Moyle et al. 2004). Splittail populations show a remarkable response to floodplain inundation as indicted by salvage during wet years following extensive floodplain inundation (Moyle et al. 2004).

Loss of riverine spawning and rearing/migration habitat (G): In the 1960s and 1970s the Army Corps of Engineers cleared and rip-rapped levees along the lower Sacramento River reducing or eliminating spawning habitat from the city of Sacramento downstream. Currently efforts are under way to improve flood protection for communities along much of the lower Sacramento River and several other valley rivers. Actions being proposed and conducted include removal of trees and riparian vegetation and armoring with rip-rap. The extensive application of riprap to the rivers for flood protection may have reduced juvenile splittail migration areas to the Delta and Suisun Marsh from spawning areas upstream. The Army Corps of Engineers current policy is for removal of all large trees and brush from levees to improve detection of weak points and failures. The Corps is working with local regulatory agencies to try to avoid the removal of all plants from levees here in California. The vegetation on the levees is the last remaining riparian habitat in the Delta and its removal would be catastrophic for numerous species.

Bioaccumulation of contaminants (C): Perhaps the greatest concern relates to Selenium. Tissues of wild caught splittail in Suisun Bay were sufficiently high in Selenium to potentially cause physiological problems, in particular reproductive abnormalities (Stewart et al. 2004). Adult splittail feed on the overbite clam (*Corbula amurensis*), which accumulates and transfers Selenium at high concentrations. With the decline of native mysid shrimp in the estuary, splittail have turned more to benthic foods such as bivalves (Feyrer et al. 2003). Young splittail, which might pick up selenium from dietary items, might also be negatively affected. Teh et al. (2004) found that young splittail fed a diet high in selenium grew significantly slower and had higher liver and muscle selenium concentrations after nine months of testing.

Toxics (C): Kuivila and Moon 2004 documented dissolved pesticides in the Sacramento-San Joaquin Delta during April-June (1998-2000) when young, growing splittail were migrating to the Delta and estuary. Since the early 1990s the use of pyrethroid pesticides has increased substantially in the Central Valley (Oros and Werner 2005). Though relatively non-toxic to mammals, it is highly toxic to aquatic organisms, including fishes. Toxicity typically occurs in the low to fractions of a part per billion (Oros and Werner 2005). Also its use on row crops (including rice) commonly grown in the Yolo and Sutter bypasses and its proclivity to adhere to particles in the water and drop to the sediment provide a dietary pathway to splittail ingestion along with detritus during feeding (see diet section above). Though there is no direct evidence, this might occur during foraging on inundated floodplains or in the estuary after the pesticides entered the water way through field drainage and were transported to and settled in the Delta. Teh et al. (2005) showed sub-lethal effects and delayed mortality to larval splittail exposed to orchard storm water runoff that contained a pyrethroid (esfenvalerate) and an organophosphate (diazinon). These fish showed higher mortality rates and slowed growth even after a three month recovery period. Combinations of low concentration toxic chemicals (Pyrethroids, Organophosphates, Organochlorines, etc.) which may have low effects on fish directly can have significant negative impacts on Chironomids (Lydy and Austin 2004), and other invertebrates (Hunt et al. 1999, Hunt et al. 2003, Amweg et al. 2005, Weston et al. 2008). Chironomids are an important food source for splittail on the floodplain where these chemicals occur. A loss of food resources at this early life stage coupled with sub-lethal toxic effects could have a substantial impact on the population but this is unknown.

Changes in Water Management (B,I): The changes in river stage resulting from diversion to storage have been little studied, but conceivably could, during low and moderate runoff years, affect splittail access to floodplains and their ability to emigrate successfully after spawning and early rearing. Increases in total and in winter water exports in the south Delta have increased salvage of a wide variety of upper estuary fishes since 2000 (Herbold et al. 2008), and may have increased salvage of age-1 and older splittail. However, the majority of splittail salvage is composed of age-0 fish, occurs May-July and occurs during years with high outflows that persist into the March-April spawning period of splittail (Sommer et al. 1997). Splittail salvage increased substantially in both 2005 and 2006, reaching a record high of over 5 million fish at the Tracy Fish Collection Facility. Post salvage survival is unknown.

Introduced (alien) species (D,F): Splittail have persisted in the estuary through numerous invasions of fishes and invertebrates. Some, such as the invasion of *Acanthomysis bowmanii* may have been beneficial as the native, *Neomysis mercedis*, was already on a steep decline. The invasion of the overbite clam (*C. amurensis*), also became a diet item, but may have detrimental bioaccumulation effects (see above). Major introduced fish predators such as striped bass and largemouth bass have been in the Delta for over a century (Dill and Cordone 1997), and splittail have persisted; however, reduced turbidity in the delta combined with increased largemouth bass habitat provided by *Egeria densa* has enhanced both the bass numbers and their ability to sight feed . A major concern is the potential invasion of the Delta by the highly predatory northern pike (*Esox lucius*). The pike, present in Lake Davis on the Feather River, is currently the target of a major eradication effort. If eradication was to fail and the pike escape to the delta, they are likely to become abundant in the same habitats used by splittail (Moyle 2002).

The following three sections are summarized well in Moyle et al. 2004 and are reproduced below (E,F):

> Fishery: One of the least appreciated aspects of the splittail migration is that they are subject to a considerable but poorly documented legal fishery from November through May. Anglers catch splittail using earthworms and cut bait. Most fish caught are kept because they are prized as food fish in Asian cuisine. Incidental data collected during creel surveys for striped bass and salmon (K. Murphy, unpublished data) suggest that at times hundreds of adult fish may be caught on a daily basis. It is possible the fishery could significantly reduce egg supply available for spawning by reducing the number of large females. However, most of the fish caught are relatively small (15-25 cm TL) so may be mostly males (J. Hileman, pers. comm.).
>
> Food Availability: Growth rates, especially in the first year or two of life, may be strongly dependent on availability of high quality food, as suggested by changes in growth rate following the invasion of the overbite clam into the marsh in the 1980s. This invasion was followed by the collapse of Neomysis populations upon which splittail historically specialized (Feyrer and others 2003).
>
> Climate Change: Global warming is occurring (Levitus and others 2000, 2001) and it will have an impact on the estuary. The most severe impacts are likely to be through changes in precipitation patterns and sea level rise. The most likely scenarios give northern California about the same amount of water but most of it comes as rain and much less is stored as snow in the Sierra. Year to year variability in precipitation will also be higher (as we are already seeing). This most likely means continued increase in large floods, increased frequency and severity of droughts, and increased difficulty of providing water for human and environmental needs. At the same time, sea level will keep rising due to melting of polar and glacial ice and thermal expansion of the ocean. A rise of 49 cm (19 in) in the next 100 years is the best estimate available of sea level rise, with a possible range of 20 to 86 cm (Warrick and others 1996). However, processes by which heat is transferred from the atmosphere to the ocean is still being assessed (Levitus and others 2001) and the role of large events, such as the 1997-1998 El Niño event, in dramatically heating the deep ocean are only beginning to be understood. It is possible that the effect of thermal expansion of the ocean is being underestimated. In the estuary, sea level rises will be amplified by tidal incursions into the narrow bays and channels because a greater volume of water will have to be squeezed into a relatively tight fixed space (Fisher and others 1979). This rise will put enormous stress on all leveed systems in the estuary, but especially in the Delta, which is almost entirely below sea level already (and many areas are 5+ m below sea level). The resulting higher tides will likely stress levees in the Delta to widespread failure, turning the Delta into a brackish bay. Suisun Bay and Suisun Marsh will become increasingly saline, resembling San Pablo Bay as it is today. Salinities in the Delta and Suisun Bay, however, will show wide variability in response to increased floods and droughts. Coupled with the stress on levees caused by rising waters is the distinct possibility of levee failure and weakening in the next few years by earthquakes (Torres and others 1999). Moreover, because the position of X2 (the 2 psu isohaline line) is related to net Delta outflow, higher sea level and concomitant higher tides will push X2 further upstream, probably resulting in decreased primary and secondary productivity (Jassby and others 1995). Fortunately, splittail will probably be able to adjust to most of the changes because the historic Central Valley and its estuary, in which they evolved, have had enormous changes through the past million or so years, both in a geologic

sense and in the sense of variability through time periods on the order of 1 to 100 years. During periods of prolonged drought the Delta would have been largely a brackish water system; Suisun Bay would have been rather saline under the same conditions. Thus the migratory behavior of splittail can be viewed as an adaptation to fluctuating conditions. Somewhere in the system there would be both flooded areas for spawning and brackish areas for rearing. Thus, under the changes predicted as the result of global warming, splittail could rear in the Delta and spawn in upstream flooded areas, such as the Sutter Bypass. They would be especially favored if levees along the Sacramento and San Joaquin rivers were set back to increase the amount of floodable land (as a way of increasing storage in flood-control reservoirs and countering the effects of sea level rise). The biggest problem they would face is likely to be the deep (3-6 m) water habitat that would dominate on flooded islands, which would be poor habitat for rearing. Thus their survival may hinge on having available large amounts of shallow tidal areas on the edges of the Delta.

Migration Barriers (H): Access on and off the flood plains are the major migration barriers that splittail face. Connectivity between floodplain and river channels is essential to the success of floodplain spawning. In addition to local access to the floodplain both adult and juvenile splittail need adequate natural edge habitats in the channels between rearing and spawning areas. The last remaining natural edge habitats should be protected and a change in riprap protocols evaluated in order to restore these migration corridors.

Disease and Parasites: When spawning males begin their migration they are often the first to arrive on the floodplains and the last to leave once spawning is over. This fact makes adult males more likely to suffer from disease and or parasites (Moyle et al. 2004) Males seldom live more than 4-5 years and post spawning mortality due to stress from parasites and or disease may be high (Moyle et al. 2004). Spawned out fish are commonly found with anchor worms and when found in the CVP/SWP fish salvage facilities have open sores on their sides (Moyle et al. 2004). These fish are probably on their way back to feeding areas in the Suisun marsh where food sources are more abundant and salinities are higher (Moyle et al. 2004). Salt is commonly used in fresh water aquaculture facilities to treat parasitic skin infections (Klinger and Floyd 2002), and returning to brackish water areas may help these fish overcome these infections and reduce osmotic stress (Moyle and Cech 2003).

Entrainment and Salvage (A): Small agricultural diversions within the planning area are not likely to have a large effect on splittail populations. Water velocities at these pumps are low and fish are able to swim past without being entrained in large numbers (Nobriga et al. 2004, Moyle and Israel 2005, Sommer et al. 2007). Power plants within the planning area have the ability to entrain large numbers of fish. Large volumes of water are pumped through the facilities which are located within splittail rearing habitat (Matica and Nobriga 2005). The State Water Project and the Central Valley Project show high rates of salvage when splittail populations are at high levels; YOY have critical swimming velocities that are near the water velocities of the large pumps and are entrained at these facilities (Young and Cech 1996). (Sommer et al. 1997, Sommer et al. 2007). When these fish are salvaged, mortalities can be quite high from over crowding within transport tanks and predation at drop off points within the Delta (Moyle et al. 2004). Actual mortality rates have not been determined. This would require mark recapture or ultra sonic tagging experiments, which have not been performed on splittail in response to salvage releases. Salvaged adults are returned to an area downstream of the pumps and farther downstream in their spawning migration. This will increase the energy needed for the fish to reach their spawning sites upriver and could reduce their ability to spawn successfully (Moyle et al. 2004).

Table 2. Stressor Description Matrix. Tables 2 and 3 do not attempt to call out all possible stressors, only stressors deemed important from a population perspective.

		A	B	C	D	E	F	G	H	I
	Habitat (*life stage*)	**Entrainment**	**Stranding**	**Toxics**	**Predation**	**Harvest**	**Food Availability**	**Habitat Loss**	**Barriers**	**Operations**
1	Floodplain/Channel Margin (*Egg/Embryo*)		Sudden dewatering	Se, Hg, Pyrethroids, Endocrine Disrupters			Less food low water year mainstem spawning	Loss of floodplains/ riprap of channel banks		Reduced Seasonal Flows
2	Floodplain/Channel Margin (*Larvae*)	SWP/CVP Small Ag	Flood plain Drainage and Connectivity	Se, Hg, Pyrethroids, Endocrine Disrupters			Less food low water year mainstem spawning	Loss of floodplains/ riprap of channel banks	Floodplain-Channel connectivity	Reduced Seasonal Flows
3	Floodplain/Channel Margin/Delta *(Juveniles)*	SWP/CVP Small Ag	Flood plain Drainage and Connectivity	Se, Hg, Pyrethroids, Endocrine Disrupters	Non-native predation	Capture for use as Bait	Less food low water year mainstem spawning	Loss of floodplains/ riprap of channel banks	Floodplain-Channel connectivity	Reduced Seasonal Flows
4	*Brackish Marsh* *(Juveniles)*	Power Plants		Se, Hg, Pyrethroids, Endocrine Disrupters	Non-native predation	Capture for use as Bait	Shift in diet with clam invasion			
5	Brackish Marsh (*Adult*)			Se, Hg, Pyrethroids, Endocrine Disrupters		Subsistence Fisherman	Shift in diet with clam invasion			
6	River (*pre-spawnAdult*)			Se, Hg, Pyrethroids, Endocrine Disrupters		Subsistence Fisherman		Loss of floodplains/riprap of channel banks longer migrations	Floodplain-Channel connectivity	Reduced Seasonal Flows
7	Floodpalin/River (*Spawner*)		Flood plain Drainage and Connectivity	Se, Hg, Pyrethroids, Endocrine Disrupters		Subsistence Fisherman	Shift in diet with clam invasion	Loss of floodplains/riprap of channel banks longer migrations	Floodplain-Channel connectivity	Reduced Seasonal Flows
8	River/Brackish Marsh (*Post-spawn*)			Se, Hg, Pyrethroids, Endocrine Disrupters		Subsistence Fisherman	Shift in diet with clam invasion	Loss of floodplains/riprap of channel banks longer migrations	Floodplain-Channel connectivity	

Table 3. Stressor Understanding Matrix. Tables 2 and 3 do not attempt to call out all possible stressors, only stressors deemed important from a population perspective.

		A		B	C				D	E	F	G	H	I
	Habitat	**Entrainment**		**Stranding**	**Toxics**				**Predation**	**Harvest**	**Food Availability**	**Habitat Loss**	**Barriers**	**Operations**
	(*life stage*)	**Samll Ag**	**SWP/ CVP**		**Se**	**Hg**	**ED**	**PY**						
1	Floodplain/Channel Margin (*Egg/Embryo*)			I = 2 U = 3 P = 3	3 1 1	3 1 1	3 1 1	3 1 1			I =2 U = 3 P = 3	I =4 U = 4 P = 4		I =4 U = 4 P = 4
2	Floodplain/Channel Margin (*Larvae*)	I = 2 U = 3 P = 3	I = 3 U = 3 P = 3	I = 2 U = 3 P = 3	3 1 1	3 1 1	3 1 1	3 1 1			I =2 U = 3 P = 3	I =4 U = 4 P = 4	I =4 U = 4 P = 4	I =4 U = 4 P = 4
3	Floodplain/Channel Margin/Delta *(Juveniles)*	I = 2 U = 3 P = 3		I = 2 U = 3 P = 3	3 2 2	3 1 1	3 1 1	3 2 2	I =3 U = 2 P = 1	I =3 U = 2 P = 2	I =2 U = 3 P = 3	I =4 U = 4 P = 4	I =4 U = 4 P = 4	I =4 U = 4 P = 4
4	*Brackish Marsh* *(Juveniles)*	I = 2 U = 3 P = 3			3 2 2	3 1 1	3 1 1	3 2 2	I =3 U = 2 P = 1	I =3 U = 2 P = 2	I =2 U = 3 P = 3			
5	Brackish Marsh (*Adult*)				3 2 2	3 1 1	3 1 1	3 2 2		I =3 U = 2 P = 2	I =2 U = 3 P = 3			
6	River (*pre-spawnAdult*)				3 2 2	3 1 1	3 1 1	3 2 2		I =3 U = 2 P = 2		I =3 U =2 P = 2	I =4 U = 4 P = 4	I =4 U = 4 P = 4
7	Floodpalin/River (*Spawner*)			I = 2 U = 3 P = 3	3 2 2	3 1 1	3 1 1	3 2 2		I =3 U = 2 P = 2	I =2 U = 3 P = 3	I =4 U = 4 P = 4	I =4 U = 4 P = 4	I =4 U = 4 P = 4
8	River/Brackish Marsh (*Post-spawn*)				3 2 2	3 1 1	3 1 1	3 2 2		I =3 U = 2 P = 2	I =2 U = 3 P = 3	I =3 U =2 P = 2	I =4 U = 4 P = 4	

Future Research

The following is reproduced from Moyle et al. 2004:

> [1]Develop a Systematic Research Program: The hypotheses in this paper indicate that there are many unanswered questions that bear on management. Particularly useful would be radio telemetry and marking studies to track migrations, to determine fidelity to spawning areas, to monitor survival of fish salvaged at the pumping plants and to locate important feeding and spawning areas. As battery technology improves, telemetry studies become more feasible. The information developed here needs to be used in hydrodynamic models of the estuary to determine if changes in flow regime affect movements of splittail between spawning and rearing areas. There is also a need for genetic studies to help determine if more than one population exists in the estuary.
>
> Continue to Use Simulation Models: Potential impacts on splittail of diversions to storage and large, rapid reductions in dam discharge have not been evaluated although appropriate information has only recently become available (Sacramento and San Joaquin River Basins Comprehensive Study, ACE). GIS and survey data should be reviewed to identify floodplain and terrace locations potentially important to splittail. Models should then be run to determine critical flows for maintenance of inundation. Such information, could be used to assess potential impacts (e.g., drying up flooded areas) and to investigate alternative flow management strategies.

Genetic and otolith microchemistry research: The Napa and Peteluma Marsh population may be important for population level production (Feyrer et al. 2005). While these stocks have been identified as genetically distinct from the Sacramento-San Joaquin population (Baerwald et al. 2005), their influence on the entire species population is unknown. A combination of genetic research and otolith microchemistry could be used to identify the exact population structure throughout splittail's geographic range (Feyrer et al. 2007). The otolith microchemistry results could be combined with contaminant exposure biomarkers and splittail could then be effectively used as an indicator species, for example toxic contaminants or changes in the food web, given the current low abundance of delta smelt and longfin smelt.

The following is reproduced from Moyle et al. 2004:

> Improve Estimates of Splittail Abundance: The various fish surveys in the estuary together can be used to provide reasonably good indications of splittail abundance trends, especially for YOY. Individually, most of the surveys suffer from not being designed to sample splittail. The U.C. Davis Suisun Marsh survey most consistently collects all size classes of splittail but the trends for YOY are not always consistent with other surveys. There is thus a need to investigate either the development of a splittail-specific survey or to find ways to improve existing surveys to sample splittail better. For example, USFWS seining surveys could sample additional locations to better assess production in the rivers. Trawling surveys might be able to add stations in shallower areas or near splittail spawning areas. It might also be useful to develop an index of abundance of spawning (adult) fish. A mark-recapture program similar to that for striped bass would likely be the most accurate means to assess adult population size, but would be very expensive. A second approach involves verifying methods of aging splittail, then implementing a means of consistently sampling the adult population annually. By consistently collecting, sexing and aging fish over a short discrete period annually – such as during the spawning migration – over time the resulting data would allow determination of the relative size of each year class and

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[1] The hypotheses indicated in this section can be found in Moyle et al. 2004 pages 24 - 32.

its potential contribution to reproduction in each year. Although this information might not provide a good estimate of current population size, it would likely provide insights into factors influencing population trends and in particular the relative contributions of wet and dry year, year classes to the population. [In addition to the research described above the following are needed as well. Under the current sampling programs splittail are an incidental catch. There is no system wide sampling program that targets splittail specifically. Possibilities include fyke nets at the return ends of floodplains to identify recruitment from floodplain spawning, bottom oriented otter trawls to target upstream migrating adults or downstream migrating juveniles.]

Protect and Enhance Remaining Floodplains and Flood Terraces: In recent years, as CALFED planned riparian restoration projects, the U.S. Army Corps of Engineers (ACE) has been proposing to clear and rip-rap sections of Sacramento River flood-terrace which presently support some of the last remaining riparian forests and splittail spawning habitat. Such areas may be critical for the limited spawning of splittail that takes place in dry years. Habitat restoration is too expensive to allow valuable habitat to be destroyed when other options may be available.

Provide Additional Access to Floodplains: Expansion of easily inundated floodplain habitat should enhance splittail reproduction and abundance, provided new areas are designed to drain properly and lack extensive areas of permanent water to harbor potential predatory fish.

Manage the Yolo and Sutter Bypasses to Benefit Splittail and Other Native Fishes: The Yolo Bypass is clearly a major splittail spawning area and there is strong indication that even partial flooding for a sufficient period can result in successful spawning and rearing by splittail, even in dry years. Ongoing studies of splittail use of the Yolo Bypass should continue, including investigations of creation of spawning and rearing areas in non-flood years. Investigations should also continue on ways to improve the frequency and duration of flows through the bypass (e.g., with gates on the Fremont Weir) for the benefit of splittail and other native fishes. The importance of Sutter Bypass to splittail is less clear but it does have some value for spawning and rearing. This needs to be documented better and ways found to manage the bypass to favor native fishes. [This approach should be used in other floodplain areas throughout the system (San Joaquin, Napa, Cosumnes) as well. This could be one of the most important restoration actions in the entire system.]

Provide Additional Channel Margin Habitat for Juveniles: Shallow margins of Delta channels appear to be important for migration and rearing of juvenile splittail. There is first a need for basic information on the kinds of habitat juvenile splittail use and how they use it, both seasonally and permanently. Means to increase suitable habitat then need to be determined, such as setting back levees, reclaiming islands as aquatic habitat, and breaching levees in marshy areas.

Provide Additional Brackish Water Rearing Habitat for Juveniles: Recent studies suggest that shallow, tidal, brackish water channels along Suisun Bay may be important rearing habitat for splittail. The characteristics of suitable rearing habitat need to be determined and incorporated into marsh restoration projects.

Evaluate Losses of Splittail at State Water Project and Central Valley Project Pumping Plants: The pumping plants in the south Delta capture large numbers of splittail in all life history stages, especially in wet years when splittail are most abundant. However, it is not known (1) what proportion of captured fish are mortalities, (2) if there are high mortalities from predation on fish drawn towards the plants, (3) if capture of adult fish affects their ability to spawn, and (4) if mortalities at the pumping plants have any impact on splittail populations. [(5) The efficacy of trucking including cost and predator induce mortality at the release sites.]

Develop (and Implement) Strategies to Reduce Entrainment Mortality: Splittail larvae and juveniles are entrained not only by the CVP and SWP pumps but probably by the Antioch and Pittsburg Power Plants and other diversions in the Delta. There is still a need to understand what impact these diversions have, if any, on splittail populations. Impacts are most likely to be significant in dry years when a higher percentage of the water is diverted and splittail populations are depleted.

Reduce Pollutant Input, Particularly of Contaminants: Recent evidence indicates adult splittail may be accumulating selenium in concentrations detrimental to reproduction, presumably by consuming the introduced overbite clam (R. Stewart, pers. comm.). There is a need to investigate further the effects of selenium and other contaminants on splittail and to find ways to reduce sources. For example, alternatives to dispose of agricultural drain water from the western San Joaquin Valley include transport and dumping into Suisun Bay. Such an eventuality, without a similar reduction in industrial input, could result in impaired reproductive function in splittail.

Develop a Management Plan for the Fishery on Spawning Migrants: A fishery management plan should be established for splittail to limit the fishery impact on spawners. The fishery should be restricted during drought years.

Important information gaps: Accurate abundance estimates, environmental tolerances of larva and eggs and toxic effects on floodplains from pesticides; selenium and mercury bioaccumulation, and endocrine disrupters; the importance of the Napa and Petaluma Marsh to the structure of the population; trophic transfer of nutrients to the food web; the process of growth, recruitment, and survival throughout splittail life history; genetic distinction and site fidelity between watersheds.

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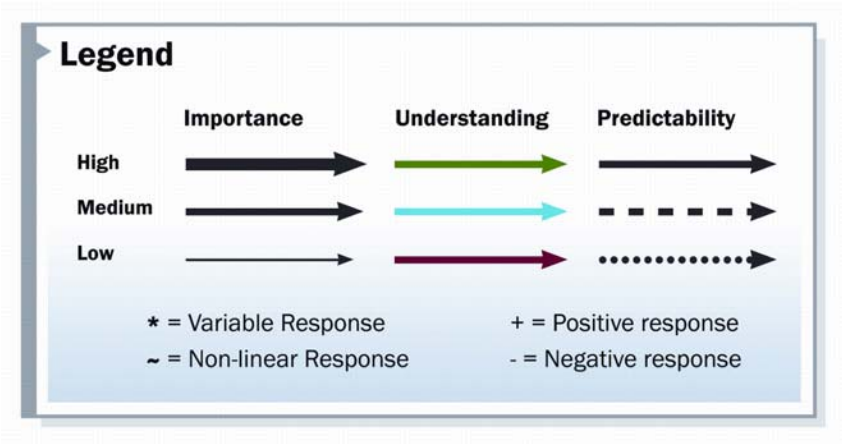
Life History Figures

Figures 5, Depicts life history of splittail under dry and wet conditions.

Figures. 6, 7 and 8 depict life stage transitions matrices.

The arrows (see key below) associated with the sub-models in Figures 5-8 depict the importance of the processes the level of understanding of the processes and the predictability of the processes. A plus sign indicates a positive effect on the transition probability, while a minus sign a negative one.



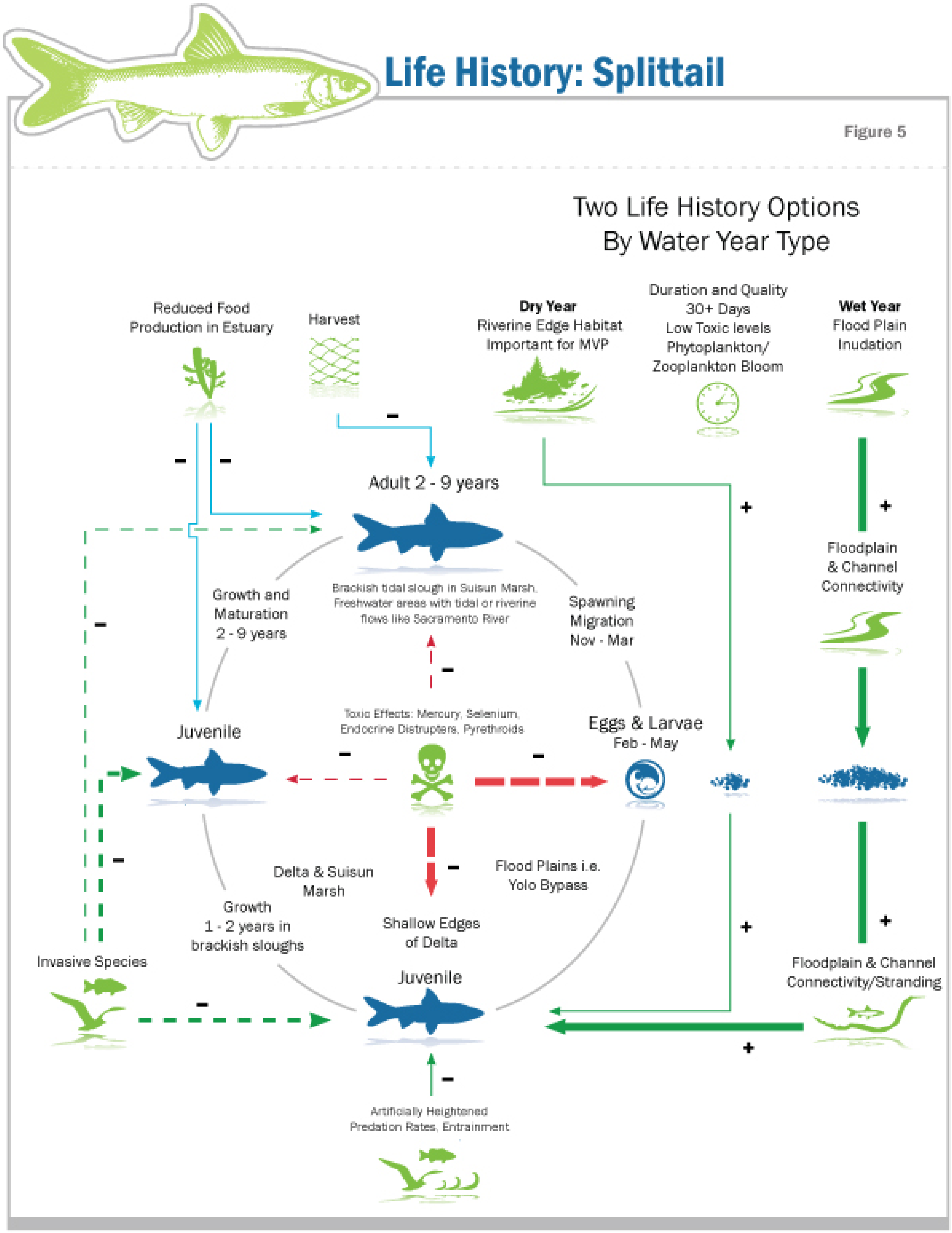


Figure 5. Sacramento Splittail Life History Stressor Model

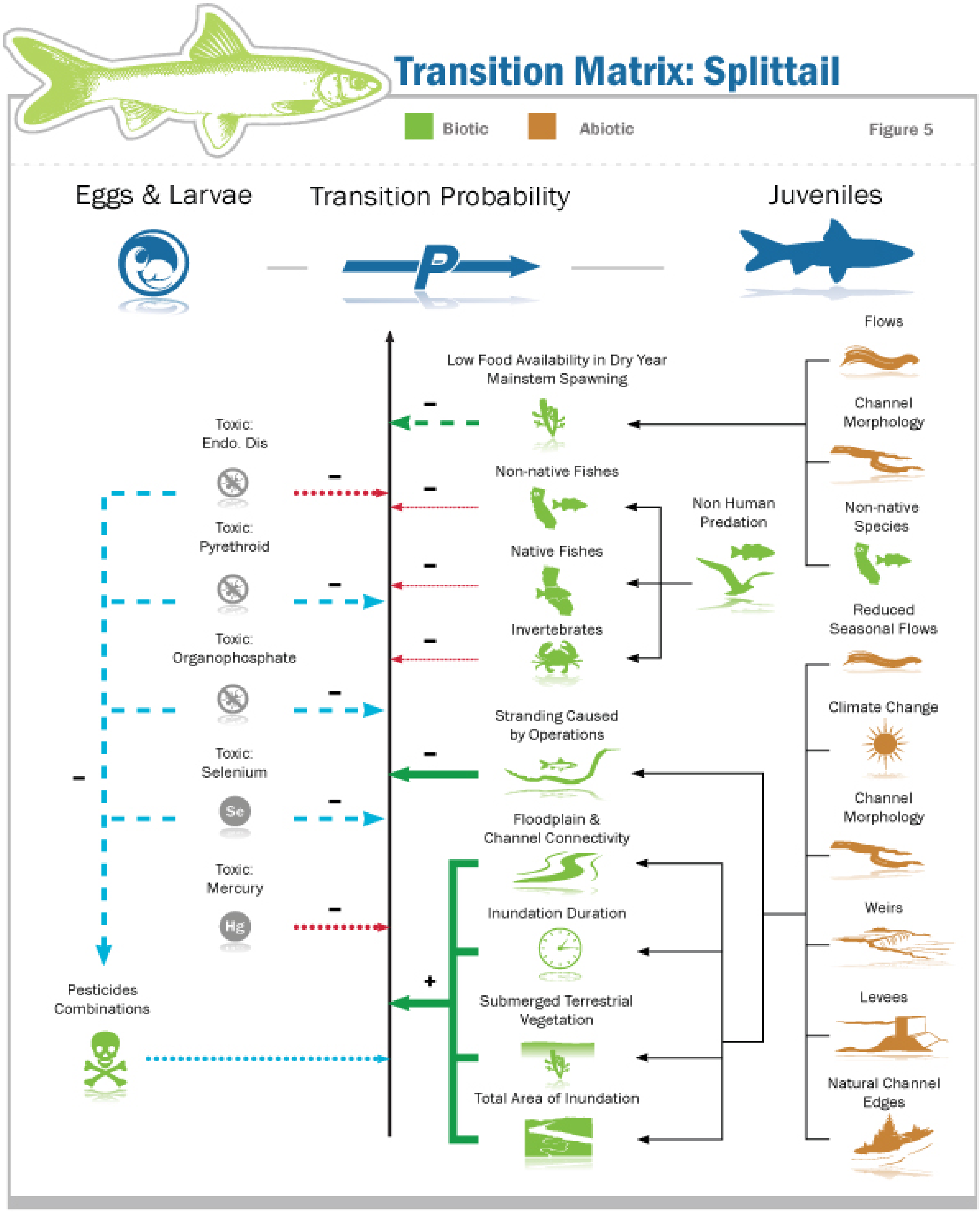


Figure 6. Sacramento Splittail Life History Stressor Sub-Model The arrows associated with these sub-models are described in key 5. A plus sign indicates a positive effect on the transition probability, while a minus sign a negative one.

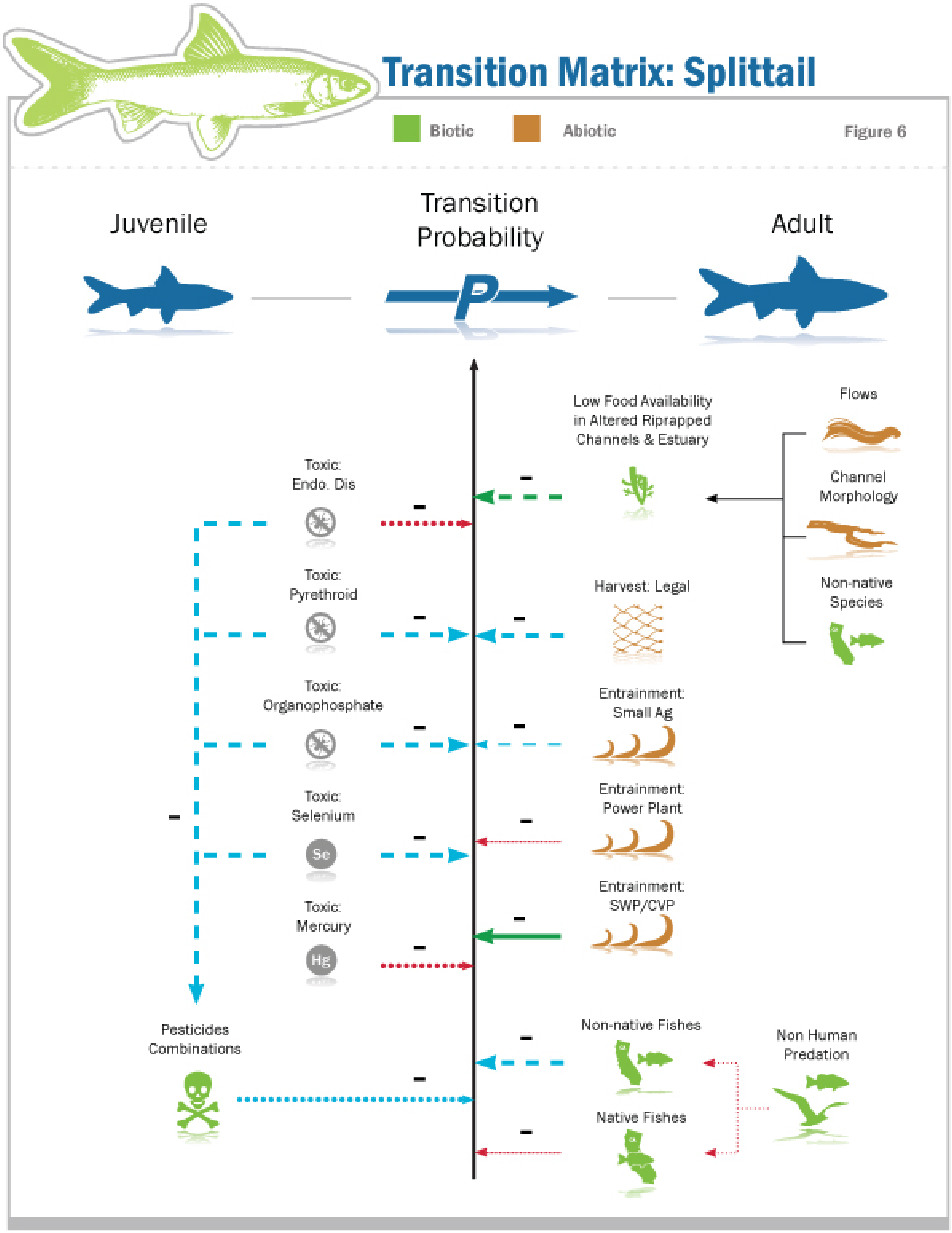


Figure 7. Sacramento Splittail Life history Stressor Sub-Model.

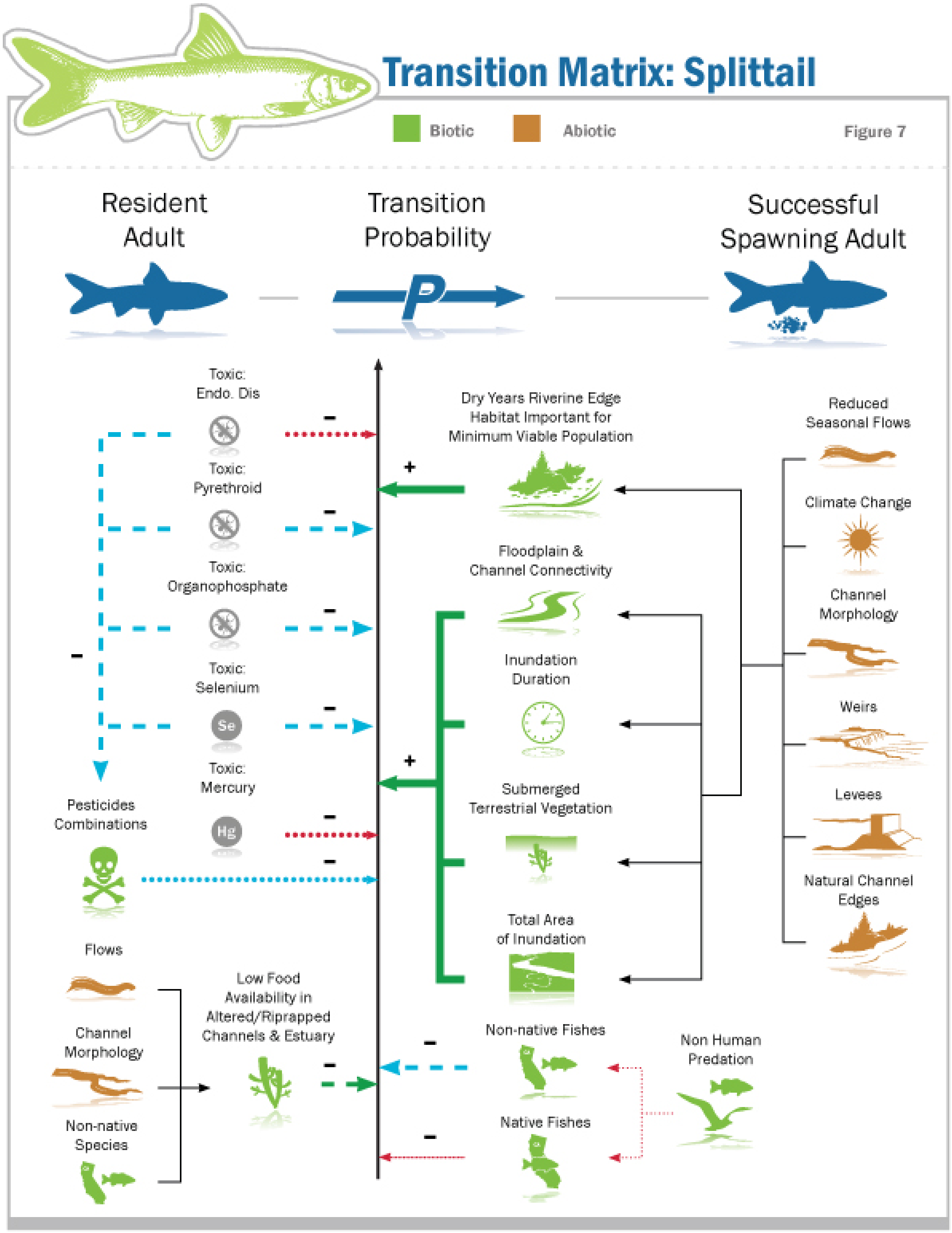


Figure 8. Sacramento Splittail Life History Stressor Sub-Model.

Interagency Ecological Program

Tidal Wetlands Restoration Monitoring Project Work Team

Delta Smelt Conceptual Model

DRAFT

Prepared by: Chris Geach, Jamie Suria, and Gardner Jones

Based on the Interagency Ecological Program's Management, Analysis and Synthesis Team's "An updated conceptual model for Delta Smelt: our evolving understanding of an estuarine fish." Technical Report 90 January, 2015.

Preface

This conceptual model narrative for Delta Smelt relies heavily on the Interagency Ecological Program (IEP) Management, Analysis, and Synthesis Team (MAST) conceptual model of Delta Smelt biology (2015). Much of the Delta Smelt model description is summarized or directly excerpted from the MAST (2015) report. The desire was to distill much of the information, more thoroughly covered in the MAST report, into a form more easily linked to other related models, developed to support and inform the creation of a generalized restoration monitoring plan. The following narrative attempts to present a subset of the MAST report in the context of habitat restoration, being pursued largely based on hypothesized benefits to Delta Smelt, to inform the IEP Tidal Wetland Monitoring Project Work Team process.

Table of Contents

Overview

Introduction

This conceptual model represents the most comprehensive, up-to-date, synthesis of scientific information available for Delta Smelt (*Hypomesus transpacificus*). In addition to information taken whole or in part from the MAST (2015) report, we have also included discussion of implications for tidal wetland restoration and associated monitoring, where appropriate.

Among other efforts to address known and suspected stressors to the Delta Smelt population, restoration of aquatic wetlands has become a focus due in large part to its potential food web benefits. Delta Smelt is a pelagic species that appears to inhabit the near shore environment at lower frequencies and densities. Delta Smelt catch in channels with widths less than 15 meters (Sommer and Mejia 2013) is rare. Habitat parameters that appear most important in determining suitability for Delta Smelt include turbid water (Feyrer et al. 2007), temperatures between 7 and 25 °C, and fresh to brackish water– most Delta Smelt spend the majority of their lives in or just upstream of the low salinity zone (LSZ, ~1-6 psu) (Feyrer et al. 2007; Kimmerer et al. 2009). There is also evidence that open water adjacent to habitats such as tidal marsh, shoal, and low-order channels may be beneficial (Sommer and Mejia 2013).

Delta Smelt are currently listed as threatened under the federal Endangered Species act and endangered under the California Endangered Species Act. Human activities over the last 150 – 200 years have dramatically altered the Delta landscape and ecological functions it supported. Loss of productive wetland habitat through diking and draining has been one of the stressors implicated in the delta smelt population decline. Reclamation of Delta wetlands has reduced available marsh by approximately 95% (Whipple et al. 2012). Habitat restoration has been identified as a potential mechanism to benefit Delta Smelt both directly, through the creation of new areas for rearing and refuge, as well as indirectly, through the subsidy of food from intertidal habitats to adjacent pelagic habitats. While habitat restoration focused at creating, largely, intertidal emergent wetlands may not provide a large benefit to Delta Smelt through direct use, the potential for that restoration to generate local increases in food and improvements to adjacent pelagic habitat, will depend greatly on how it is designed and located relative to the occurrence of Delta Smelt.

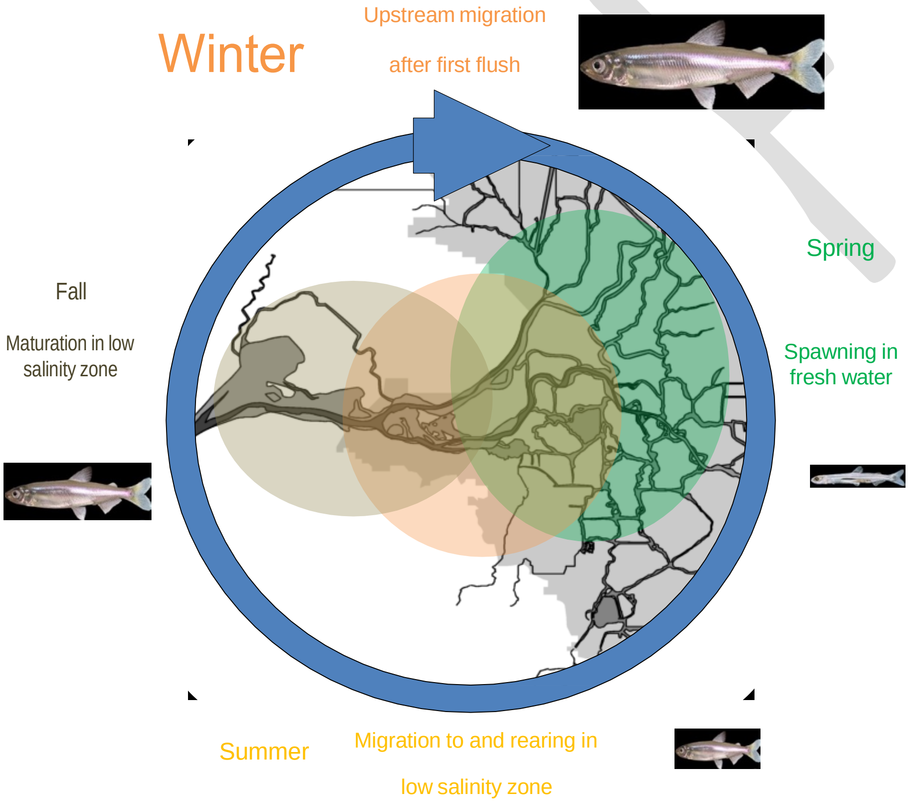
Delta Smelt

The Delta Smelt is a small estuarine fish, endemic to the San Francisco Estuary, that spends its entire, predominantly annual, life-cycle in the Sacramento-San Joaquin Delta, Suisun Bay, and Grizzly Bay(Fig. 5 from MAST report). Although once abundant, it is now rare and protected under both the Federal Endangered Species Act (listed as threatened in 1993) and California Endangered Species Act (listed as threatened in 1993, elevated to endangered 2009). The Delta Smelt population has gone through two apparent step declines; the first in the early 1980's and the second associated with the "pelagic organism decline" (POD) in approximately 2002. Aside from a short-lived increase in abundance in 2011, likely in response to favorable conditions for spawning, growth, and survival, the population has remained at very low levels.

Excerpt from MAST (2015) report:

> “The 2011 increase in the Delta Smelt abundance index demonstrated that the species still has the ability to rebound to higher abundance levels. Delta Smelt has often been called an indicator – or canary in the coalmine – for overall ecosystem conditions in the estuary. The 2011 increase suggests that the system has not yet irreversibly shifted into an altered state that will no longer support native species. Given the profound habitat alterations in the San Francisco Estuary, continued study of the environmental drivers and habitat attributes and the subsequent responses of the Delta Smelt population seem critical to the wise management of the species.”

Figure 5. Simplified life cycle of Delta Smelt (modified from Bennett 2005). Colors correspond to different seasons with the low salinity zone changing position with season.



Model Structure

Excerpt from MAST (2015) report:

> “The Delta Smelt general life cycle conceptual model recognizes the pervasive, year-round importance of the tier 1 landscape attributes and the seasonal importance of the various tier 2

> environmental drivers and tier 3 habitat attributes to the tier 4 life stage transitions of Delta Smelt in the four tier 5 “transition seasons” (Fig. 45). Some habitat attributes – food, toxicity, and predation – affect life stage transitions in all seasons, while other habitat attributes – temperature, entrainment and transport, size and location of the low salinity zone, and harmful algal blooms – affect some life stage transition more than others. Clearly, adequate food must be available at all life stages for Delta Smelt to survive. Toxicity is included during all seasons because we know that contaminants of various types are present throughout the year; however, little is known about the direct or indirect effects of contaminants at ambient concentrations on individual Delta Smelt or the population as a whole. Predation is included in all seasons because we recognize that predation is likely the ultimate cause of mortality for most individual fish; however, responses of Delta Smelt to other habitat attributes and environmental drivers such as food availability and turbidity can modify predation risk.
>
> The mechanistic linkages between landscape attributes, environmental drivers, habitat attributes and Delta Smelt responses in the four life stage seasons are depicted as one-way arrows in four new “life stage transition” conceptual models (Figs. 46-49, please see MAST report). As mentioned in Chapter 3, the life stage transition conceptual models are nested components of the general life cycle conceptual model. Each life stage transition conceptual model includes the habitat attributes hypothesized to affect the transition of Delta Smelt from one life-stage to the next. Hypotheses selected for detailed consideration in Chapter 7 are indicated by “H” in the diagrams. The models also show the landscape attributes and environmental drivers. While the models include many linkages among individual landscape attributes, environmental drivers, and habitat attributes, they do not include linkages between individual habitat attributes and the specific biological processes (growth, survival, reproduction) underlying the life stage transitions. The primary reason for this simplification is that the available data are generally inadequate to fully describe and differentiate among specific functional relationships and mathematical modeling that could help estimate them is beyond the scope of this report. Instead, the combined effects of all habitat attributes on the life stage transition probability are depicted by one upward arrow in each life stage transition conceptual model. This does not imply, however, that all habitat attributes have an equal role in determining life stage transition probability and population success or that the role of each habitat attribute remains constant from year to year.”

Tier 1: Landscape Attributes

Proximity to Ocean

Proximity to the ocean is important for several reasons. It references a site’s position along the ecocline, and relative location within the Bay-Delta landscape. Proximity to the ocean is also important in influencing other habitat parameters including air and water temperature, salinity, and surrounding terrestrial landscape.

Link to Contaminant Model

Proximity to Discharges

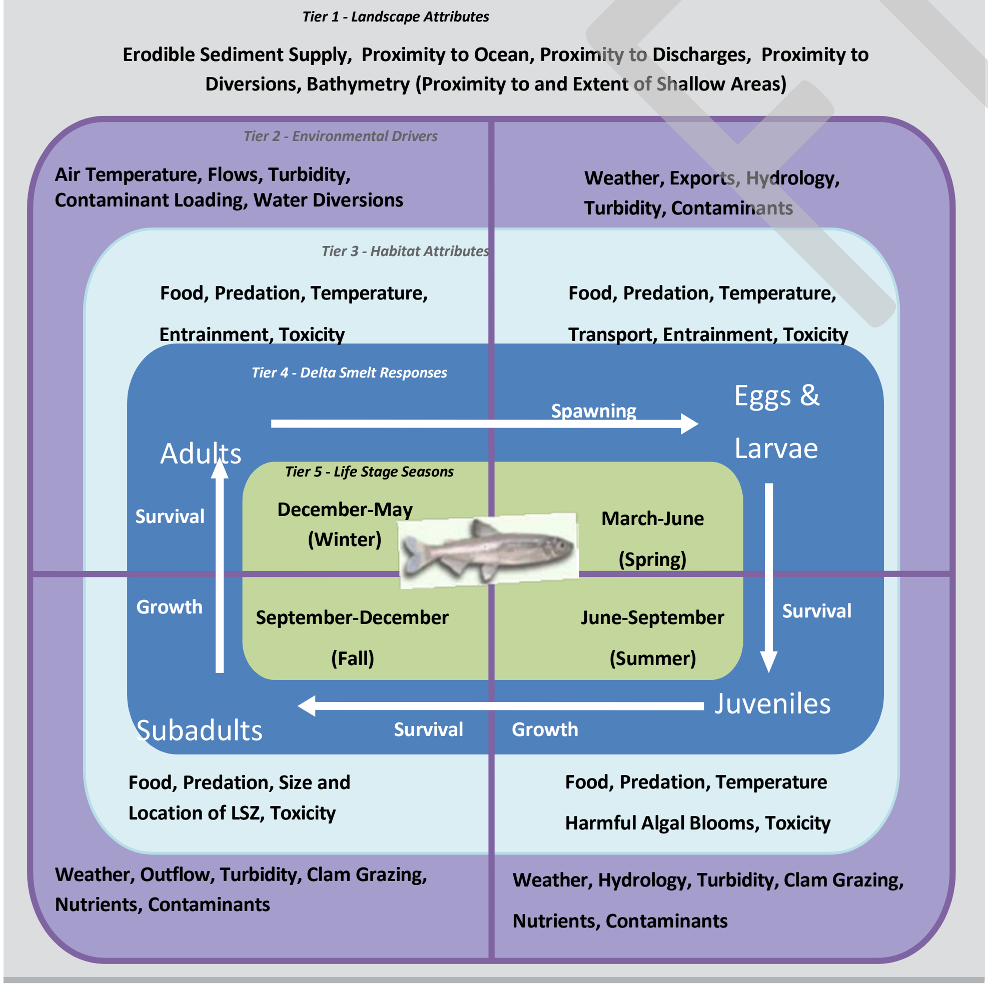
The distance to contaminant discharge sites includes proximity to agricultural, industrial, urban and mining sources, which may be sources of contaminants (see Contaminants conceptual Model).

Link to Wetland Evolution Model

Erodible Sediment Supply

The erodible sediment supply controls the input of suspended sediments into the Delta and is a key feature contributing to turbidity. In combination with bathymetry and hydrology, the erodible sediment supply also effects wetland evolution (see the Wetland Evolution Conceptual Model).

Figure 45. **Delta Smelt general life cycle conceptual model.**



Proximity to Diversions

The proximity to diversion structures, and their relative zone of influence, can affect the movement of water and entrainment of aquatic organisms. This is especially true for pelagic species, such as Delta Smelt and the productivity that supports them (e.g. pelagic phytoplankton and zooplankton), with relatively minimal swimming capabilities. There are numerous small water diversions throughout the freshwater portion of the estuary, but the largest water diversions are at the pumping facilities of the federal Central Valley Project (CVP) and the State Water Project (SWP) that export water from the southwestern Delta to agricultural and urban areas to the south.

Bathymetry (Proximity to and Extent of Shallow Areas)

Bathymetry refers to water depth and effects structural complexity and food production and retention. Water depth will influence residence time of water and phytoplankton, which will have bottom-up effects on food production (both amount and type) and retention for Delta Smelt. Bathymetry plays an important role, in conjunction with hydrodynamics and wind events, in affecting turbidity (Ruhl and Schoellhamer 2004). Additionally, the construction or natural formation of channels will affect Delta Smelt's capability to access the habitat's capacity. The penetration of tidal channels and water depth will determine the duration and frequency of access to restored sites. Restoration of tidal wetlands has the potential to greatly increase the extent of shallow areas.

Location of LSZ in relation to Clam Distribution

The size and location of the low salinity zone (LSZ) appears to play an important role in determining the quality and extent of Delta Smelt habitat for most if not all life stages. The degree to which the LSZ overlaps with and determines the abundance of introduced clams (*Potamocorbula amurensis* in more saline water and *Corbicula fluminea* in fresher water), can modulate the amount and type of food available for Delta Smelt.

Excerpt from MAST report:

> "A dynamic salinity gradient from fresh water to salt water is one of the most characteristic features of an estuary (Kimmerer 2004). It originates from the mixing of fresh inland water with salty ocean water through tidal dispersion and gravitational circulation (Monismith et al. 2002). Many estuarine-dependent organisms occur in distinct salinity ranges (e.g., Kimmerer 2002a) and the extent and location of water with suitable salinities is thus an important habitat attribute for estuarine organisms. Over the time period of available monitoring data, there is no clear long-term trend in salinity levels and distributions in the estuary. Significant increases and decreases linked to changing flow patterns have been detected for various stations and months (e.g., Jassby et al. 1995, Enright and Culberson 2009, Shellenbarger and Schoellhamer 2011, Cloern and Jassby 2012).
>
> The brackish (oligohaline) "low salinity zone" (LSZ) is an important region for retention of organisms and particles and for nutrient cycling. In the SFE, the LSZ provides important habitat for numerous organisms including Delta Smelt (Turner and Chadwick 1972, Kimmerer 2004,

Link to
Clams
Model

> Bennett 2005). In this report we define the LSZ as salinity 1-6; however, other salinity ranges have been used by others, such 0.5-6 (Kimmerer et al. 2013) or 0.5-5 (Jassby 2008)."

Also, see the Invasive Bivalve Tidal Wetland Conceptual Model.

Tier 2: Environmental Drivers

Hydrology, Flows and Outflow

Hydrology influencing the Sacramento San-Joaquin Delta is highly variable both within and across years. Flows into and through the Delta influence both the quantity and quality of aquatic habitat utilized by Delta Smelt

Excerpt from MAST report:

> "Because of California's Mediterranean climate, the SFE experiences large interannual and seasonal flow variations, which are modulated by tides and human management of the rivers within the Delta watershed (Moyle et al. 2010). These hydrological variations lead to a dynamic estuarine salinity gradient. In the winter and spring fresh water often extends into San Pablo Bay, while in the summer and fall brackish water can intrude into the western Delta. These seasonal differences are exacerbated by pronounced interannual differences in precipitation in the watershed. Extremely dry years with little precipitation and very wet years with widespread flooding do not occur in predictable patterns (Dettinger 2011)."

Outflow is discussed, with regard to Delta Smelt, mainly in regard to its influence on the location and extent of the LSZ, which is recognized as playing an important role in determining the extent and quality of habitat for the majority of the Delta Smelt population across multiple seasons.

Air Temperature

Air temperature is driven by its regional climate; the warming or cooling of the surrounding terrestrial landscape, proximity to the ocean and Pacific-related teleconnections, such as PDO (Pacific Decadal Oscillation) and PNA (Pacific North American Pattern) (Prowse et al. 2007). In tidal systems, such as the Delta, water temperature at a particular location is determined by the interaction between atmospheric forcing (e.g., air temperature and wind), tidal dispersion and riverine flows across the estuarine landscape (Monismith et al. 2009). In particular, estuarine water temperature is driven by heat exchange at the air–water interface and mediated by tidal and riverine flow dynamics and estuarine geomorphology (Enright et al. 2013). In the Delta, water temperature is commonly estimated by air temperature and the prior day's water temperature (Wagner et al. 2011).

Turbidity

The Delta Smelt MAST conceptual model incorporates turbidity as a "modifier of several important linkages between environmental drivers and habitat attributes that are important to Delta Smelt, primarily food visibility for small larvae and predation risk for all life stages."

Excerpt from MAST report:

> "In addition to salinity gradients, estuaries often have turbidity gradients. Turbidity is an optical property of water, which is the loss of transparency due to scattering of light by suspended particles. Typically, the upper reaches of estuaries have areas with high levels of suspended particles known as "estuarine turbidity maxima." In many estuaries, these areas are located in or near the low salinity zone and are associated with higher numbers and enhanced growth for larvae of some species (Sirois and Dodson 2000a, b, Shoji et al. 2005). In the SFE, turbidity is largely determined by the amount of suspended inorganic sediment in the water (Cloern 1987, Ganju et al. 2007, Schoellhamer et al. 2012), although organic components can also play a role (USGS 2008). Sediment particles are constantly deposited, eroded, and resuspended, and are transported into, within, and out of the estuary. The amount of sediment that is suspended in the water column depends on the available hydrodynamic energy, which determines transport capacity, and on the supply of erodible sediment in the estuary and suspended sediments from the watershed.
>
> In the upper SFE there are two main physical processes controlling turbidity. Suspended sediment is transported from the tributary watersheds into the system during high flows associated with winter and spring storm runoff (Schoellhamer et al. 2012). The first large storm of the rainy season often carries the highest concentrations of suspended sediment. Some portion of the transported sediment moves through the system to San Pablo and San Francisco Bay and the remainder is stored within the system as bottom sediment. During the remainder of the year, turbidity is primarily caused by interactions of this stored sediment with other environmental drivers (Schoellhamer et al. 2012). Water moving with the tides can resuspend fine sediments because of turbulence resulting from interactions between the bottom and water moving at high tidal velocities. At a larger scale, irregularities in the bottom topography may define geographic regions of greater turbulence and greater turbidity. In the upper estuary, such regions occur at a large bathymetric sill between Carquinez Strait and Suisun Bay and at another location within Suisun Bay (Schoellhamer 2001). Sediments may also be resuspended by turbulence related to wind waves. This process is mainly limited to areas with fine sediments on relatively shallow shoals where wind wave turbulence reaches the bottom. This process is most important in the shallows of Suisun, Grizzly, and Honker Bays and Liberty Island (Ruhl and Schoellhamer 2004, Warner et al. 2004, Morgan-King and Schoellhamer 2013). Thus, turbidity at any particular location is the result of several environmental drivers, including hydrology (transport from the watershed) and weather (wind and precipitation) interacting with the

> physical configuration of the upper SFE. Further, annual variation in these factors may have important effects. For example, during a drought there is little transport of suspended sediment and the same wind patterns during the summer may result in less turbidity than would occur after a wet year because less sediment was stored as benthic sediment during the winter. There is also evidence of longer term changes in turbidity (Schoellhamer 2011, Hestir et al. 2013), along with regional differences.
>
> In addition to the inorganic component of turbidity, organic matter (e.g., phytoplankton) also contributes to both suspended solids and the sediment load on the bed that is re-suspended with wind and wave action (McGann et al. 2013). In the SFE, phytoplankton concentration varies spatially, seasonally, and on an inter-annual scale (Cloern et al. 1985, Jassby 2008, Cloern and Jassby 2012), and is controlled by multiple factors, including benthic grazing, climate, river inflows (Jassby et al. 2002), and nutrient dynamics (Glibert et al. 2011, Parker et al. 2012, Dugdale et al. 2013), which in turn are likely to affect the organic component of turbidity."

Link to Vegetation Model

Restoration sites, with their increased shallow areas and topographic heterogeneity, may have the potential to increase turbidity by allowing resuspension of sediments, however this will depend on sediment availability, wind fetch, and establishment of aquatic vegetation.

Water Diversions

Large scale water diversions (e.g., Delta Cross Channel, Central Valley Project and State Water Project pumps, etc.) and the cumulative effect of small water control structures (e.g., agricultural siphons and pumps, weirs, etc.) can affect Delta Smelt by altering hydrodynamics (Monsen et al. 2007) and/or causing entrainment (Grimaldo et al. 2009). The influence of diversions on Delta Smelt varies by life stage due to changes in swimming ability (larvae have minimal swimming capability) and behavior (e.g. adults migrating to spawning locations alter their risk of entrainment). Additionally, water diversions can remove primary and secondary productivity from the estuary and thereby affect the availability of food for Delta Smelt. Therefore, restoration sites located near water diversions may provide lower quality food and habitat than those located further from diversions. At a minimum, diversions near restoration sites should be monitored for smelt entrainment.

Link to Transport Model

Also, please see the Transport Conceptual Model

Submerged Aquatic Vegetation (SAV)

Link to Vegetation Model

Submerged aquatic vegetation is naturally occurring throughout the Estuary, however invasive vegetation (most notably the invasive aquatic macrophyte *Egeria densa*) has proliferated and displaced many native species. This proliferation, as well as physical attributes of these invasives, has altered much of the aquatic habitat and associated species communities throughout the Delta. In particular, Egeria densa provides favorable habitat for many invasive fishes (e.g. the voracious predator Largemouth Bass *Micropterus salmoides*), and can occur in large dense beds that slow water velocities and trap suspended sediment, measurably reducing turbidity (Brown and Michniuk 2007, Hestir 2010). Extended shallow areas common in restoration sites may encourage growth of submerged aquatic vegetation if water velocities through the site are too low.

Also, please see the Aquatic Vegetation Conceptual Model.

Contaminants

Contaminants are a large factor impairing water bodies throughout the United States. In particular the San Francisco Estuary is currently listed as impaired by the State Water Resource Control Board's 303(d) list due to various types and levels of contaminant pollution (SWRCB 2010). Of those contaminants they include, but are not limited to: pesticides, metals, contaminants of emerging concerns (CECs), and nutrient discharge from watershed or land-based sources (IEP MAST 2015). The transport, residency time, and effect of contaminants on the aquatic environment are driven by a suite of hydrological conditions, many of which are likely to change during the process of tidal wetland restoration. Individual contaminants may not affect fish species at certain concentrations, but there has been evidence that mixture of compounds show adverse effects (Baas et al. 2009, Silva et al. 2002, Walter et al. 2002). Although the frequency and magnitude of the effects on Delta Smelt are still being assessed in the SFE (Johnson et al. 2010, Brooks et al. 2012), we know that other fish species are sensitive to these chemical composition changes which may have behavioral and physiological effects (Radhaiah et al. 1987).

Link to Contaminants Model

In addition to direct effects on fish, contaminants may have indirect effects on prey species. Food limitation is a possible concern for Delta Smelt when herbicides are known to affect primary producers and insecticides may affect invertebrate prey species (Brander et al. 2009, Weston et al. 2012). When Smelt prey upon the contaminated food it could lead to endocrine disruption (Ankley et al. 2005), swimming behavior, immune response, detoxification, and growth and development issues (Connon et al. 2009).

Also, please refer to the Contaminants Conceptual Model.

Weather

Weather influences multiple physical processes which affect the aquatic environment. Key processes in particular are water temperature, rain events, and snowpack. As water temperature rises the bioenergetics demand is greater and therefore creates a need to forage more to meet those requirements. When these temperatures exceed the thermal tolerance and food availability is limited, organisms will die (IEP MAST 2015). In the SFE water temperature varies seasonally and daily (Kimmerer 2004). In tidal systems like the SFE water temperature is dictated by air temperature, wind, tidal dispersion and flow dynamics and estuarine geomorphology (Monismith et al. 2009, Enright et al. 2013).

Rain events occur annually and differ spatially within the context of the Estuary. "The 'first flush' is defined as an increase in flow and turbidity associated with the onset of winter rain (IEP MAST 2015)."

It is seen as a migration cue for sensitive species such as Delta Smelt to move northward to spawning grounds from areas like Suisun Bay (W. Bennett, U.C. Davis, unpublished data). The first flush may bring about contaminant runoff from land-based sources affecting Smelt. The degree in which they are affected is relatively unknown, but as shown from previous sections contaminants may have negative effects. During wetter years river flow from the melting snowpack and rain events keep the low salinity zone (LSZ) closer to Suisun Bay. To that effect there is a larger freshwater habitat for spawning and rearing, which may decrease the probability that migrant adult and larvae Smelt will be entrained in water diversions (IEP MAST 2015).

Exports

Although the SFE is the largest wetland of its kind in the western United States, the Delta has been transformed and engineered to meet freshwater demand for water agencies across the Central Valley and south of Delta users via the Central Valley Project (CVP) and State Water Project (SWP) pumping plants (Kimmerer 2004). Combined with over 2,000 agricultural diversions for local irrigation and 300 floodgate diversions for wetland management, these water exports can have a detrimental impact on Delta fish species (Herren and Kawasaki 2001). Fish can be directly entrained within the two major pumping facilities when exports are high enough. When pumping rates are able to cause negative net flows through Old and Middle River (OMR) (Grimaldo et al. 2009), fish are trapped in the pumping facilities. Within the SWP and CVP fish are either lost due to pre-screen losses due to predation or unfavorable environmental conditions (Kano 1990, Brown et al. 1996, Gingras and McGee 1997, Clark et al. 2009, Castillo et al. 2012). Those that are fortunate to survive are diverted into salvage facilities where they are counted as salvage. Spatial distribution of the population also affects whether or not they are affected by water exports. Delta Smelt larvae that hatch in the San Joaquin River could be more susceptible to entrainment due to their proximity to the major pumping plants (Kimmerer and Nobriga 2008). When fish are not affected directly by water diversions they have the ability to indirectly alter the regional and local hydrodynamics, water quality, and the food available in the SFE (Monsen et al. 2007, Jassby et al. 2002). The degree in which these indirect effects have on species such as Delta Smelt have not been quantified, but “Cloern and Jassby (2012) reported that phytoplankton losses equivalent to 30% of the primary production in the Delta.”

Nutrients

The SFE is relatively nutrient-rich, but lower in productivity compared to other nutrient-rich estuaries. This is thought to occur due to light limitation (Cloern 2001). The Estuary’s nutrient input derives from multiple sources both natural and anthropogenic. Agricultural drains and wastewater treatment plants are among the major sources of urban nutrient input (Hager and Schemel 1992). Currently the Sacramento and San Joaquin River exhibit higher concentrations of nitrogen and phosphorus in part thanks to lower flow and greater agricultural land use (Monsen et al. 2007). Studies have shown that there is a preferential uptake of ammonium in phytoplankton, even when nitrate is available, and sequential uptake of ammonium first then nitrate (Reed et al. 2014). Diatoms and chlorophytes exhibit lower growth rates when they uptake ammonium compared to nitrate. This has implications of a community composition shift towards species that are more efficient at utilizing ammonium (Gildert et

Link to
Food web
Model

al. 2014b). The operation of the Sacramento Regional Wastewater Treatment Plant (SRWTP) is thought to be responsible for the increase of total nitrogen and ammonium in the Delta and Suisun Bay (VanNieuwenhuyse 2007, Jassby 2008). The high ambient ammonium concentrations that originate from the SRWTP have caused the phytoplankton to inefficiently take up any form of nitrogen, which leads to an extremely low biomass (Parker et al. 2012). When a system becomes food limited, Delta Smelt must expend more energy foraging on species that may or may not be energetically beneficial due to the shift in the planktonic community. Nutrients can play a large role in steering phytoplankton dynamics in the SFE as light availability increases (IEP MAST 2015), but environmental factors such as submerged and floating aquatic vegetation may affect that productivity.

Also, please refer to the Food Web Conceptual Model.

Clam grazing

Since the invasion of the overbite clam *Potamocorbula amurensis,* which became abundant by the late 1980s (Kimmerer 2002b), the SFE has experienced a long-term phytoplankton reduction in the upper regions post 1985. The Asian clam *Corbicula fluminea* has also posed a threat to the health of the Estuary. Due to their efficient filter feeding capabilities, grazing can exceed rates of primary production within the aquatic community (Lucas et al. 2002, Lopez et al. 2006), and decrease the amount of food available to species such as Delta Smelt. Clams are in competition with pelagic feeders. Their effect can be seen beyond its brackish salinity range as a result of tidal dispersion of phytoplankton-limited water (Kimmerer and Orsi 1996, Jassby et al. 2002).

Also, please refer to the Invasive Bivalve Tidal Wetland Conceptual Model.

Predators

Predation can be a huge cause of mortality for all life stages of Delta Smelt. The magnitude at which other fish species have on each life stage is still being assessed, but studies have given us a clearer picture as to what is happening within the SFE. Striped Bass, Pikeminnow, and catfish utilize the littoral habitat, therefore having pressure on Smelt due to their spatial and temporal distribution (IEP MAST 2015). It is thought that age 1-3 Striped Bass are the most likely predators of smelt (Baxter et al. 2010), but evidence is still inconclusive (Nobriga et al. 2013). Larvae are more susceptible to predation due to their reduced swimming ability. It is thought that predation may be a dominant source of mortality for larvae, in addition to starvation and dispersion to unfavorable habitats (Hjort 1914, Hunter 1980, Anderson 1988, Leggett and Deblois 1994). The introduced Mississippi Silverside *Menidia audens* is believed to have the most significant impact on Delta Smelt larvae (Bennett and Moyle 1996, Bennett 2005, Baerwald et al. 2012). Alien species invasions and habitat change can have a synergistic affect resulting in predator-prey dynamics that favor predators. Changes to predator-prey dynamics can result in predation becoming a larger factor in fish mortality than prior relationships would have suggested (Kitchell et al. 1994). Habitat restoration that promotes native fish will potentially benefit the survival of Smelt, but will also have the ability to promote native predators like the Sacramento Pikeminnow. If tidal wetland restoration increases aquatic vegetation, restoration may promote non-native predators such as Largemouth Bass and catfish as well.

Also, please refer to the Food Web Conceptual Model.

Link to Foodweb Model

Food Production and Retention

“Food production and retention refer to the physiochemical conditions that maintain prey communities within a restored wetland (Simenstad and Coredell 2000).” Productivity within the Delta has declined due to a multitude of factors which include invasive clams, decline in sediment flux, and changing river flow (Jassby et al. 2002). Sommer et al. (2007) attributes the recent pelagic fish declines to limited food supply, among other factors. As previously noted in the nutrients section above, the phytoplankton production and biomass is quite low in the SFE compared to other estuaries (Jassby et al. 2002, Kimmerer et al. 2005, Wilkerson et al. 2006, Cloern and Jassby 2012). Detritus produced in wetlands has been shown to play a major role in the local food web (Howe and Simenstad 2011) . Microbial consumption of organic detritus in pelagic waters is important, but studies have shown that metazoan production primarily feed on phytoplankton rather than detritus (Sobczak et al. 2002, 2005, Mueller-Solger et al. 2002, 2006, Kimmerer et al. 2005). The complexity between primary production, grazing, and transport time all drive food production and retention within the system (Lucas et al. 2002, 2009a,b, Lucas and Thompson 2012). “The quantity of food available to Delta Smelt is a function of several factors, including but not limited to seasonal trends in prey abundance and prey species specific salinity tolerances, which influence distribution (Kimmerer and Orsi 1996, Hennessy and Enderlein 2013).” Restoration sites with high residence times and low bivalve grazing are predicted to have highest food production.

Also, please refer to the Food Web Conceptual Model.

Tier 3: Habitat Attributes

Food Availability

Historically, the San Francisco Estuary (SFE) was predominately comprised of wetlands and less open water habitat compared to present conditions. It’s believed the microbial food web was responsible for much of the food production in the estuary due to the large extent of wetland habitats. The recent reduction of wetland habitats as a result diking, draining, dredging, and diverting water, has flipped the ratio of wetlands to open water from 14:1, historically, to a current ratio of 1:6 (Whipple et al. 2012, Herbold et al. 2014). Because of this, it’s believed several aquatic species are now primarily dependent on food production from the pelagic food web as opposed to more benthic contributions in the past. For Delta smelt, in particular, phytoplankton production is now recognized as a limiting factor in population success; however, Delta smelt do occasionally take advantage of benthic/epibenthic trophic pathways, which may dominate in tidal wetland sites (Moyle et al. 1992, Lott 1998, Feyrer et al. 2003, Slater and Baxter, 2014, Steven Slater, California Department of Fish and Wildlife, unpublished data).

Phytoplankton is essential to many zooplankton species consumed by Delta Smelt. Phytoplankton production is considered to be primarily light- and temperature- limited, as nutrient concentrations in the Delta are believed to be sufficient for growth. However, changing nutrient ratios and clam grazing has likely altered the quantity and composition of phytoplankton species in recent years (IEP MAST 2015). In addition, residence time of water within a wetland site and riverine flows can influence phytoplankton production (Jassby 2008), with growth increasing as residence time increases. High residence times can increase water clarity and limit nutrient flushing, both of which are important for phytoplankton growth. The highest concentrations of phytoplankton biomass (measured as chlorophyll a) in the Delta are typically in the spring and early summer, with lowest concentrations in the winter. Generally, chlorophyll *a* concentrations across the Delta have been declining since monitoring began (IEP EMP). Much of this decline can be attributed to declines during the May-October period, including the substantial reduction in "spring blooms" in May.

Link to Clams Model

As mentioned above, clam grazing and fluctuating nutrient ratios may be contributing to the changing abundance and composition of phytoplankton communities. The invasive clam, *Potamocorbula amurensis (*overbite clam*),* introduced in the mid 1980's, is believed to be a major contributor to phytoplankton biomass declines in recent history (particularly in the May-Oct time periods)(Alpine and Cloern 1992). *P. amurensis* is capable of removing large amounts of phytoplankton from the pelagic environment in short periods of time. In addition, *Corbicula fluminea* (Asian clam) has been known to reduce phytoplankton abundance, as grazing rates for this species frequently exceed rates of primary production (Lucas et al. 2002, Lopez et al. 2006). Increases in ammonium/ammonia (NH4/NH3), largely attributable to Sacramento Regional Wastewater Treatment Plant (SRWTP) (VanNieuwenhuyse 2007, Jassby 2008) discharges, could be affecting phytoplankton production in the estuary as well. For example, ammonium inhibition can prevent nitrate(NO_3) uptake and possibly growth (debated) for important algal groups like diatoms (Reed et al. 2014), which are believed to have higher nutritional value to Delta smelt prey species. Higher concentrations of NH_4/NH3 tend to favor growth of flagellates and cyanobacteria (Glibert 2012), which can reduce the health of copepod species that are regularly consumed by Delta smelt (Ger et al. 2009, 2010a,b). Other factors such as water export pumping operations can reduce phytoplankton abundance in the estuary (Cloern and Jassby 2012). The susceptibility of phytoplankton to entrainment is highest when water is transported through the Delta towards the export facilities during times when phytoplankton production is high.

A multitude of factors, including shifts in phytoplankton abundance via clam grazing, salinity tolerances of benthic and pelagic invertebrate species, and flows and transport within the estuary can all affect the abundance and distribution of Delta smelt prey species. Excerpt from MAST (2015) report:

> "The changes in phytoplankton production and invasion and establishment of the overbite clam *P. amurensis* were also accompanied by a series of major changes in consumers (Winder and Jassby 2011). Many of these changes likely negatively influenced pelagic fish production, including Delta Smelt. The quantity of food available to Delta Smelt is a function of several factors, including but not limited to seasonal trends in prey abundance and prey species specific salinity tolerances, which influence distribution (Kimmerer and Orsi 1996, Hennessy and Enderlein 2013). Seasonal peaks in abundance vary among calanoid copepods consumed as prey by Delta Smelt, *Eurytemora affinis* in April-May, *Pseudodiaptomus forbesi* in July, and *Acartiella sinensis* in Sep-Oct. Upstream, the calanoid copepod S. doerrii is most abundant May-June. The seasonal trend in cladocerans and mysid prey are similar, being most abundant in summer.
>
> Larger fish are able to take advantage of mysids, cladocerans, and amphipods (Moyle et al. 1992, Lott 1998, Feyrer et al. 2003, Steven Slater, California Department of Fish and Wildlife, unpublished data). The presence of several epibenthic species in diets therefore indicates that

> food sources for this species are not confined to pelagic pathways. Such food sources may be especially important in regions of the estuary where there is extensive shoal habitat such as Liberty Island (Steven Slater, California Department of Fish and Wildlife, unpublished data)."

High quality prey items for Delta smelt, including *E. affinis* and mysids, have shown long-term declines in past years (IEP MAST 2015), including a major step-decline for *E. affinis.* The substantial reduction of *E. affinis* abundance could be related to both indirect and direct effects from clam grazing of copepod food resources (i.e. phytoplankton) and clam predation (Kimmerer et al. 1994). Furthermore, the invasion of non-native clams may have contributed to the decline in the mean size of zooplankton (Winder and Jassby 2011), thereby reducing the quality of prey species for Delta smelt. For more information on the role of clams in the Delta food web, please see the Clam Conceptual Model.

The introduction of non-native copepod species has likely altered the estuarine food web as well. For example, P. *forbesi* may be outcompeting *E. affinis* through its ability to select high-quality algal species and avoid low-quality species such as *Microcystis aeruginosa* (Mueller-Solger et al. 2006, Ger et al. 2010a). However, the distribution and abundance of P. *forbesi* in the Delta and Suisun region has been somewhat variable over the years. It's important to note that while P. *forbesi* isnot native to the *SFE,* it's believed to be an important prey item to Delta smelt. Another invader, *Limnoithona tetraspina,* is now the most abundant copepod species in Suisun Bay and the confluence region of the estuary (Bouley and Kimmerer 2006, Winder and Jassby 2011), presumably due to relatively low mortality rates (Gould and Kimmerer (2010). It's been observed that larval Delta smelt grow more slowly on *L. tetraspina* compared to other copepod species such as *P. forbesi* (Kimmerer et al. 2011). The non-native calanoid copepod, *Acartiella sinensis,* has also been relatively abundant in Suisun Bay and the western Delta region (Hennessy 2010), but its quality as a prey item for Delta smelt is not clear.

Alterations to the Delta food web, including reductions and changes to phytoplankton and zooplankton species, could be having effects on Delta smelt population success. Excerpt from MAST (2015) report:

> "Changes in the quality and quantity of available prey may have contributed to the observed reduction in the mean size of Delta Smelt in the fall since the early 1990s (Sweetnam 1999, Bennett 2005); however, mean size subsequently increased. The importance of food resources as a driver is supported by Kimmerer (2008), who showed that Delta Smelt survival from summer to fall is correlated with biomass of copepods in the low salinity zone, the central 50% of the summer Delta Smelt distribution. Other variations of this correlation were shown by Maunder and Deriso (2011) and Miller et al. (2012). Miller et al. (2012) have tested for an explicit influence of prey density during the fall. Miller et al. (2012) found a stronger correlation between Delta Smelt abundance during the fall and prey density during the fall than for prey density during the summer."

A recent study has indicated Delta smelt feeding success can vary based on their location in the SFE. Juvenile smelt present in Suisun Bay, for instance, generally exhibited higher nutritional stress and had less food in their stomachs compared to fish captured in Suisun Marsh (Hammock et al. 2015). This suggests habitat conditions in Suisun Marsh may be more beneficial to Delta smelt than other various regions in the SFE, where food limitation is affecting Delta smelt health. Future tidal restoration projects may benefit Delta smelt by stimulating local food production, including increased primary production leading to higher abundance of important zooplankton prey species and increased abundance of benthic/epibenthic organisms. Monitoring natural and restored habitats, in addition to the appropriate use of comparative reference sites is vital to understanding ecosystem processes that can enhance food web production to benefit native fish species like Delta smelt. Monitoring actions can also improve our understanding of habitat connectivity and ancillary benefits across the SFE, as it relates to the export of food web subsidies to regions outside the original source area.

For more information on the estuarine food web, please see the MAST (2015) report and the Delta Food Web Conceptual Model.

Predation

Predator-prey dynamics influence species abundance and distribution in many ecosystems. An important function of the interactions between predator and prey is energy transfer from lower trophic level (prey) to higher (predator) trophic level species. Given their small size throughout their lifecycle, Delta smelt are vulnerable to a variety of fish species, including Mississippi silverside (silverside), striped bass, largemouth bass, and possibly even terrestrial species. It’s believed that predation alone is not responsible for the long-term decline in Delta smelt abundance. However, additional stressors may have perturbed the ecosystem (Nobriga et al. 2013) , resulting in a system favoring non-native predatory species over native species like Delta smelt. For example, Mahardja et al. (2016) showed that silverside abundance increased, particularly in the western Delta region, during the same time period as the decline of several pelagic species, indicating a potential Delta ecosystem shift. Excerpt from MAST (2015) report:

> “Predation may become an issue when established predator-prey relationships are disrupted by habitat change or species invasions (Kitchell et al. 1994). The SFE has been extensively modified (Nichols et al. 1986, Cohen and Carlton 1998, Whipple et al. 2012, Cloern and Jassby 2012) so disrupted relationships between predators and prey are certainly plausible. For example, prey may be more susceptible to predation if they are weakened by disease, contaminants, poor water quality, or starvation. Similarly, the creation of more “ambush habitat” (e.g. structures, weed beds), declines in turbidity levels, or the introduction of a novel piscivore also may dramatically shift the existing predator-prey relationships (Ferrari et al. 2014). All of these changes have in fact taken place in the estuary, especially in the central and south Delta (Feyrer and Healey 2003, Nobriga et al. 2005, Brown and Michniuk 2007).”

A myriad of factors can influence Delta smelt predation rates in the Delta, including turbidity, water temperature, SAV, and location in the Delta, among others. Turbidity in a given region is based on the amount of suspended organic and inorganic particles in the water column and can either limit (higher turbidity levels) or increase (clearer water) the ability of visual predators to locate and capture prey species (Ferrari et al. 2014). Higher water temperatures can increase predation rates due to increased energy demands of both predator and prey species (Walters and Juanes 1993), as described in the Water Temperature section. Delta smelt that are in close proximity to shallow water habitats and areas with high densities of SAV could also be more susceptible to predation. Generally, predatory fish species, such as largemouth bass and silversides, are known to inhabit shallow shoreline areas, increasing predation risk for Delta smelt that are near these areas. While tidal restoration projects are generally intended to benefit native species like Delta smelt, careful consideration during the design phase should be taken to minimize potential increases in predators that may also utilize restored habitats. For example, specific habitat features can be incorporated into restoration projects to provide predator refugia for native species and minimize predator-prey encounters.

Population-level effects on Delta smelt by striped bass predation (Mac Nally et al. 2010, Thomson et al. 2010, Maunder and Deriso 2011, Miller et al. 2012, Nobriga et al. 2013) are uncertain. Effects of Largemouth Bass predation on Delta smelt are even less clear, although it's been documented that largemouth bass will readily consume Delta smelt when given the opportunity (Ferrari et al. 2014). Largemouth bass populations have grown with the expansion of SAV (Brown and Michniuk 2007), specifically *Ergeria densa*; however, it's believed Delta smelt are not a substantial part of their diet (Nobriga et al. 2005, Baxter et al. 2010; L. Conrad, California Department of Water Resources, unpublished data). For more information on Delta smelt predation risk, please see MAST (2015) report.

Water Temperature

Water temperatures in the Delta are largely driven by the ambient air temperature and heat exchange at the air-water interface. Air temperature varies across the estuary into the Sacramento Valley, with hotter and drier climate conditions inland and cooler and moister conditions towards the coast. Variations in water temperature are also influenced by seasonal hydrology, winter/spring weather patterns, and tidal dispersion. Tidal interactions with marsh topography can mediate temperatures in tidal sloughs at small scales (Enright et al. 2013). Water temperatures in the Delta vary throughout the year, with median temperatures ranging from 9°C in January to 22°C in July. The highest temperatures are usually found in the south Delta and cooler in the north and western Delta regions and Suisun Bay (IEP Environmental Monitoring Program (EMP).

Many biological, chemical, and physical processes in the estuary, including nutrient cycling and physiological responses of aquatic organisms (Vannote and Sweeney 1980, Poole and Berman 2001, Null et al. 2013) are driven by water temperature. Delta smelt, in particular, have an optimal temperature range in which growth, reproduction, and basic metabolic functions are most efficient and the risk of mortality due to high water temperature is low. The MAST (2015) report discusses the relationship between water temperature and fish physiology:

> "In general, the total metabolic rate of a fish will increase with temperature to an optimum temperature at which, given unlimited food, there is the maximum ability to grow and develop reproductive products (eggs or sperm) in addition to maintaining the basal metabolic rate required for survival, which also increases with temperature (Houde 1989, Hartman and Brandt 1995). As temperature increases beyond the optimum, metabolic rate continues to increase but physiological processes become less and less efficient and more energy is required just to meet the basal metabolic rate of the organism. Eventually, the metabolic rate begins to decline as temperatures approach the physiological limits of the organism and the basal metabolic rate can no longer be maintained."

Predator-prey interactions, food requirements, and contaminant and disease resistance can all be affected by water temperature. Excerpt from MAST (2015) report:

> "Higher water temperatures increase energetic requirements and thus the food requirements of fish. To meet the increased need for food, it is possible that Delta Smelt spend more time foraging during the day. Since greater foraging time during the day increases visibility to predators, and those predators would also increase their foraging rates at higher temperatures, the encounter rate of predator and prey would likely increase at higher water temperatures. The net effect could be an increase in Delta Smelt predation risk (e.g., Walters and Juanes 1993). In addition, resistance to disease and contaminants can also be affected (Brooks et al. 2012). The responses to contaminants can vary depending on the type of contaminant. For example,

> low temperatures can decrease the toxicity of organophosphate insecticides, but increase the toxicity of pyrethroid and organochlorine insecticides (Harwood et al. 2009), a characteristic that has been used in toxicity identification and evaluation (Weston and Lydy 2010)"

Higher water temperatures (at least 19 °C) can also promote Harmful algal bloom (HAB) growth (Lehman et al. 2005), which may reduce Delta smelt habitat quality in the summer and early fall (Baxter et al. 2010). Potential effects on Delta smelt include direct toxicity through exposure to microcyctins, as well as indirect effects through alterations of the Delta food web. For more information on HAB and their impacts on Delta smelt, please see the HAB section.

As it relates to tidal habitat restoration, newly restored areas may either decrease or increase local water temperature, depending on the geomorphology of the site and synchrony of tidal action with time of day (Enright and Culberson 2013). These factors, in addition to other habitat features that can influence water temperature (e.g. shade from vegetation) are important when considering the desired habitat characteristics which are most beneficial to the target species. For more information on estuarine water temperatures and its effects on Delta smelt, please see the MAST (2015) report.

Entrainment and Transport

Link to Transport Model

Excerpt from MAST (2015) report: "The egg, larval, and juvenile stages of estuarine fishes and invertebrates along with small and weakly swimming adult stages are subject to involuntary transport (advection) by riverine and tidal flows. Entrainment is a specific case of involuntary transport. It refers to situations when altered flows misdirect and transport fish and other organisms in directions in which they would not normally travel or where they will encounter unfavorable conditions and increased risk of mortality. In this conceptual model, we use the term entrainment to specifically refer to the incidental removal of fishes and other organisms in water diverted from the estuary, primarily by CVP and SWP export pumping (Arthur et al. 1996, Grimaldo et al. 2009, Castillo et al. 2012)."

Transport and entrainment of Delta smelt in the estuary are influenced by many factors on both spatial and temporal scales. The proximity to water diversions, particularly the CVP and SWP water export facilities, inter- and intra-annual variation in hydrology, and tidal flows can all affect the seasonal distribution of Delta smelt and their risk to entrainment.

For Delta smelt, negative or reverse OMR flows, largely attributable to CVP and SWP export operations, can create a zone of entrainment in the central and south Delta, significantly increasing the chances of fish being transported to the pumping facilities and lost to entrainment (assuming are fish not salvaged). In addition, pumping can create negative flows on the lower San Joaquin River, increasing Delta smelt entrainment risk in that region as well. Risk of entrainment is largely dependent on the magnitude and direction of OMR flows and Delta outflow, and different combinations of both of these factors can either increase or decrease risk of entrainment. For example, in 2005, 2006, and 2011 high inflows into the San Joaquin River dampened the effects of negative OMR flows, while exports remained constant. In contrast, the combination of low inflow and high exports can create conditions with stronger flood tides compared to ebb tides, potentially leading to greater fish entrainment as more fish are transported towards the pumping facilities (IEP MAST 2015). Therefore, areas in the South Delta near OMR and the CVP and SWP export facilities are generally considered poor for tidal wetland restoration projects.

Intrinsic behaviors and life history strategies unique to Delta smelt can influence their position and entrainment potential in the estuary. Generally, entrainment risk is highest for Delta smelt in their early and later life stages (IEP MAST 2015). During spawning migrations, environmental cues such as increases in turbidity around the time of the "first flush" (Sommer et al. 2011, Bennett and Burau 2014) can trigger adult movement from brackish water regions into fresh water upstream reaches. Studies suggest that Delta smelt can be transported upstream by positioning themselves near the shoreline on ebb tides to minimize downstream advection, while moving towards the channel during flood tides to maximize upstream movement with the prevailing flow (i.e. tidal transport) (Bennett and Burau 2014). The timing and extent of spawning movements, as it relates pumping operations and proximity to export facilities, will affect their entrainment risk (Sweetnam 1999, Sommer et al. 2011). Larval Delta smelt are particularly vulnerable to involuntary transport towards the export facilities, given their limited swimming capabilities. Susceptibility to entrainment is dependent on the location of adult spawning and subsequent hatching of larvae (i.e. closer the larvae are to the CVP and SWP export facilities, the higher the risk of entrainment during pumping operations). Most larval fish move from freshwater hatching areas towards brackish water nursery areas near Suisun bay (Moyle et al. 1992), using higher flows in the spring to facilitate this migration. These nursery areas provide favorable habitat conditions, including increased feeding success, growth rates, and survival (Laprise and Dodson 1989, Sirois and Dodson 2000a, b, Peterson 2003, Hobbs et al. 2006). While both adult and young life stage fish are vulnerable to entrainment during spawning migrations and movements downstream, some fish will remain in upstream reaches year-round, specifically in the Cache Slough region, appreciably reducing risk of entrainment. High outflow years can also be beneficial in the sense that the freshwater-low salinity zone (LSZ) interface is pushed westward, creating a larger freshwater habitat near Suisun Bay for spawning and larval Delta smelt. This would provide improved habitat conditions in an area that is located further away from the south Delta export facilities, effectively decreasing risk of entrainment. For this reason, the Cache Slough area has been targeted as the highest priority area for tidal wetland restoration projects.

Toxicity

Toxic effects from contaminants have been a growing concern for many species that use the Delta. Exposure to certain contaminants can affect aquatic organisms by reducing fecundity (Ankley et al. 2005), altering behavior and physiological systems (Radhaiah et al. 1987), and even reducing the quantity and quality of food resources (e.g., Brander et al. 2009, Weston et al. 2012). There is a litany of contaminants and classes of contaminants that are present in the SFE. These include pesticides, NH4/NH3, metals, Polycyclic Aromatic Hydrocarbons (PAHs), and Polychlorinated Biphenyls (PCBs), among others. The sources of most of these can be traced back to historic mining operations within the watershed, agricultural drainage, discharge from wastewater treatment plants, and other watershed- and land-based sources. While Delta smelt are certainly exposed to many potentially harmful chemical compounds, the effects to the population are unclear (Johnson et al. 2010, Brooks et al. 2012).

Link to Contaminants Model

Pesticide concentrations in the Delta vary on a seasonal basis, with concentrations typically highest during winter and spring periods, as pesticides from the inland watershed are transported toward the Delta during higher flow events. Some believe pesticides are one of the key contaminants affecting Delta smelt (Johnson et al. 2010, Brooks et al. 2012, NRC 2012). Direct effects of pesticides are still relatively unknown, although studies on other species have indicated sublethal effects on fish health (Werner et al. 2008, Werner et al. 2010a, Werner et al. 2010b). Toxicity to fish is dependent on ambient conditions and various water quality parameters, including temperature and salinity (Coats et al. 1989, Lavado et al. 2009). As such, the magnitude of effects may either increase or decrease depending on the time of year, environmental conditions in the estuary, and what life stage is present.

Concentrations of other contaminants (i.e. NH4/NH3, metals, PAHs, and PCBs) have been trending higher in the Delta as well, although the degree they are affecting Delta smelt population abundance is uncertain. As discussed in the Food Availability section, NH4/NH3 can affect phytoplankton densities and composition by favoring certain algal groups over others, to the detriment of several estuarine species. Direct effects to Delta smelt are less clear; however, a recent study suggests Delta smelt exposed to high concentrations of NH3 (higher than concentrations typically seen in the Delta) can exhibit membrane destabilization, which may lead to increased membrane permeability as well as increased susceptibility to synergistic effects of multi-contaminant exposures (Connon et al. 2011a, Hasenbein et al. 2013b). Historical mining operations, agriculture, and wastewater discharge operations have all contributed significant amounts of potentially toxic metals and other elements of concerns to the Delta, including mercury and copper. Studies have indicated these elements can impact the health of fish by reducing fertility and growth (Boening 2000, Chapman et al. 2010). An unintended consequence of restoration is the potential introduction of mercury to the aquatic food web via mercury methylation, resulting in the formation of methylmercury (MeHg). Methylation of mercury occurs in anaerobic environments and concentrations of MeHg are highest in wetlands that undergo wetting-drying cycles compared to areas that are permanently inundated (Alpers et al. 2008) Initial site conditions and potential for increased mercury bioavailability (and other contaminants) should be evaluated prior to implementing any restoration project to determine potential effects to environment. The effects of PAHs and PCBs on Delta smelt are less understood, but could be of particular importance during the “first flush” when these sediment-based compounds are transported to the Delta.

Overall, contaminants have the potential to negatively affect Delta smelt health and even food web community structure; however, population-level impacts are not clear. In addition, many studies on contaminant affects have been in laboratory settings that are not representative of actual conditions that Delta smelt experience in the SFE. Additional studies are likely needed to fully evaluate the effects of contaminants on Delta smelt. For more information on contaminants in the Delta, please see the Contaminant Conceptual Model and the MAST (2015) report.

Size and Location of LSZ

One of the most critical habitat areas in the SFE is the LSZ, which is an important region for retention of organisms and particles and for nutrient cycling. This area is defined by a range of salinity from 1-6 ppt, with the center of its position defined as X2. X2 is the distance (km) from Golden Gate moving upstream to the location in the estuary where the bottom salinity isohaline is measured at 2 (Jassby et al. 1995). The size and location of the LSZ are considered key factors determining the quantity and quality of low salinity rearing habitat available to Delta Smelt and other estuarine species. LSZ size and location are determined by the interaction of dynamic tidal and riverine flows across the stationary topography of the region (Reclamation 2011, 2012, Kimmerer et al. 2013). Other factors, including water export operations, can also affect Delta outflow and the LSZ. Typically, high Delta inflow would translate into a proportional increase in Delta outflow; however, Cloern and Jassby (2012) observed that south Delta pumping can moderate the effects of this relationship, resulting in Delta outflow that is lower than would be expected during non-pumping periods. Generally, high Delta outflow pushes the LSZ westward towards Suisun Bay and Marsh, in contrast to lower outflow periods where the LSZ is located above the Sacramento-San Joaquin River confluence. It’s believed the LSZ provides a combination of habitat conditions that are important to many aquatic organisms, including Delta smelt (Turner and Chadwick 1972, Kimmerer 2004, Bennett 2005). The following is a brief description of Delta smelt LSZ habitat use from the MAST (2015) Report :

> "Delta Smelt are found in the estuary at salinities up to 18 (Bennett 2005), but are most common in the in the LSZ (< 6) (Moyle et al. 1992, Sommer and Mejia 2013, Kimmerer et al. 2013). Sommer et al. (2011a) described Delta Smelt as a "diadromous species that is a seasonal reproductive migrant." In the winter, adult Delta Smelt move upstream into fresh water for spawning. In the spring and summer, young Delta Smelt are transported or swim downstream into the LSZ (Dege and Brown 2004). Delta Smelt usually rear in low salinity habitat in the summer (Nobriga et al. 2008) and fall (Feyrer et al. 2007), although some Delta Smelt remain year-round in fresh water (Sommer et al. 2011a, Merz et al. 2011, Sommer and Mejia 2013)."

The location of X2 (LSZ) during certain times of the year has received considerable attention in the scientific community, with numerous estuarine species responding to the position X2 (Jassby et al. 1995, Kimmerer 2002a, Kimmerer et al. 2009). Delta smelt summer abundances have shown statistically significant negative relationships with prior fall X2 and fall abundance (i.e. as X2 moves seaward, abundances increase) (USFWS 2008, Mount et al. 2013). The exact mechanisms behind this relationship are unclear but are likely related to complex interactions of both abiotic and biotic factors (i.e. nutrients, turbidity, invasive species, and subsequent food web responses). Excerpt from MAST (2015) report:

> "For example, LSZ position affects recruitment of the invasive clam Potamocorbula amurensis, which may in turn affect phytoplankton and zooplankton biomass, size, and production (Thompson 2005, Winder and Jassby 2011), and has likely affected fish-X2 relationships (Kimmerer et al. 2002a)."

Modeling efforts have indicated when X2 overlaps with Suisun Bay, large areas of shallow (Kimmerer et al. 2013), low salinity habitat are available to Delta smelt. This is also one of the most turbid areas in the region with turbidity being closely associated to Delta smelt presence (Kimmerer 2009). However, a recent study indicates water velocity may be more important to Delta smelt than shallow water depth, with Delta smelt presence (at FMWT stations) best explained by secchi depth, salinity, and maximum depth-averaged current speed (Bever et al. 2016). The depth of the LSZ is important in that it can stimulate or limit food production for lower trophic level species. In shallow waters, light can penetrate throughout the water column, increasing the potential for phytoplankton growth (Cloern et al. 1983; Lucas et al. 1999). However, increases in phytoplankton can also be mediated by clam grazing, as clams filter a higher fraction of phytoplankton biomass compared to deeper water areas (Lucas and Thompson2012). Restoration sites, with their large extents of shallow water, may be particularly beneficial for Delta Smelt if positioned in the LSZ.

Harmful Algal Blooms

Toxic blue-green algae, *Microcystis aeruginosa*, has become a growing concern in the Delta. Blooms usually occur around August and September and are most prevalent in the south and east Delta, as well as the lower San Joaquin River regions. Its distribution may be expanding northward, however, as it has been occasionally observed in the north Delta (Morris 2013). *Microcystis* growth has been correlated to low flows and high residence times, elevated nitrogen levels, and clear water (Lehman et al. 2005, Baxter et al. 2010, Lehman et al. 2013, Morris 2013), with these conditions most prevalent in dry years. Submerged aquatic vegetation (SAV) can influence growth by capturing suspended particles in the water column, thereby reducing turbidity levels and increasing water clarity. The interaction of all of these factors, along with potential grazing effects of clams, is believed to affect the distribution and growth of *Microcystis* in the SFE. In addition, riverine flows may transport *Microcystis* towards downstream reaches (Baxter et al. 2010) where Delta smelt are more likely to occur. If restoration sites have particularly high residence times, low flows, or elevated nitrogen, they should be monitored for potential increases in HABs.

It's hypothesized that the proliferation of *Microcystis* is reducing the quality of habitat and even affecting the distribution of Delta smelt (Baxter et al. 2010). Toxic effects on Delta smelt could occur by direct exposure to *Microcystis*, as demonstrated by studies on other Delta fish species such as Mississippi silversides (Lehman et al. 2010). Generally, early - to mid-life stage Delta smelt are most susceptible to *Microcystis* exposure, as the presence of these life stages temporally overlap with *Microcystis* when it's at its highest concentrations. However, direct toxic effects on Delta smelt are uncertain. Indirectly, *Microcystis* could be impacting Delta smelt by disrupting the Delta food web (Lehman et al. 2010), particularly relating to changes in the abundance, health, and species of copepods that are consumed by Delta smelt (Ger et al. (2009). For more information on the effects of *Microcystis* on Delta smelt, please see the MAST (2015) report.

Tier 4: Delta Smelt Responses

Information below is taken and summarized from MAST (2015) report.

Eggs and Larvae (Survival) (March-June)

The abundance of eggs and larvae of the Delta Smelt is often used as an indicator of the following year's recruitment. Their small size at this point in their life cycle limits the size of what prey items are available to ingest and makes them more vulnerable to a wider range of predators. Due to their limited swimming ability, larvae are more susceptible in being diverted to the major water export facilities. However, some studies have shown that this point in their life cycle may not play as large of an importance as an indicator in regards to the other life stages. While spawning habitat requirements are unknown, monitoring restoration sites for smelt larvae may allow managers to identify areas where smelt are spawning.

Delta Smelt larval survival is greatly affected by water temperature. Those that hatch later in the spawning season may experience higher temperatures, which reduce survival. As water temperatures rise, metabolic demand increases until it reaches a point where stress levels have detrimental effects in mobility and behavioral function. Bennett (2005) showed that optimal hatching and larval survival in culture experiments occurred between 15 and 17˚C. When temperatures exceed that threshold, basal metabolism of the developing embryo expends more energy on survival, leaving less energy for growth.

Distribution of larvae in the SFE is thought to be largely dictated by the location of the LSZ and accompanying salinity and turbidity gradients, which have cascading affects to other habitat attributes. Abundance of larvae was well explained in Kimmerer et al. (2009) by salinity and Secchi depths, which is used as a proxy for turbidity. "Jassby et al. (1995) showed that from 1968-1991 the highest fall abundance indices for Delta Smelt had coincided with intermediate values of average April to July X2 when the LSZ was positioned in Suisun Bay." Post-larval and small juvenile distributions in the spring were upstream of X2, but as they grew distribution approached X2 (Dege and Brown 2004). Although this notion suggests that the LSZ is correlated to growth, recent evidence suggests that it may be more variable than previously thought. Some smelt remain in upstream areas, such as Cache Slough, throughout the year. (Sommer et al. 2011a, Contreras et al. 2011, Merz et al. 2011, Sommer and Mejia 2013). Turbid water is essential to late-larval and juvenile survival (Sommer and Mejia (2013) and Nobriga et al. (2008)). Certain turbidity ranges allow for optimal foraging and reduce overall environmental stress (Lindberg et al. 2013). It is thought that the suspended particles in the water column provide a background of stationary particles that help larvae detect moving prey and forage more easily. A study by L. Sullivan (SFSU, unpublished data) showed that vertical distribution of larval smelt shifts upward in the water column when turbidity, measured by NTU, is increased to 24 NTU thereby suggesting that larvae are utilizing the turbid water to safely forage in surface waters that are food-rich.

Food availability, competition, and predation all affect survival and fitness of eggs and larvae. As stated above, this particular life stage is vulnerable to a host of different predators. Anthropogenic effects have drastically altered the physical and environmental aspects of the SFE. Major water diversions, such as the CVP and SWP, and contaminant runoff from agriculture and urban areas have had negative impacts in the aquatic realm. Although larvae that are less than 20 mm fork length have not been reported in the salvage data of the state and federal pumping facilities, entrainment losses have been calculated to be substantial under some circumstances (Kimmerer 2008). Larvae are most vulnerable to being diverted into the facilities as they move from freshwater in the southern and central Delta into the brackish water of Suisun Bay to rear. Contaminants are particularly important to consider when assessing Smelt survival since it is apparent throughout the year. When peak densities of larval and juvenile Smelt are present sometimes corresponds in space and time with elevated concentrations of dissolved pesticides in the spring which persist up to 2-3 weeks (Kuivila and Moon 2004).

Juveniles (Growth and Survival) (June-September)

The juvenile life stage occurs within the warmest period of the year, so experiences the highest water temperature. In the south Delta, water temperatures are warmest in July and August. As they migrate towards their rearing area in Suisun Bay, energetic costs are higher due to the warmer temperatures. Lab studies of wild-caught juvenile Delta Smelt have shown that they could not tolerate temperatures higher than 25.4˚C after being acclimated to 17˚C (Swanson et al. 2000). "Based on the TNS (Nobriga et al. 2008) and the 20 mm Survey (Sommer and Mejia 2013), most juvenile Delta Smelt were predicted to occur in field samples when water temperature was below 25 °C." Thermal critical limits of Delta Smelt have increasingly been of concern with the current drought and forecasted climate change. Several factors have prompted juvenile distribution to utilize certain areas of the Delta to grow efficiently. Juveniles primarily rely on calanoid copepods, such as *E. affinis* and *Pseudodiaptomus forbesi*, in freshwater and the Cache Slough-Sacramento River Deep Water Ship Channel, but there is variability among regions. It is likely that the turbidity range with the highest feeding rate (12-120 NTU) is within those that are experienced by juvenile Delta Smelt in typical summer conditions (Hasenbein et al. 2013).

In the south Delta there has been a long term decline in turbidity observed, which may be in part why juveniles are now rarely seen in that area during the summer (Nobriga et al. 2008).

Outflow in the Sacramento and San Joaquin Rivers are drivers that increase entrainment in this critical period of Delta Smelt life history. Additionally, Morinaka (2013a) and Kimmerer (2008) found that juveniles that are less than 30 mm fork length are less efficiently captured at the water conveyance facilities.

Contaminants and harmful algal blooms (HABs) affect both growth and survival. They can have direct and indirect effects that threaten aquatic species. Connon et al. (2011b) demonstrated that Delta Smelt that are exposed to copper exhibited reduced swimming velocities and had digestive and neurological effects. Contaminants may not be directly impactful, but herbicides and insecticides may affect primary producers and invertebrate prey species respectively (Brander et al. 2009, Weston et al. 2012). In doing so this has a harmful effect on Delta Smelt diet whereby the toxins will be transferred via bioaccumulation. Pesticides in particular may result in metabolic disturbances, growth retardation, and reduction in longevity and fecundity (Murty 1986). Direct effects of *Microcystis* have not been observed on Delta Smelt, however it is known to be toxic to native Sacramento Splittail (Acuña et al. 2012a) and non-native Threadfin Shad (Acuña et al. 2012b). With rising water temperatures during the summer months, *Microcystis* poses a greater threat. Blooms require temperatures of at least 19˚C for initiation, and low flows during the summer increase residence time of water and may allow *Microcystis* to bloom.

Subadults (Growth and Survival) (September-December)

Subadults are still rearing in the LSZ habitat during the summer and fall (Feyrer et al. 2007). Although temperatures are declining as the year progresses, MacNally et al. (2010) found that high summer temperatures have a negative effect on subadult abundance in the fall. Water exports are continually affecting Delta Smelt, whether it may be indirect by contributing to habitat alteration and changing outflow, position of the LSZ, or entraining prey species (Jassby et al. 2002) or direct effects ending in entrainment at the salvage facilities. Strict regulations dictate water operations, and it has been observed that Delta Smelt benefit when fall X2 is located in the shallow bays of Suisun however juvenile and subadult response to spring X2 is currently unclear.

As juveniles and subadults rear in Suisun Bay, food availability shifts as seasonality and environmental conditions change. The LSZ functions as a nursery for Delta Smelt, where they grow into adults in order to continue the species for the next generation. In the SFE *Limnoithona tetraspina* has become the most abundant copepod species (Bouley and Kimmerer 2006, Winder and Jassby 2011). This shift in planktonic availability has left Delta Smelt, and other pelagic fishes, with an abundant prey species that is thought to function as an inferior food due to its small size and ability to detect and avoid predators (Bouley and Kimmerer 2006, Gould and Kimmerer 2010). If fish are unable to meet metabolic requirements due to in part by inefficient prey, it will starve and affect its growth and survival. Subadult diet is similar to that of juveniles, but with increased variability that coincides with the seasonal decline in pelagic zooplankton (Moyle et al. 1992, Lott 1998, Steven Slater, CDFW, unpublished data).

Adults (Spawning) (December-May)

Adult Delta Smelt spawning success in part plays a vital role in the following year's recruitment and population stability. Due to their annual life cycle, the adult stage is primarily focused on reproductive viability. Many factors whether natural or anthropogenic affect overall fitness of adult Smelt to expend energy into metabolic processes instead of reproductive success.

Like other life stages, adults are commonly found in the low salinity zone (Moyle et al. 1992, Sommer and Mejia 2013, Kimmerer et al. 2013) generally centered in Suisun Bay depending on water operations. Rising water temperatures creates a spawning window that can be sub-optimal. The spawning window can be defined as the beginning and duration of the spawning season (Bennett 2005, Mac Nally et al. 2010). Culture studies have shown that there is a range of temperatures that are ideal for spawning success (Bennett 2005, Wang 1986, Baskerville-Bridges et al. 2004b). If adults are provided with a long enough spawning window, individual females have the means to repeat-spawn during the season (Bennett 2005, J. Lindberg, U.C. Davis, personal communication 2013, L. Damon, CDFW, written communication 2012).

In addition to ideal water temperatures, fecundity of the population is based on the size and numbers of females present (Bennett 2005, DFW unpublished, Lindberg et al. 2013). Food availability drives the survival of the adults and energy expended on egg production. Sweetnam (1999) documented that the mean size of adults declined in the early 1990s, but substantially recovered in the 2000s. Delta Smelt diet has shifted to primarily consist of benthic invertebrates, which is believed to be caused by a food limitation with the decline and change in composition in the pelagic food web (Slater and Baxter 2014). Regional differences in food use and availability demonstrate this notion. In 2012 adults occupying the LSZ had a diet composition which included cyclopoid copepods, other than *Limnoithona* species, amphipods, cladocerans, cumaceans, and larval fish. Adult Smelt diet in the Cache Slough-Sacramento River Deepwater Ship Channel was predominately made up of the calanoid copepod *Sinocalanns doerii* (Steven Slater, CDFW, unpublished data).

Rain events, such as the winter "first flush" event and water exports have a suite of effects to the migrating Delta Smelt to their spawning groundsEntrainment risk is much higher for those adults that move upstream to the Sacramento River, but is much lower for the Smelt that tend to stay in the Cache Slough region or do not move upstream at all.

With the first flush triggering migration towards freshwater comes a greater concern for contaminants affecting fecundity and biology. The northern Delta, where spawning and larval nursery areas are located, is of particular concern. This region is at risk of exposure to ammonia and ammonium discharges from the Sacramento Regional Wastewater Treatment Plant (Connon et al. 2011a). These discharges have the potential to affect all life stages since nutrients are discharged year-round. Sub-lethal concentrations have resulted in histological effects such as gill lamellae fusions and deformities in other species (Benli et al. 2008). McKenzie et al. (2008) demonstrated that neurological and muscular impacts of ammonia and ammonium resulted in slowed escape response which may lead to mortality. If adults are being predated on due to these contaminants, then they have no chance of spawning and releasing eggs into the substrate.

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Interagency Ecological Program

Tidal Wetlands Monitoring Project Work Team

Tidal Wetlands Conceptual Model

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Based on the DRERIP Tidal wetlands conceptual model:

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PREFACE

This Conceptual Model is part of a suite of conceptual models developed by the IEP Tidal Wetland Monitoring Project Work Team to guide monitoring of tidal wetland restoration sites within the Sacramento-San Joaquin Delta (Delta) and Suisun Marsh. The conceptual models have been developed based on the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) models, and are designed to aid in the identification and evaluation of monitoring metrics for tidal wetland restoration projects. These models describe our understanding of how tidal wetland restoration may benefit at-risk fish species.

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Introduction

Tidal wetlands are (or were, historically) dominant features of the intertidal zones of many temperate and subtropical estuaries and coastal regions. They can be characterized by salinities ranging from full-strength seawater to freshwater, and extend geographically well inland within gently sloping river valleys. Though the term “marsh” is often used synonymously with “wetland,” the wetland concept used here includes much more than marsh plains populated with emergent, hydrophytic plants. Tidal wetlands are mosaics of varied interconnected habitats, which may include shallow open water, subtidal and intertidal channels, vegetated marsh plains, mudflats, salt pannes, and upland transitions (Mitsch and Gosselink 1993). There is a growing appreciation for the importance of heterogeneity of habitat types and the interfaces between those habitats for structuring biological communities within wetlands and facilitating proper ecosystem function (Nobriga et al. 2005; Gewant and Bollens 2012; Cohen et al. 2014; Jager 2014; Whitley and Bollens 2014).

Though the economic valuation of wetlands is not particularly straightforward (Mitsch and Gosselink 2000), tidal wetlands provide a host of ecosystem services including the maintenance of biodiversity, fish and wildlife habitat, water quality, flood abatement and carbon sequestration (Rabenhorst 1995; Costanza et al. 1997; Bottom et al. 2005; Zedler and Kercher 2005; Barbier et al. 2010). Despite their critical functions, many wetland landscapes have been destroyed or irreparably altered either incidentally or intentionally by human activities (Holland et al. 2004; Zedler and Kercher 2005; Cloern and Jassby 2012; Schile et al. 2014).

In this conceptual model (Figure 1), we have attempted to capture the essential aspects of tidal wetland function with application to the Sacramento- San Joaquin Delta (“Delta”) and Suisun Marsh (collectively “upper estuary” or “San Francisco Estuary” (SFE); see Figure 2). Within the SFE, multiple stressors assail the aquatic ecosystem, which has resulted in the listing of several fish species under the federal Endangered Species Act (ESA) and/or the California Endangered Species Act (CESA). Among them are winter-run Chinook Salmon *Oncorhynchus tshawytscha* (ESA and CESA endangered), spring-run Chinook Salmon (ESA and CESA threatened), Delta Smelt *Hypomesus transpacificus* (ESA threatened, CESA endangered), and Longfin Smelt *Spirinchus thaleichthys* (CESA threatened). Restoration of tidal wetland habitats that function in a similar way to the habitats with which these species evolved is now required under ESA and CESA to partially mitigate for ongoing human activities within this highly altered estuary (CDFG 2009, NMFS 2009, USFWS 2008).

Operation of the federal Central Valley Project (CVP) and State Water Project (SWP) is among the most significant ongoing human activities in California, as these projects, along with other water projects, effectively re-plumb the state’s water. The combined watersheds of the Sacramento and San Joaquin Rivers cover 163,000 km^2, but much of the runoff they collect (median 39% of unimpaired inflow) does not reach the San Francisco Bay (Cloern and Jassby

2012). Rather, dams, diversions, and aqueducts re-route the water for agricultural and municipal use throughout the state. The CVP and SWP alter the magnitude, variability, and sometimes even the direction of flow, fundamentally changing the functioning of the aquatic ecosystem of the upper estuary and of wetlands that remain today (Cloern and Jassby 2012).

In addition to modern altered hydrology, historical conversion of tidal wetlands to other land uses has impacted the SFE ecosystem, most dramatically in the Delta. Reconstructions of the historical ecology of the Delta depict a pre-1850 landscape dominated centrally by perennial tidal freshwater emergent marshes interspersed with numerous, highly sinuous tidal and subtidal channels. The northern and southern peripheries of the Delta graded into non-tidal freshwater wetland, riparian forests, and seasonally inundated floodplains (Whipple et al. 2012). The majority of today's Delta consists of agricultural tracts protected from tidal waters by man-made dikes or levees, which are commonly armored with rip-rap. The de-watered, rich peat soils of these created islands have supported abundant agricultural production, but have oxidized, compacted, and blown away in the process, causing significant subsidence (Deverel and Leighton 2010). Occasional levee failures turn islands into lakes; a few large shallow lakes remain after accidental levee breaches were not repaired. Overall, today's Delta contains 97% less freshwater tidal wetland than its historical state and nearly double the open water area (Whipple et al. 2012). In Suisun Marsh, large tracts of wetland remain, but many are diked and managed as "duck clubs." Tidal action on the duck clubs is muted or prevented entirely, which may restrict habitat used by fish

Most aspects of the fish community and population dynamics of the pre-1850s SFE are not documented in a systematic way; quantitative data were collected only after significant human alteration of the landscape. A strong relationship between tidal wetland extent and the status of native fishes has yet to be supported by the available data; however, the paucity of relict tidal wetland remaining in the upper estuary, combined with multiple other stressors, may conceal the true importance of tidal wetlands to native fishes (Brown 2003; Herbold et al. 2014). In systems that retain a significant extent of tidal wetlands, their value to fishes is strongly supported, not only for the habitat that they provide, but also for their high productivity and contributions to the food web (e.g., Kneib 1997; Castellanos and Rozas 2001; Peterson 2003). Most of these studies have occurred in salt marshes; freshwater tidal wetlands have not received as much research attention due to their lower abundance, though salinity clearly influences wetland function (Więski et al. 2010; Schile et al. 2014; Sutter et al. 2014). Tidal wetland restoration in the upper estuary, supported by strong experimental and monitoring components, is necessary to determine if the same habitat and food-web benefits realized elsewhere will be provided to species of interest in this system (Brown 2003; Herbold et al. 2014).

Food limitation and lack of appropriate habitat are among the multiple factors thought to contribute to the decline of fish species native to the SFE (Baxter et al. 2010; Sommer et al. 2007; Mount et al. 2012). Therefore, the potential for tidal wetland restoration to benefit ESA and CESA listed species has garnered much interest. Although restoration of tidal wetlands will affect a multitude of other species, our conceptual model focuses on the aspects of wetlands that affect these at-risk fish species. The purpose of this model is to aid construction of a monitoring program best suited to quantifying the effects of restoration on habitat and food supply for salmon and smelt.

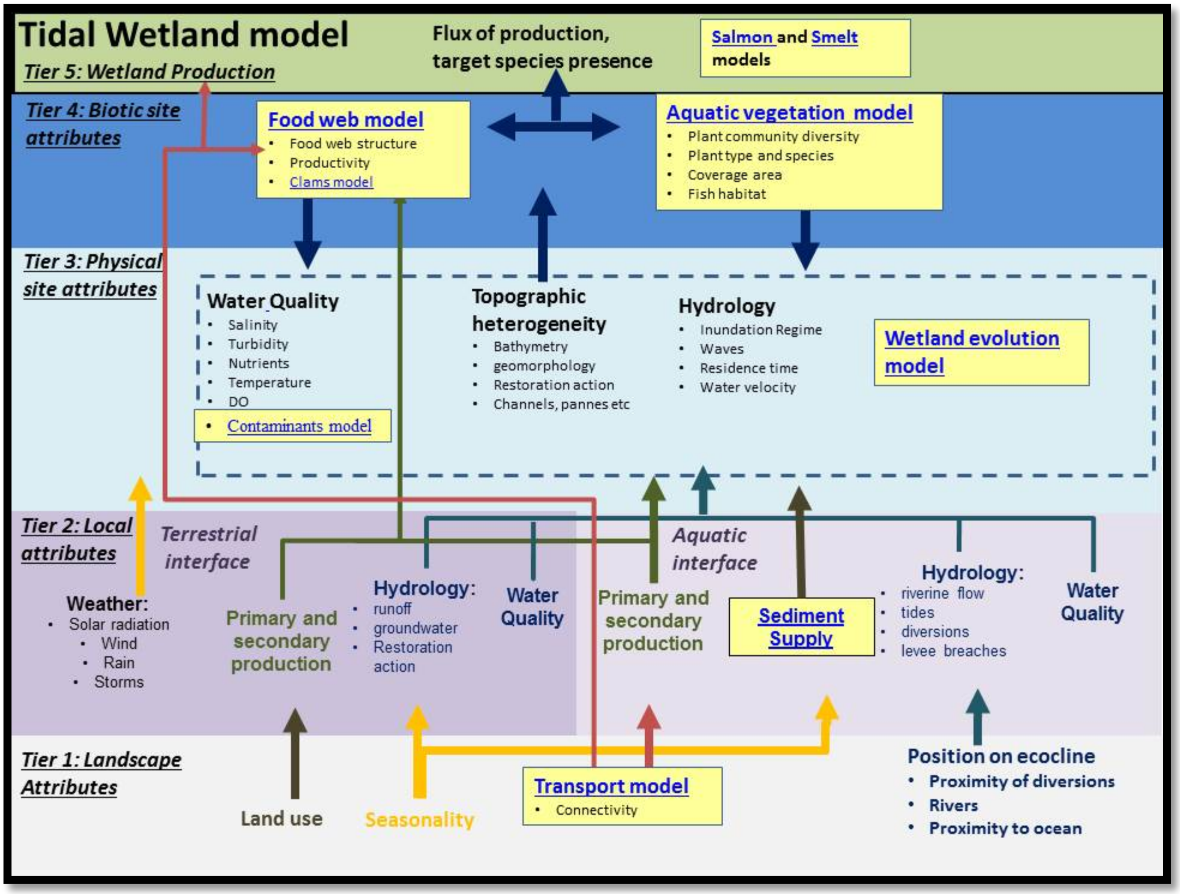
Our conceptual model is based on the "Tidal Marsh" element of the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP; Kneib et al. 2008) suite of conceptual models, which were designed to aid in identification of restoration actions. The DRERIP models characterize the dynamics of "equilibrium" marshes at their mature state of geomorphic and ecological functioning (Reed 2002; Williams and Orr 2002). The rationale in focusing on mature marshes was that the trajectories of change within actively restoring wetlands are variable and trying to capture intermediate stages of development (e.g., positions along a development "trajectory"; [(Simenstad and Thom 1996) however, see (Zedler and Callaway 1999)] would introduce too much variability for a single model. Further, the ultimate objective of restoration is the self-sustaining, equilibrium condition. However, there is now a pressing need for models of wetland evolution with an emphasis on immediate and ongoing impacts on listed fish species, in light of restoration required by the US Fish and Wildlife Service and National Marine Fisheries Service Biological Opinions for the SWP and CVP operations. Therefore, in our conceptual model we have sought to put greater emphasis on processes that influence restoration trajectories and affect ecosystem responses, and especially those of fish, in various landscape settings. Because these relationships are complex and difficult to portray in simple illustrations, we also reference sub-models which provide more detailed explanations of the hypothesized relationships between model components. For a more thorough discussion of wetland evolution and succession, see these sub-models, as well as the Suisun Marsh conceptual models developed as part of the Suisun Marsh Habitat Management, Restoration and Preservation Plan (Engle et al. 2010; Raabe et al. 2010; Siegel et al. 2010a; Siegel et al. 2010b)

As in the DRERIP tidal marsh model, we emphasize interactions between aspects of the habitat that change over different time scales (e.g. elevation changes over the course of several years vs. temperature changes over the tidal cycle) and consider factors operating at local (restoration site and adjacent areas) and regional scales (Zeff 1999; Enright et al. 2013). The interactions of these factors support the desired ecosystem functions included in tidal restoration planning: creation of habitat for at-risk fish species (e.g., Visintainer et al. 2006; Sommer and Mejia 2013), subsidies to their food web (e.g., Kneib 2003; Cohen et al. 2014, but see Lehman et al. 2010), and resulting increases in population stability and resilience to stressors (Herbold et al. 2014).

Heterogeneity of habitat types within a wetland is also important in this model, as habitat interfaces *within* a wetland mediate local biogeochemical processes between water, sediments, and nutrients. These within-wetland interfaces between distinct habitat types are also the environment sensed by individuals of our species of interest, and are actively transforming material and energy within the wetland (e.g. Rozas et al. 1988; Allen et al. 2013). Furthermore, heterogeneity may be crucial to recovery of native species within this historically highly variable estuary (e.g. Dean et al. 2005; Moyle et al. 2010).

Because knowledge of the details of the physical, chemical, and biological interactions operating within wetlands is still developing, the model and submodels are not comprehensive, but highlight processes most important for planning and monitoring of tidal restoration projects. Such monitoring will be important to determining the success of restoration and will contribute to our understanding of processes which will presumably result in better planning for subsequent restoration.

Figure 1. The Tidal Wetland Overview Conceptual Model diagram.



Model structure

The influences on tidal wetland function are a combination of attributes acting at the landscape, local, and wetland-specific scales. Therefore, we have organized this model into a series of "tiers" moving from the landscape to the site scale. The sub-models which have been developed in concert with this general wetland model are organized on a series of similar tiers going from the landscape level to proximate and ultimate drivers of the model outcome. While the sub-models are structured similarly, they may have different tiers depending on model subject.

This model and all its sub-models share several conventions for the accompanying diagrams. Attributes surrounded with dotted lines interact strongly. Arrows represent specific, describable linkages or relationships between attributes, though not all potential linkages are represented by arrows. Thickness of arrows generally describes hypothesized importance; however, arrows do not indicate certainty or strength of effect, as is the convention in the DRERIP models. Thick arrows moving between tiers that do not connect to specific factors represent relationships with all attributes in the following tier. Arrow and text color is used to aid in following arrows and does not convey meaning. Yellow boxes indicate links to sub-models which provide greater elaboration on the specifics of particularly important processes, structures, or species.

For the general tidal wetland model, there are five tiers: 1) Landscape Attributes, which are factors acting on the scale of the entire San Francisco Bay and Central Valley ecoregions, 2) Local Attributes, which are factors acting in areas directly adjacent to the focal wetland (on the scale of 1-10km; this tier is sub-divided into factors that act over the terrestrial interface and the aquatic interface), 3) Physical Site Attributes, which are physical and hydrologic properties and processes within the focal wetland, 4) Biotic Site Attributes, which are food web and vegetation structure within the wetland, and 5) Wetland Production, the “outcome” tier which describes the exported primary and secondary production and increased population support for target species.

Tier 1: Landscape Attributes

Position on the ecocline:

The position of an individual wetland patch within the estuarine gradient, or ecocline –distance from the ocean, nearest river, proximity and connectivity to other tidal wetlands, proximity to water control structures, and local land use patterns – is a major determinant of fundamental wetland characteristics (Lopez et al. 2006). Thus, no two wetlands are exactly alike, but all are influenced by dynamic forces working on the landscape scale. High natural variability between wetland sites and landscape-scale forcing can complicate interpretation of tidal wetland restoration results if reference sites are not considered for comparison. Reference sites need not be “pristine,” and can even be other restoration sites at different stages of development (Osenberg et al. 2006). This definition of “reference” has been used to assess restoration effects in systems as diverse as the Lower Columbia River Estuary (Borde et al. 2012), coastal Louisiana (Coastwide Reference Monitoring System: http://lacoast.gov/crms2/home.aspx), and the Anacostia River, which runs through Washington D.C. (Hammerschlag et al. 2005). Using consistent monitoring metrics on many sites throughout a region provides essential context in understanding a restoring wetland, and also has the potential to illuminate restoration effects on broader ecosystem function (Neckles et al. 2002). A levels-of-evidence or weight-of-evidence approach, such as that commonly used in ecotoxicology, has been shown to aid assessment of cumulative and potentially synergistic effects of multiple wetland restoration projects, but requires monitoring at many reference sites and explicit consideration of regional geography and hydrology (Diefenderfer et al. 2011; Golet et al. 2013).

The regional hydrology is dominated by the major rivers feeding the estuary and the force of the tides from the ocean. Watersheds of the Sacramento and San Joaquin rivers collectively drain 45% of California's area and funnel that water through the Delta west toward the ocean. Currently, riverine inputs into the estuary comprise approximately 84% Sacramento River water from the north and about 13% San Joaquin River flow from the south, with the remainder coming from smaller rivers entering from the east (Jassby and Cloern 2000). Wetlands in the estuary vary in the riverine influence they experience not only as one moves from east to west, but also from north to south, giving two axes to the "ecocline" dictating major controls on hydrodynamics and geomorphology.

Wetlands in the north Delta interact with the larger volume of Sacramento River water, which carries most of the suspended sediment imported to the system (Wright and Shoellhamer 2005), as well as treated wastewater from the city of Sacramento - a major source of ammonium (Glibert et al 2014b). The mainstem and tributaries of the Sacramento River harbor the only currently accessible spawning habitat for threatened spring-run and endangered winter-run Chinook Salmon; thus, naturally produced (as opposed to hatchery produced, which are often trucked downstream to avoid the Delta) juveniles of these races must pass through the north Delta on their journey to the ocean (del Rosario et al. 2013). Year-round residence of Delta Smelt has also been documented in the North Delta, in the Cache Slough region, including the Sacramento Deep Water Ship Channel and in restoring wetlands of Liberty Island (Merz et al. 2011, Sommer and Meija 2013). This area is the terminus of the Yolo Bypass, a large and productive floodplain that is inundated in years of high precipitation, with documented benefits for rearing of Chinook Salmon (Sommer et al. 2001). East side tributaries (Cosumnes and Mokelumne Rivers) deliver generally small volumes of water to the northeast Delta. However, the flows are flashy; consequently, islands in the northeastern Delta, such as the McCormack Williamson Tract, have flooded frequently, providing the impetus and opportunity to construct multibenefit flood control and restoration projects at the interface of floodplains and tidal wetlands.

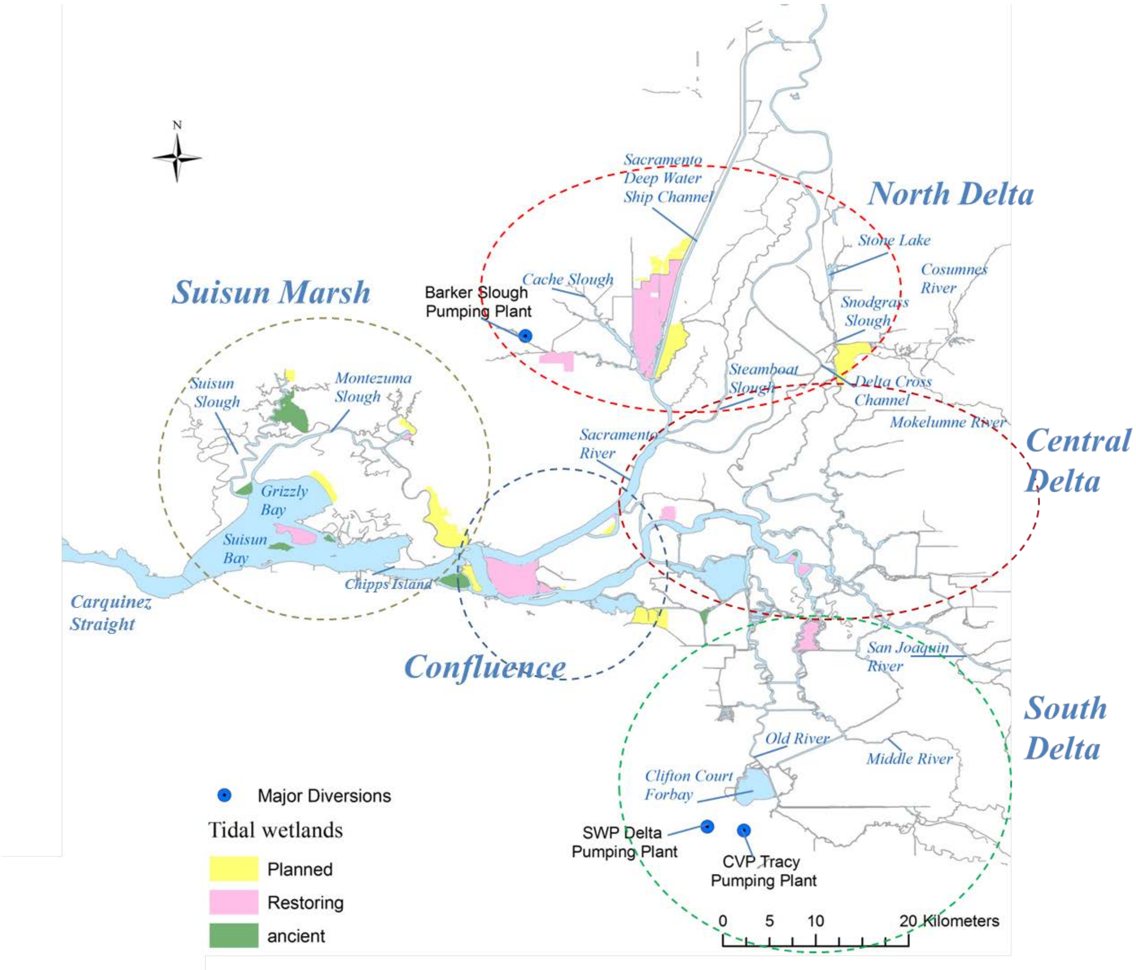
Compared to the Sacramento River, the San Joaquin River delivers a small volume of water to the south Delta, and that water generally carries less sediment and more chemical pollutants than Sacramento River water. During much of the year a large portion of the streamflow from the San Joaquin River consists of agricultural return flows from the vast expanses of agricultural land of the San Joaquin Valley. The large diversions of the CVP and SWP, along with their associated infrastructure, are located in the south Delta. These diversions not only entrain fish, but they also alter flow patterns within the estuary (Kimmerer 2004, 2008). During periods of high pumping rates relative to river inflow, net water movement is more from north to south than it would be naturally. This can confuse migratory fish, draw them into low-quality habitats, and reduce their survival. Delta Smelt especially are in danger of direct entrainment if they are in the south Delta while flows of Middle and Old Rivers are negative (movement “upstream”), because of their poor swimming ability (Swanson et al. 1998). The perils of the south-central Delta make this area less desirable as a target location for wetland restoration.

Wetlands in the west Delta, and through Suisun Marsh, are more directly influenced by the tides than the areas to the east and can encounter more variable salinities, depending on the volume of Delta outflow. The Low Salinity Zone (LSZ) is generally found in this region. Its exact location, and the area it covers, varies considerably depending on freshwater inflow. A metric commonly used to indicate the position of the LSZ is X2, the daily average distance of the 2psu isohaline (measured at the bottom of the water column) from the Golden Gate Bridge (Jassby et al. 1995). X2 is correlated with the estuarine turbidity maximum as well as with the occurrence or abundance of several aquatic species (Jassby et al. 1995; Kimmerer 2002a,b; Kimmerer et al. 2009). Salinity also is a major determinant of plant productivity and community composition. Because ecosystem engineering abilities differ between plant species, salinity is an indirect driver of marsh evolution (Engels and Jensen 2010; Schile et al. 2014). See the Wetland evolution and Aquatic Vegetation models for more information.

In addition to position relative to tidal and riverine influences, a wetland restoration site’s proximity to diversions will play a role in its utility to at-risk fishes. The effects of the CVP and SWP diversions are discussed above, but while they are the largest, they are far from the only diversions in the Delta. Smaller intakes draw drinking water for the city of Stockton, North Delta, and Contra Costa County. A multitude of small and medium-size agricultural diversions (> 2000 siphons, Jassby 2005) are scattered throughout the Delta. Particularly if unscreened, these diversions may pose an entrainment risk for young fishes in the vicinity, but the risk varies on a diel basis and with habitat use (Nobriga et al. 2004). The impact of small diversions on fish access to a restoration site is likely to vary greatly depending on local circumstances.

The restoration of a tidal wetland site will clearly be influenced by its position in the landscape; conversely, these “new” habitats can also affect adjacent areas, both physically and ecologically. For example, tidal energy is one of the main factors driving productivity in tidal wetland ecosystems, but extensive tidal wetland restoration may alter the tidal energy in a specific location by spreading it over a greater area (Herbold et al. 2014). Breaching of levees is also likely to influence flow dynamics, and thus material trapping or dispersal, at nearby channel junctions (Gleichauf et al. 2014). New communities present within restoration sites may provide modes to export productivity or propagules to the food web in the surrounding channels through hydrodynamic transport or trophic relay (Kneib 1997). The extent of the subsidy from restoration sites will depend on how the material is transported off site (see below and Transport Model).

Figure 2. The Sacramento-San Joaquin Delta and Suisun Marsh.



Link to the Transport Model

Connectivity/Transport:

Particularly when the goals of tidal wetlands restoration include export of productivity and/or providing nursery habitat for migratory fishes, consideration of connectivity is critical to the design and assessment of restoration projects (Beck et al. 2001, Howe and Simenstad 2014). Restoration designs featuring high hydraulic connectivity have been shown to facilitate exchanges of organic material, leading to faster attainment of “functional equivalency” with more mature wetlands than restoration projects with more limited connectivity (Howe and Simenstad 2014). Additionally, wetlands can only be useful as habitat for at-risk fish species if they can access it, and if used as nursery habitat, leave to recruit to the adult population. Particularly for juvenile salmon, the placement of many productive, refuge-offering wetlands along their seaward migration route may increase survival and encourage development of life history diversity. The size and placement of entrances to a restoring wetland are also crucial to promoting fish use of the site (Cordell et al. 2011).

The physical evolution of a restoring wetland can also depend on site placement and access. Delivery of mineral sediments from upstream sources clearly is a driver of evolution. Plant community composition and subsequent vegetation-driven ecosystem engineering is also influenced by the species extant in areas surrounding and with connectivity to the restoration site (Neff et al. 2009; Diggory and Parker 2011; Leck 2013; Schile et al. 2014).

The distance that materials can travel from one wetland to another, and the ability of wetlands to subsidize the food web of pelagic waters depends on tidal excursion, freshwater flow, size of intervening channel(s), transformations of the material between the source and target site, and site geomorphology. See the transport model for details.

Land use:

Current and past activities within the estuary and its tributary watersheds profoundly affect the characteristics of estuarine soils and water. Land use within the Delta is dominated by crop agriculture. Coordinated reclamation of peat-rich lands for agriculture began in 1869 and was complete by 1930 (Whipple et al. 2012). Subsequent soil oxidation, erosion, burning, and compaction has led to land subsidence to elevations more than 8m below mean sea level in some locations (Deverel and Leighton 2010; Mount and Twiss 2005).The cumulative impact of subsidence has severely reduced the area of land at appropriate intertidal depths for restoration, particularly in the central Delta (Mount and Twiss 2005). Reduced sediment supply imported from the estuary’s watersheds exacerbates the problem of subsided baseline conditions in many would-be restoration sites, as it may place limits on the ability of wetlands to accrete enough sediment for vertical growth to intertidal elevations, and ultimately to keep pace with sea level rise (Schoellhamer 2011).

In contrast to the Delta, land use is Suisun Marsh is dominated by rangeland and non-tidal or muted tidal wetlands managed to attract waterfowl (“duck clubs”), which may affect nutrient inputs and dissolved oxygen in the surrounding sloughs and wetlands. The Delta and Suisun Marsh are also widely used by recreational boaters, hunters, and fishermen, so land use most proximate to potential restoration sites could include marinas, campgrounds, and public boat ramps. There are also unique rural communities within both the Delta and the Marsh, and some larger urban and industrial centers (e.g. Sacramento, Stockton, Pittsburg and Antioch) on its periphery. The estuary encompasses the pathways of large commercial shipping vessels to the ports of Stockton and Sacramento.

Urbanization and agriculture within, upstream of, and surrounding the estuary contribute man-made chemical contaminants, metals, nutrients, and concentrated salts via run-off, agricultural irrigation return ditches, untreated stormwater drainage, and waste water treatment plant discharge. Some contaminants are no longer used in large quantities, but are a legacy of past industrial or agricultural applications. Legacy mercury contamination from gold-rush era hydraulic mining is a particular concern in restoration because of the potential for restoration-mediated methylation and subsequent bioaccumulation and biomagnification in the food web and health impacts for people who eat SFE wildlife (Henneberry et al. 2012). Understanding the implications of current and past land uses that may be affecting a wetland site is critical to effectively and efficiently monitoring for contaminants (see contaminant sub-model).

Seasonality:

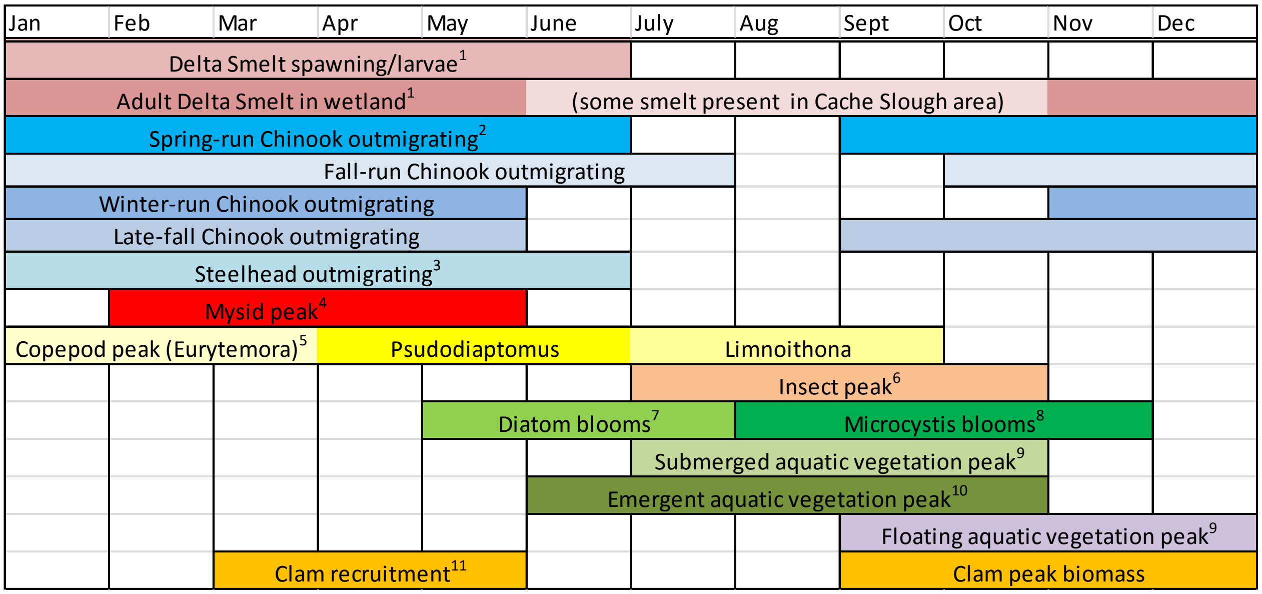
Due to California's Mediterranean climate, riverine inputs to the upper estuary vary greatly by season. Mean freshwater inflows average 1700 +/- 300 m^3/s during the cool, wet winters, but only 540 +/-40 m^3/s during the hot dry summers (Jassby and Cloern 2000). Large flood pulses associated with winter storms and the spring snowmelt will have strong impacts on sediment transport, as well as extent of floodplain inundation and resulting allochthonous inputs to the Delta. High flows provide sediment and nutrient recharge, reduce establishment of invasive aquatic vegetation, and provide greater connectivity for nekton to enter wetland habitats.

The strongly seasonal climate has major impacts on species' life histories and ecosystem productivity within the SFE (Figure 3). Larval fish assemblages and proportion of natives versus alien larvae in the estuary shift seasonally, as would be expected given the spawning habits of the adults (Feyrer 2004; Sommer et al. 2004). Migratory individuals of the species of most concern in this model tend to mobilize in concert with weather and/or flow events. For example, those Delta Smelt that move from the low salinity zone into fresh water for spawning (as opposed to freshwater residents) appear to begin their migration during the "first flush" of high turbidity water that accompanies early winter/late fall storms (Grimaldo et al. 2009; Sommer et al. 2011). Longfin Smelt migrate from the ocean or San Francisco Bay into the estuary during late fall/early winter (Moyle 2002; Fish et al. 2009). Chinook Salmon also time their migrations by water flows; the largest spawning run occurs in fall and winter (Moyle 2002). During the summer, most young-of-the-year Delta Smelt have moved into the Low Salinity Zone of the estuary and Longfin Smelt have moved downstream into brackish water or to the ocean (Fish et al. 2009). Meanwhile, changes in wetland primary productivity accompany seasonal changes in temperature and hydrology.

During lower average flows common in the summer, higher residence times and shallower water depths allow greater phytoplankton production (Lopez et al. 2006; Lucas and Thompson 2012). Summer is also the season of greater macrophyte growth due to longer day lengths. The macrophytes primarily considered here and in the Aquatic Vegetation conceptual model are herbaceous aquatic or wetland species including emergent vegetation, such as tules (*Schoenoplectus spp*.), floating aquatic vegetation (FAV), like water hyacinth (*Eichhornia crassipes*), and submerged aquatic vegetation (SAV), many of which are non-native and considered pests, as is Brazilian waterweed (*Egeria densa*).While macrophytes provide relatively little immediately bio-available organic carbon when compared to phytoplankton, they do contribute to the food web via detrital pathways and are important in providing structural heterogeneity and supporting physical processes of wetland evolution (see Aquatic Vegetation and Wetland Evolution models). The evapotranspiration rate of the vegetation correlates with temperature and day length, so within-Delta water consumption is also strongly seasonal (Jassby and Cloern 2000). However, increased salinity during the low-flow summer months may decrease productivity of certain plant species in the low-salinity zone of the estuary (Janousek and Mayo 2013). Decomposition occurs at a faster rate in warmer summer temperatures, though rates of peat decomposition are highly dependent on salinity, soil oxygen content, and nutrient content (Mendelssohn et al. 1999).

Interannual variability adds to seasonality to control habitat heterogeneity, diversity, and productivity. Due to the high natural year-to-year variability and frequency of extended droughts, some native fish populations have evolved high physiological tolerances for stressful conditions (Moyle and Light 1996). Human modifications of the natural flow regime have reduced both annual and inter-annual variability in flow, effectively flattening the hydrograph of the system, which favors many of the non-native species; the success of native species is favored by intermittent high-flow events (Brown and Bauer 2010; Kiernan et al. 2012).

Figure 3. Seasonality of organisms in the upper San Francisco Estuary.



Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec
Delta Smelt spawning/larvae[1]	■	■	■	■	■						
Adult Delta Smelt in wetland[1]	■	■	■	■	(some smelt present in Cache Slough area)	■	■	■	■	■	■
Spring-run Chinook outmigrating[2]	■	■	■	■	■			■	■	■	■
Fall-run Chinook outmigrating	■	■	■	■	■	■			■	■	■
Winter-run Chinook outmigrating	■	■	■	■						■	■
Late-fall Chinook outmigrating	■	■	■	■				■	■	■	■
Steelhead outmigrating[3]	■	■	■	■	■						
	Mysid peak[4]	■	■	■							
Copepod peak (Eurytemora)[5]	■	■	Psudodiaptomus	■	■	Limnoithona	■	■			
						Insect peak[6]	■	■	■		
				Diatom blooms[7]	■	■	Microcystis blooms[8]	■	■	■	
						Submerged aquatic vegetation peak[9]	■	■	■		
					Emergent aquatic vegetation peak[10]	■	■	■	■		
								Floating aquatic vegetation peak[9]	■	■	■
		Clam recruitment[11]	■	■				Clam peak biomass	■	■	■

1. (Baxter et al 2015, see Delta Smelt Model)
2. (Yoshiyama et al 1998, and Kevin Reece (DWR), pers. comm, see Salmon Model)
3. (Moyle 2002)
4. (Hennessy and Enderlien 2013, M. Young, UC Davis pers. comm, see food web model)
5. (Hennessy and Enderlien 2013. Bollens et al 2014, see food web model)
6. (Howe et al 2014, M. Young, UC Davis, pers. comm. , see food web model)
7. (Gilbert et al 2014)
8. (Lehman et al 2015)
9. (Hestir et al 2008, see Aquatic Vegetation Model)
10. (Miller and Fujii 2010, see Aquatic Vegetation Model)
11. Corbicula and Potamocorbula (Thompson and Parchaso, 2012, 2013, see Clam Model)

Tier 2: Local Attributes

Interfaces between tidal wetlands and adjacent upland and aquatic environments:

The attributes that we describe as “local” include the physical, chemical, and biological characteristics of the water and lands in the immediate vicinity of the restoration site, as well as the weather experienced by the site. These characteristics will vary within the context of the broader landscape attributes, and in turn will interact with conditions on the restoration site via flux of materials and energy across interfaces between the wetland and surrounding terrestrial and aquatic habitats. Flux includes movement largely controlled by water flow (dissolved substances, plankton and particulate matter), as well as active movement of materials and energy via terrestrial and aquatic animal movement (Allen et al. 2013; Morgan-King and Schoellhamer 2013; Howe et al. 2014). The nature of interfaces between adjoining habitats depends on the particular site, and therefore interfaces restrict communication between the site and its surroundings to varying degrees (Williams and Orr 2002). For example, quantity and quality of runoff entering a tidal wetland bordered by a wide swath of vegetated upland would be quite different than runoff entering a wetland mostly surrounded by steep armored levees (Hart and Hunter 2004).

Restoration action may change these interfaces, and therefore change how physical and biological drivers affect wetland productivity. The breach(es) in a formerly diked area represents the main interface between the aquatic ecosystem and a newly restored wetland. The number of breaches in a given site, the size and depth of the breaches, and the position of these breaches all influence internal hydrology and the effect of external hydrology on the restoration site (Williams and Orr 2002). In restoration of a diked area, levee banks are the main terrestrial interface, and slope of levees, type of banks on levees, and vegetation on the banks all contribute to the way inputs from the terrestrial system affect the wetland (Talley et al. 2006). Slope of the terrestrial interface is not only critical to terrestrial inputs; it also determines how resilient the site is to future sea level rise (Stralberg et al. 2011; Schile et al. 2014). Managers have control over some of these components during construction of a restoration project, and they are among the most important decisions to make when designing a restoration site.

Because tidal wetlands are transition zones, their structure and function is highly dependent on drivers acting on the system via the aquatic and terrestrial interfaces. The main aquatic inputs considered here are: 1) instream hydrology (tides, freshwater input, and diversions), 2) water quality (nutrients, pollutants, DO, temperature, etc), 3) imported nekton and productivity, and 4) sediment supply. The inputs from terrestrial and aerial interfaces have an equally strong, though often overlooked, effect on wetland structure and function. Terrestrial inputs are chiefly 1) hydrology (groundwater, runoff, and rain), 2) water quality, 3) weather, and 4) terrestrial production. While these inputs interact in the surrounding environment before entering the wetland, this model does not discuss how they are linked in depth and instead focuses on how their final form affects the internal dynamics of the wetland.

Inputs via the aquatic interface:

Link to the Transport model

External Hydrology:

Riverine flows deliver sediment, chemical constituents (e.g., nutrients, contaminants), and living organisms to downstream wetlands, as discussed above, but in a tidal system with variable bathymetry and numerous channel junctions, dispersion and transport of materials and organisms through the estuary is not a simple one-way flow (Gleichauf et al. 2014). Tides provide key exchange between the vegetated marsh complex and surrounding open-water habitat. The interaction of tidal “sloshing” with wetland geomorphology is a major determinant of the fate of plankton produced on the wetland, and whether a given habitat will function as a sink or source of productivity to surrounding channels (Lucas et al. 2002).

The local hydrology at aquatic interfaces strongly influences wetland evolution within a restoration site and has the potential to regulate animal use of the site (Mitsch and Gosselink 1993). Conditions in the channel adjacent to a wetland entrance (i.e., a levee breach) could prevent or discourage at-risk fishes from accessing the site. These conditions include excessive velocities, scour holes used as havens by predators, near-by water diversions, and turbidity lower than species preference (especially for Delta Smelt). Conversely, external hydrology may maintain conditions amenable to wetland use, as is the case with current conditions at Liberty Island in the Cache Slough complex. High turbidity there is sustained by wind-waves, limited tidal excursion, and tidal asymmetry (Morgan-King and Schoellhamer 2013).

The occurrence and magnitude of flood events, especially when coincident with major tidal exchanges, are of particular importance from the standpoint of contributions of sediments, nutrients and some organisms, and physical disturbance processes that influence wetland geomorphology (Friedrichs and Perry 2001; Oh et al. 2013). Both seasonal variation (floods) and inter-annual variability in freshwater inputs are particularly important for native fish in comparison to invasive warm-water specialists such as Largemouth Bass and catfish (Feyrer and Healey 2003; Feyrer 2004; Grimaldo et al. 2004; Greene and Beamer 2011).

Water quality:

Nutrients and pollutants, as well as sediments and a variety of other water constituents enter a wetland via the aquatic interface. Within-wetland processes will greatly change many of these water characteristics (temperature, dissolved oxygen, turbidity, and chemical loads; see below) but external source loading of nutrients, pollutants, or water low in dissolved oxygen have the capacity to greatly affect the wetland. For example, managed wetlands draining into a restored wetland may contain extremely high loads of decomposing organic carbon and low dissolved oxygen (Barthman-Thompson et al.). Upstream urban wastewater inputs also alter abundance and form of nutrient input (e.g. ammonium vs. nitrate) which can influence the types of producers within the wetland, with undesirables such as the cyanobacterium *Microcystis* potentially favored with higher ammonium concentrations (Lehman et al. 2015). Excess ammonium and water-borne pesticides also can have toxic effects on consumers and at-risk species (Connon et al. 2011). See the food web and contaminants models for further detail.

Imported primary and secondary production:

While a tidal wetland may or may not be a *net* exporter of productivity, tides and freshwater inflow will import a significant amount of both primary production (phytoplankton, dissolved organic material, and detritus), and secondary production (zooplankton, emergent insects, and nekton) (Dean et al. 2005; Lehman et al. 2010; Boucek and Rehage 2013). Import of nekton is particularly relevant for species such as Chinook Salmon that use the wetland as rearing habitat but migrate to other areas to complete their life cycle (Bottom et al. 2005; Greene and Beamer 2011). One study of organic matter sources in the estuary estimated that tributaries annually contributed organic matter, including phytoplankton, detritus, and dissolved organic carbon, at five times the amount of all autochthonous inputs combined; however, tributary contribution varies seasonally and was greatly reduced overall in drought years (Jassby and Cloern 2000). This tributary input was higher from the Sacramento watershed (41-52% of total allochthonous carbon) than the San Joaquin due to the higher flow in the Sacramento River, but San Joaquin input was still relatively high (20-42%) due to longer residence times and greater percentage of agriculture (Jassby and Cloern 2000), also see Land Use under Landscape attributes). Transport of production between wetlands is also accomplished via aquatic interfaces. Howe and Simenstad (2007) demonstrated the ability of adjacent wetlands to exchange organic material, and suggest that food resources of newly restored wetlands increase more quickly when proximate to more mature wetlands.

Imported sediments:

Link to the DRERIP sediment model and Wetland Evolution Model

Water flux can also transport sediment, which is critical to tidal wetland restoration and to resilience of marshes to sea level rise (Brand et al. 2012; Schile et al. 2014). Sediment can accrete over time, raising the elevation of marsh plains and potentially creating new intertidal marsh habitat in currently subtidal areas. In subsided islands restored to wetlands the situation is less clear, and whether subsided islands will be accretional or erosional depends on local conditions (see physical process below, and Wetland Evolution model) (Morgan-King and Schoellhamer 2013). The chief source of sediment influx into the wetland system is high riverine input during storm events (Friedrichs and Perry 2001; Vandenbruwaene et al. 2011). The degree of complexity of the channel – wetland interfaces across which water-borne sediment flows greatly influences the potential for accretion. There may also be a smaller sediment influx via the terrestrial interface (not shown) if the surrounding landscape is highly erodible. Sediments play a key role in relative elevation shaping processes within the wetland, as well as contributing to turbidity and the diversity and productivity of the vegetation community.

Inputs via the terrestrial interface:

Hydrology:

Freshwater can enter the system from the terrestrial interface as well as the aquatic. Seasonal precipitation falls directly on the wetland through the aerial interface, and surface flow, groundwater, stormwater, and agricultural run-off also contribute fresh water to the system. Terrestrial inputs can be particularly important in regions such as the north Delta, which sits at the base of the extensive seasonal Yolo Bypass floodplain (Sommer et al. 2004).

Water quality:

Link to the contaminants model

While hydrologic input from the terrestrial environment into the Delta was historically fresh, diversions for agriculture may concentrate salts naturally found in ground water (Quinn 2014). Before water reaches the Delta, it may have been reused multiple times, greatly concentrating any dissolved materials such as nutrients, salts, heavy metals, and pollutants through evaporation (Krupa et al. 2011). Therefore, in some circumstances, terrestrial inputs may have a greater effect on wetland water quality than tidal inputs.

Terrestrial inputs may also carry more nutrient inputs across the aquatic interface because they have often not been diluted by freshwater flows from other sources. While excessive nutrient enrichment has historically not caused eutrophication in the Bay-Delta estuary to the degree that it has in other estuaries (Cloern 2001), changes to nutrient stoichiometric ratios caused by agricultural runoff and urban wastewater have been implicated in changes to the aquatic food web (Glibert et al. 2011;Glibert et al. 2014b; Lehman et al. 2015).

Weather:

Wind:

Wind is the driving factor behind wave formation, vertical mixing of the water column, erosion of sediment, sediment re-suspension, and water column turbidity (Williams and Orr 2002; Tonelli et al. 2010; Marani et al. 2011). An expanse of open water aligned in the direction of prevailing or storm winds can result in significant wave effects on the morphology of a wetland site. The greater the expanse of open water (fetch), the greater the wind-driven waves will be and corresponding erosion and sediment resuspension. For example, the eastern levee of the southern part of Liberty Island is much more degraded than the western side due to wind influence on the large fetch of the southern shallow open waters.

Rain:

Rain falling directly on the wetland may have some effect on local water quality and salinity, but the larger effect of rain and storms comes through freshwater flows resulting from precipitation upstream of the Delta. The first storms of the water year not only mobilize sediments and other water constituents (such as pesticides, Weston et al. 2014), but the larger flows of turbid water also cue movements of Delta Smelt (Sommer et al. 2011), salmonids, and other native species (Sommer et al. 2014) through the estuary.

Solar radiation*:*

All primary production depends on solar radiation; however, the availability of light fueling primary productivity will depend on physical factors affecting attenuation. High turbidity and shading of other producers by emergent or floating macrophytes can reduce photosynthetic rates of submerged plants, macroalgae, and benthic and planktonic microalgae. The effect of reduced light penetration on production in the wetland depends on species composition and the photosynthetic optima of the producers within the community.

Terrestrial primary and secondary production:

While this model is specifically designed to inform restoration outcome for aquatic species, numerous tidal wetland restoration projects have been undertaken in the Bay-Delta Estuary for avian and terrestrial species of concern. These include the salt-marsh harvest mouse (*Reithrodontomys raviventris*), Ridgeway's rail (*Rallus obsoletus obsoletus*, formerly California clapper rail), and other wetland-dependent birds (Brand et al. 2011; Brand et al. 2012; Zhang and Gorelick 2014). These terrestrial animals benefit from many of the same wetland features as aquatic organisms, and may affect wetland function themselves. Birds in particular may provide important controls on wetland food webs; for example, diving ducks have been described decimating populations of invasive clams (Poulton et al. 2002), and grazing by geese may prevent vegetation establishment (Simenstad et al. 2005).

Terrestrial detritus also contributes to wetland ecosystems, especially during flood events that wash nutrients, dissolved organic carbon, and particulate organic matter into the aquatic environment (Oh et al. 2013).

Tier 3: Physical Site Attributes

Within the wetland, the external drivers combine with the restoration action to influence the physical structure and processes of the site. The major components of this physical structure are: 1) Internal hydrology (waves, residence time, and inundation regime), 2) Wetland Evolution (changes in relative surface elevation through accretion, erosion, desiccation, and sedimentation), 3) Topographic Heterogeneity (complex horizontal structure comprised of channels, pannes, and vegetation), and 4) internal water quality parameters (DO, nutrients, pollutants, temperature, turbidity etc).

Internal Hydrology:

Movement of water within a wetland and the occurrence of erosional and depositional features are among the architects of wetland structure. High velocities at breaches and on the outside of creek meanders can carve pools, while slower velocities encourage deposition. Waves can also be a major force in shaping wetland morphology (See wetland evolution model for more). Areas subject to a long fetch length and at least seasonally strong winds can limit vegetation colonization and restrict the area of intertidal elevations (Williams and Orr 2002).

Wetland surface elevation interacts with tides to produce an inundation regime. Inundation regime is also a dominant determinant of emergent vegetation distribution and productivity (Janousek and Mayo 2013, Schile et al. 2014). Similarly, phytoplankton production and biomass are strongly influenced by the combination of tides, river flow, and wetland geomorphology as they define water residence time. Shallow waters with long residence times should promote water column productivity by providing time for phytoplankton to access relatively high nutrient levels in depths that reduce light limitation. Tidal asymmetry then has the potential to govern whether the accumulated production is transported out of the wetland (Lehman et al. 2015b). However, the presence of invasive bivalves can greatly offset production (Lucas and Thompson 2012). Slower flow associated with long residence times can also favor other invasive species, such as *Egeria densa*, which harbors mobile grazers that further reduce phytoplankton biomass (Yarrow et al. 2009). One challenge in designing tidal wetland restoration plans is to balance encouraging phytoplankton productivity while simultaneously discouraging colonization by invasive species; this may be best accomplished with a heterogeneous wetland that allows for mixed residence times.

Wetland Evolution:

Link to the wetland evolution model

Restoration of full tidal influence to a site will begin the process of wetland evolution. Though channels may be dredged, vegetation planted, and/or berms and hummocks built prior to a levee breach, a wetland restoration site will not look the same “as built” several years later; hydrology, sea level, weather, and biology will interact to continually change the wetland geomorphology and biotic community. It is expected that geomorphic change will be rapid early in the process and may be greatly influenced by storm events, but the rate of change will decrease after several years, requiring less frequent monitoring of attributes such as bathymetry and channel structure.

As described in the Wetland Evolution sub-model, “baseline conditions” are fundamental determinants of the eventual form taken by a restoring wetland. Initial surface elevation is particularly important in planning restoration actions because much of the Delta is currently comprised of highly-subsided islands. Levee breaches for restoration on these islands may lead to formation of homogeneous subtidal lakes rather than tidal wetlands if vertical accretion does not occur rapidly (Mount and Twiss 2005). In cases of deep subsidence, additional actions to increase elevation may be required before breaching; e.g., deposition of sediment dredged from surrounding channels, or using emergent vegetation along with a managed or muted tidal hydrology to build peat elevations (Beauchard et al. 2011). Accretion within restored wetlands is highly variable, so it is difficult to predict the long-term effect of tidal restoration on relative surface elevation (Brand et al. 2012).

Increase in wetland elevations occurs through a combination of allochthonous sediment deposition and organic matter accumulation. Allochthonous sediment supply may be limiting in some situations, and will vary with site position in the estuary as well as with water year. Wetland plants contribute to vertical accretion both indirectly by trapping sediment as well as directly by contributing organic matter to peat soil formation (Drexler 2011; Temmerman et al. 2005). Models of possible climate change scenarios suggest accretion may increase the long-term sustainability of marsh plains during sea-level rise if they have adequate sediment and organic input (including internal peat formation), though this is only possible if the terrestrial interface is adequately gradual in slope, allowing inland migration (Stralberg et al. 2011).

Wetland elevations in general can decrease due to microbial oxidation and consolidation of soil organic matter when exposed to air. Wind erosion and burning were prominent drivers of historical subsidence, but have decreased with changing land management practices (Deverel and Leighton 2010). The relative interaction of processes that increase and decrease relative elevation of the wetland has been described in more detail in the Wetland Evolution Model as well as in the Suisun Marsh Relative Surface Conceptual Model (Siegel et al. 2010).

Topographic Heterogeneity:

The elevation-shaping processes described above interact with vegetation structure and hydrologic processes that may result in a complex network of intertidal channels, pannes, ponds, and/or vegetated areas (Morzaria-Luna et al. 2004). A complex arrangement of features across the wetland and at both the aquatic and terrestrial interfaces results in higher biodiversity through development of a variety of habitat types (Herbold et al. 2014). Intertidal creek order and form have implications for nekton species composition and abundance as well as movement and feeding behavior, with different species exhibiting different geomorphic preferences (Moyle et al. 1986; Allen et al. 2007; Visintainer et al. 2006; Williams and Zedler 1999). In the SFE, across the salinity gradient, tidal channel size is associated with particular fish assemblages (Gewant and Bollens 2012). An abundance of edge habitat may increase foraging success for fishes, and should be a design consideration for restoration sites, along with tidal connectivity (Gewant and Bollens 2012). A complex network of wetland channels also affects internal hydrology by providing mixed residence times; for example, dead-end sloughs may have relatively long residence times, allowing high productivity, while larger channels and breaches may have lower residence times, allowing higher DO and turbidity. Differences in the morphology of even similarly-sized channels (e.g. leveed versus more natural cross-section) can also interact with tides and weather to produce disparate flow and temperature dynamics between them (Enright et al. 2013).

Water Quality:

Dissolved Oxygen: The Delta is generally shallow and well-mixed, and the lack of vertical stratification (Kimmerer 2004) precludes persistent hypoxia in much of the area, with the exception of the Stockton Ship Channel on the most downstream reach of the San Joaquin River. Lack of dissolved oxygen there sets up when surface warming, long residence time, and high biological oxygen demand coincide (Jassby 2005). This can be exacerbated by water management activities, such as the removal of the Head of Old River Barrier, which increases water residence time in the channel (Monsen et al. 2007). Low dissolved oxygen is a noted barrier to salmonid migration through the Stockton Ship Channel area (Jassby 2005).

Localized hypoxia also occurs when waters from managed wetlands in Suisun Marsh are released into surrounding channels. Not only is the water low in dissolved oxygen, but it also has high concentrations of organic matter, and thus carries high biological oxygen demand and can cause fish kills. The effects are most pronounced in small dead-end sloughs (Barthman-Thompson et al. 2010).

Turbidity: Turbidity may have strong effects on productivity via its attenuating effect on solar radiation (May et al. 2003; Jassby et al. 2002). This water characteristic can also impact trophic interactions above the base of the food web. Turbid waters can act as cover for open water species, such as Delta Smelt, limiting predation by largemouth bass and other non-native fish predators (Ferrari et al 2014). Likewise, non-native fish that generally prefer zooplankton will prey on benthic macroinvertebrates in higher proportions under conditions of high turbidity (Howe et al. 2014). Laboratory studies of larval Delta Smelt indicate that at very high turbidities, their feeding is also inhibited; however, those occasions are rare in nature. Intermediate turbidity, coupled with adequate light appears to be beneficial to Delta Smelt as it may provide contrast for the larvae to see their planktonic prey and optimize feeding success (Baskerville-Bridges et al. 2004), as well as provide refuge from visual predators (Sommer and Mejia 2013).

Salinity: Though many estuarine species are tolerant of a range of salinities, some are more so than others, and each species has a particular range of tolerance, with an optimum for fitness within that range. Salinity, therefore, is a key determinant of the capacity of a wetland to support any particular species (see Food Web model for more information on salinity's effect on aquatic organisms). Salinity and its variability within a wetland site will be influenced primarily by the site's position on the ecocline (see Landscape, tier 1), but may change relatively quickly due to changes in riverine inputs or runoff.

Link to the Aquatic Vegetation Model

Generally, more saline marsh plains are characterized by a relatively low diversity of salt-tolerant emergent vascular plants. Many of these plants would grow better in fresher waters, but are excluded by a greater diversity of competitively superior plants. This classic zonation pattern has been described in many estuaries (e.g. Engels and Jensen 2010), including the San Francisco Bay – Delta estuary. In the latter estuary, a recent study extending from Bothin Marsh in the Bay proper to Sand Mound Slough in the Delta has documented a change in marsh plant distribution over the past several decades, with a "pervasive" expansion of salt tolerant species

(Watson and Byrne 2009). However, effects of increasing salinity were modulated by increasing inundation, as salts do not accumulate as rapidly with frequent tidal flushing. Sea level rise-induced salinity increases at a particular site will interact with elevation to determine the path of evolution in plant community structure (Watson and Byrne 2009). In the short term, macrophyte productivity is likely to be diminished by the interactive stress of greater inundation frequency and higher water salinity accompanying sea level rise (Janousek and Mayo 2013), but site productivity will ultimately depend on which species thrive in the new tidal/salinity regime (Watson and Byrne 2009).

Salinity gradients structure species distributions, and in some cases salinity modulates competitive interactions. For example, the invasive submerged aquatic species *Egeria densa* out-competes native *Stuckenia spp*. in fresh water, but *Stuckenia* prevails at higher salinities (Borgnis 2013). Similarly, the species of invasive Asian clam in a given area is correlated with salinity, with *Corbicula fluminea* dominating freshwater and *Portamocorbula amurensis* inhabiting more saline water, though this distribution is likely more related to the tolerance of early life stages than that of adults (Miller and Stillman 2013).

Nutrients: The low primary productivity that characterizes the SFE is not a consequence of generally low nutrient concentrations. Rather, other factors, including light limitation, more often control production in the estuary (Jassby et al. 2002). However, the relative abundance of nitrogen species (ammonia/um and nitrate) has been related to the composition of the pelagic producer community as well as to occasional spring algal blooms under certain conditions (Dugdale et al. 2007; Glibert et al. 2014a,b; Lehman et al. 2015). While nitrification of the plethora of ammonium contributed to the estuary via Sacramento wastewater effluent has apparently led to beneficial diatom blooms (Glibert et al. 2014a), high levels of ammonium may favor blooms of the toxin-producing cyanobacterium *Microcystis* (Glibert et al. 2014b; Lehman et al. 2015). Laboratory studies have shown adverse effects of dietary exposure to these cyanotoxins on the health of SFE Threadfin Shad and Splittail (Acuna et al. 2012a,b). In addition to the indirect effect of promoting toxic cyanobacteria blooms, high ammonia levels can also directly impact species of concern. Connon et al. (2011) found that Delta Smelt genes contributing to cellular membrane integrity, among others, were affected by ammonia exposure, which may increase the susceptibility of the smelt to other contaminants.

Temperature: Temperature affects a wide range of ecological functions and characteristics, including, but not limited to, the amount of dissolved oxygen held by water, the toxicity of contaminants, decomposition rates of organic matter, and metabolic rates of poikilotherms. As with salinity, estuarine organisms have differing ranges of tolerance to temperature, so it is a primary determinant of wetland residency. Heterogeneity in temperature is an important aspect of refuge and metabolic regulation for some species (Armstrong et al. 2013; and see salmon conceptual model).

Water temperature within a wetland restoration site will correlate with air temperature and vary with its position on the ecocline, but is also modified by physical characteristics of the wetland itself. Tidal water that inundates the marsh surface functions much like the fluid in a heat exchange system, removing or contributing heat to the wetland depending on season or time of day, though whether tidal restoration will result in a net increase or decrease in temperature depends on the inundation regime (Enright et al. 2013).

Changes in temperature associated with water management and climate change have the capacity to affect the distribution, and possibly persistence of a given species. For example, Brown et al. (2013) predict worsening conditions for Delta Smelt in the face of climate change.

Link to the Contaminants model

Chemical Stressors: Water-borne pollutants may be imported to a wetland restoration site from surrounding areas, or chemical stressors may already be present in site soils as a legacy of past use. Either may compromise the ability of a restoring wetland to attain its desired ecological functions and/or constrain what restoration actions are allowed. A major concern for the health of humans and wildlife is the potential for the reintroduction of tidal action to a restoring wetland to increase the methylation rate of mercury, increasing its mobilization into the food web (Henneberry et al. 2012). Both mercury and selenium, an element delivered in abundance to the Delta from agricultural irrigation return water, bioaccumulate and biomagnify in certain lower-level consumers, resulting in significant impacts to higher trophic levels. For example, Stewart et al. (2004) used stable isotope ratios to show that predators, such as Sacramento Splittail, that routinely consume the filter-feeding *Potamocorbula* have higher tissue concentrations of selenium (and resulting deformity occurrence) than consumers of pelagic crustaceans. Laboratory work has also shown the ill effects of dietary selenium on Splittail (Teh et al. 2004).

Pesticides and herbicides, some of which are commonly directly applied to restoration sites to control non-native aquatic vegetation, do not appear to be acutely toxic to vertebrates at environmentally relevant concentrations, but some (or adjuvants commonly applied in formulations with the active ingredients) have been linked to sublethal effects that reduce fitness in fishes (Xie et al. 2005). For example, the pyrethroid insecticide esfenvalerate impairs swimming ability in larval Delta Smelt (Connon et al. 2009). Endocrine disrupting activity associated with environmentally relevant mixtures of chemicals can also lead to skewed sex ratios, with implications for population-level growth effects. In Suisun Marsh, Brander et al. (2013) discovered significantly more male than female Silversides in urban-influenced waters, which were characterized by elevated estrogenic and androgenic activity, as compared to waters bordering more rural areas. Those males also showed poor gonad development, and slower somatic growth when compared to Silversides from less contaminated waters and to females from the same urban sites (Brander et al. 2013).

In addition to sublethal effects on fishes of interest, contaminants could have indirect effects on population growth through acute toxicity to invertebrate prey species at very low concentrations (Weston et al. 2014). The overall effects of these chemicals on populations of wetland animals are exceedingly difficult to quantify, as combinations and differing formulations of chemicals may act synergistically or antagonistically. Additionally, concern is growing that other man-made chemicals found in pharmaceutical and personal care product residues (that would seem to be much more innocuous than poisons we intentionally apply to the environment) are having detrimental effects on aquatic species (Brandão et al. 2013). Laboratory studies of juvenile Coho Salmon have demonstrated that environmentally relevant concentrations of the synthetic estrogen that is commonly used in oral contraceptives alter the expression of a host of genes associated with pituitary function. Changes in reproductive function and circadian rhythm signaling are among the more dramatic potential responses to such environmentally induced gene regulation in populations subject to wastewater treatment plant effluent (Harding et al. 2013).

Tier 4: Biological Site Attributes

The internal physical wetland structure and processes are the major drivers behind the biological structure and processes, though feedback loops exist at multiple levels. The chief biological attributes considered in this model are 1) Food web structure and in situ production and 2) Plant biodiversity and architecture. While vegetation is a component of the food web through both limited direct grazing and the detrital pathway, we have separated the two functions because of the physical structure and habitat provided by vegetation apart from its role in food web support.

Food web structure and productivity:

Link to the Food Web model

Despite relatively high concentrations of inorganic nutrients, the modern SFE supports variable, but generally low primary productivity (Jassby et al. 2002). This stands in stark contrast to the characterization of estuaries as among the most productive ecosystems in the biosphere (Odum 1956). In well-functioning estuaries, the interaction of tides and freshwater flow within shallow basins tends to maximize inorganic nutrient mixing and regeneration, which can allow for high in situ photosynthetic rates. The interaction of complex geomorphology with tidal flows creates heterogeneity in water residence times and temperature distributions, further influencing photosynthetic rates (Enright et al. 2013). Additionally, a variety of physical habitats supports microbial processes acting on abundant plant material, supporting detrital food web pathways. Within the SFE, multiple factors are thought to limit productivity, including excessive nutrient inputs (Dugdale et al. 2007), grazing by invasive bivalves (Alpine and Cloern 1992; Kimmerer 2002), light-attenuating turbidities (Jassby et al. 2002), and simplification of habitat structure (Jassby et al. 2002; Moyle et al. 2010).

Habitat heterogeneity is also important to support estuarine food web function at higher trophic levels. Many animals move predictably with the tides as preferred feeding habitats (e.g. marsh plain or edge) are periodically made available by the flooding tide (Kneib and Wagner 1994; Rozas 1995; Bretsch and Allen 2006). In some cases, these movements can serve as positive feedback for high wetland primary productivity as nutrients regenerated through nekton excretion in subtidal areas during low tide are pulsed back onto the marsh surface with the flood (Haertel-Borer et al. 2004; Allen et al. 2013). Tidal wetlands along all coasts of the U.S. have been identified as nursery habitats for numerous estuarine-dependent species, largely due to their productivity (Minello et al. 2003; Hughes et al. 2014). The paucity of remaining tidal wetlands in the upper estuary precludes a robust system-wide assessment of their importance to our species of concern (Brown 2003; Herbold et al. 2014), but studies of nearby relict and restoring wetlands indicate autochthonous production, even in newly restored wetlands, can be significant (Howe and Simenstad 2011).

Link to the clams model

As food limitation is hypothesized to be an important bottom-up driver of precipitous declines in native fish populations (Sommer et al. 2007; MacNally et al. 2010), and tidal wetlands are excellent sources of food elsewhere, the interest in restoring tidal wetlands in the Delta is understandable. However, within an estuary that has been described as "the most invaded estuary in the world" (Cohen and Carlton 1998), food web support for at-risk species becomes more complicated. Shifts in the composition of producer and consumer assemblages due to invasion by non-native species can alter the food web to the detriment of native fishes. This appears to have been the case in the SFE. The presence of two Asian clams, *Potamocorbula amurensis* and *Corbicula fluminea*, is implicated in the depression of phytoplankton and zooplankton biomass within the estuary (Kimmerer et al. 1994; Lucas et al. 2002; Feyrer et al. 2003; Winder and Jassby 2011, see Invasive Clams model for more info)). Grazing by clams and microzooplankton can exceed in situ phytoplankton production within parts of the estuary (Lopez et al. 2006; Kimmerer and Thompson 2014) and clam grazing may have a particularly large impact during the lower productivity period of the winter and spring, which is a critical period for young at-risk fishes (Miller and Stillman 2013).

The quality of pelagic prey items that remain for fishes exacerbates the effects of declines in quantity. For example, in the Suisun region, the cyclopoid copepod *Limnoithona tetraspina* now accounts for more crustacean biomass than the larger calanoid copepods that previously were more common components of pelagic fish diets (Winder and Jassby 2011). Although *Limnoithona* appears to be food-limited itself, it is thriving due to low mortality rates partially explained by the difficulty visual predators have in picking out this small and relatively inactive copepod; it represents a "trophic dead end" (Gould and Kimmerer 2010).

Link to the smelt model

While increasing the availability of pelagic food resources for fishes is a goal of tidal wetland restoration, other prey communities should also develop within a restored wetland complex. Insects, infauna, and epibenthic macroinvertebrates have been significant components of fish diets in several restoration sites within the estuary, including on Liberty Island and in west Delta locations (Howe et al 2014; Whitley and Bollens 2014). Even Delta Smelt, which are generally described as pelagic predators, take advantage of larval insects and amphipods produced on recently restored wetlands and other Delta habitats (Moyle et al. 1992; Whitley and Bollens 2014; Slater and Baxter 2014). These diverse invertebrate prey communities can become established early in the restoration trajectory (Tanner et al. 2002).

Though potential for food competition exists, the extent to which it actually occurs is not well understood for our species of interest. There is some evidence within restoring wetlands of resource partitioning by a fish community, which would tend to reduce competitive pressure (Whitley and Bollens 2014). A trophic interaction of greater concern is predation by non-native fishes, which is counted among potential top-down drivers of recent fish declines (Baxter et al. 2010). Striped Bass, Largemouth Bass, catfish, and Inland Silversides eat listed fish species, though the magnitude of predation varies and is modulated by the environment in which they interact (Baerwald et al. 2012; Nobriga et al. 2013; Ferrari et al. 2014).

Link to the Aquatic Vegetation Model

Plant architecture and biodiversity:

While not all of a restored wetland will be covered with macrophytes, the physical presence of vegetation has great implications for a host of other processes. As primary producers, vascular plants play a role in the food web, but they are arguably more influential as providers of habitat complexity that can provide substrate for epiphytes, refuge for prey, or cover for ambush predators, thus modulating interactions of other food web components (Ferrari et al. 2014, Yarrow et al. 2009). The composition of the plant community will be partially determined by environmental variables, but at the same time, plants change water quality and hydrology and act as ecosystem engineers to alter the geomorphology of the wetland (Temmerman et al. 2005, Yarrow et al. 2009). The particular effects of vegetation on biotic interactions as well as on the physical and chemical characteristics of the wetland will depend on the composition of the plant community. In addition to the emergent vegetation that is typical of marsh plains, submerged and/or floating aquatic vegetation, the most common species of which are nonnative and invasive, may be present within a wetland complex. These different types of vegetation may have very different effects on physical wetland structure and aquatic community structure (Toft et al. 2003, Brown and Michniuk 2007, Whitley and Bollens 2014).

Given the impact of vegetation community composition, factors affecting plant establishment are important considerations in planning a tidal wetland restoration project. These factors include, among others, the connectivity of the project site to existing patches of vegetation, water velocity and quality (especially salinity), and inundation regime. In some cases, natural dispersal and germination will be sufficient to create a desirable plant community on a restoration site, but other cases may require active human intervention to establish desirable species and/or discourage the growth of undesirable species. The appropriate course of action for a particular restoration project is not always clear, but monitoring will allow the possibility of adaptively managing any vegetation gone awry (Mitsch et al. 2012).

As described in the Aquatic Vegetation and Wetland Evolution Conceptual Models, there are two-way interactions between surface elevation and plant establishment, growth, and decomposition rates (Reed 2002; Williams and Orr 2002, Culberson 2001). Likewise, alterations in water quality due to plant presence provide feedback to further primary production. For example, as water flowing through rooted vegetation slows and drops some of its sediment load, some primary producers may benefit from the increased water clarity. On the other hand, mats of floating aquatic vegetation or dense stands of emergent vegetation can serve to shade out any producers beneath the canopy. These and other vegetation types can act as ecosystem engineers or keystones and exert considerable control over community composition (Yarrow et al. 2009). Early colonizers of a wetland undergoing restoration thus can serve to facilitate or inhibit subsequent production (Bertness and Shumway 1993; Leck 2013).

Animals are also affected by changes in water clarity, velocity, and temperature brought about by plant presence. These dynamic aspects of fish habitat, in combination with the physical structure of the plants and geomorphology influence the types of organisms that will be attracted to the wetland and their success therein. For this reason, an objective of many wetland restoration projects is to discourage colonization by invasive weeds, such as Brazilian waterweed (*Egeria densa*) which creates conditions favored by non-native fishes, some of which may feed upon our species of concern (Nobriga et al. 2005; Nobriga and Feyrer 2007; Brown and Michniuk 2007; Ferrari et al. 2014).

Tier 5: Wetland Production

Because this model is being revised and updated specifically to monitor restoration sites designed to benefit Delta Smelt, Longfin Smelt, and Chinook Salmon, we have concentrated on outcomes mandated with the Biological Opinions for these species.

Link to the Food Web model

Exported Production:

In addition to providing habitat for at-risk species, a rationale behind tidal wetland restoration is that productivity generated on the wetland will be exported to surrounding waters to provide benefits to individuals that do not directly use the wetland. The “outwelling” hypothesis, the concept that estuarine wetlands are exporters of productivity, has been most often explored in the context of salt marsh systems (Teal 1962; Odum 1980). Salt marsh-derived carbon can end up in pelagic fish tissues (Weinstein et al. 2000), but studies in these systems suggest that questions about export of materials and energy must be asked with regards to temporal scale (Murray and Spencer 1997). For example, while the highly productive marshes of North Inlet, SC, are consistent exporters of dissolved organic carbon, inorganic nutrients, and macrodetritus, the net direction of flux of phytoplankton and zooplankton varies seasonally (Dame et al. 1986). Intermittent events, such as storms, can also greatly affect the magnitude and timing of any subsidy to pelagic habitats (Odum 2000).

Though tidal freshwater outwelling studies are relatively rare, those that have been published echo the variable nature of exchanges seen in salt marshes. Fluxes of materials to and from Liberty Island (in the Cache Slough region of the Delta) showed overall seasonal patterns over the course of a 17 month study, with the wetland exporting chlorophyll in fall and mesozooplankton during most of the year except summer (Lehman et al. 2010). Fluxes were driven predominately by tidal forces during that study, but the authors noted that interannual variability is likely to be high in consideration of vast swings in riverine flow from year to year. They also found that production and transport of some materials varied on the scale of hours to days (Lehman et al. 2010), as had been found in other Delta wetlands. In the shallow waters of flooded Mildred Island, the diurnal cycle of productivity superimposed on the semi-diurnal tidal cycle produced a bi-weekly cycle of chlorophyll export, illustrating the importance of high-frequency sampling to understand these processes (Lucas et al. 2006). Even if the net export of production is close to zero on an annual basis, export of production on these smaller time scales may make it available to consumers foraging in areas directly adjacent to the wetland. As pointed out by Lehman et al. (2015b), movement of materials with the tides is more relevant than seasonal or yearly fluxes "because it works within the life cycle of plankton."

Spatial variability in primary production is also apparent within a given wetland (Lucas et al. 2006; Lehman et al. 2010, Lehman et al. 2015b). While this is partially due to the effects of differences in residence time and light exposure on photosynthetic rates, biotic controls are also important. Whether high phytoplankton growth rates are realized as high biomass available for export will be largely determined by the degree of top-down regulation asserted by filter feeders, in particular invasive clams (Lopez et al. 2006; Lucas and Thompson 2012). Differences in hydrography between wetlands and concentration of invasive clams can explain how one wetland is a net source of production while another that seems to be similar habitat acts as a sink (Lucas et al. 2002).

Beyond knowing which wetlands are sources of organic materials and the timing of export, the form of organic matter is also a subject of interest. Though phytoplankton-derived organic matter is a relatively small component of particulate organic matter in the upper estuary, it is highly bioavailable and is thought by many to be the primary foundation for pelagic food webs (Jassby and Cloern 2000; Sobczak et al. 2002; Sobczak et al. 2005). However, other investigators present evidence that detritus can be important to the pelagic food web despite its lower nutritional value because it is more consistently available (Howe and Simenstad 2011). Stable isotope work confirms that wetland patches with connectivity can subsidize the food webs of one another, and suggests that adjacent shallow habitats may also benefit from wetland-derived organic matter of all types (Howe and Simenstad 2007). Wetland-derived productivity can also be exported actively as animals forage within the wetland for a time and then depart (The "trophic relay" hypothesis) (Kneib 2000; Beck et al. 2001).

Link to the
Delta Smelt
and Chinook
Salmon models
(d S)

Target species - populations and habitat:

The fish species native to the SFE evolved in a highly complex and dynamic environment. Even species characterized by short lives and narrow ranges of abiotic tolerances, such as Delta Smelt, thrived through drought and high flow periods alike as they were able to find the appropriate combination of water quality and physical structure somewhere within the historical estuary (Moyle et al. 2010). Chinook Salmon life history diversity was also supported as the estuary acted as rearing grounds for populations with various run timings, freshwater residence periods, and emigration rates. Though significant declines in populations of native fish species were observed well after the conversion of the majority of historic tidal wetlands, it is important to note that fish population trends were not measured prior to wetland conversion, and that loss of habitat in this critical area likely reduced species' resilience to the multitude of other stressors affecting them. The restoration of tidal wetlands containing dendritic channels and shallow subtidal areas is intended to contribute to at-risk species recovery by providing habitat to support survival and growth of one or more life stages of each species (Moyle et al. 2010). For a more detailed description of how tidal restoration may affect at-risk fishes, see the conceptual models for individual species.

Juvenile Chinook Salmon may rear for up to several months in the SFE before emigrating to the ocean (Sommer et al. 2001; del Rosario et al. 2013). Though few tidal marshes remain in the upper estuary to assess their benefits for juvenile Salmon, inundated vegetated floodplains in the region have been shown to support greater apparent growth and survival than individuals rearing in adjacent habitats (Sommer et al. 2001). Studies of a series of tidal wetland restoration sites in Oregon's Salmon River Estuary revealed juvenile Salmon rearing on restored sites, as well as regional-scale effects such as increased diversity of residence times and size at migration compared to the pre-restoration population (Bottom et al. 2005). It is hoped that the Central Valley runs of Chinook will exhibit similar improvement with estuarine tidal wetland restoration.

Delta Smelt are estuarine specialists. They only occur within tidally influenced areas with narrowly-bounded water quality values; in the Delta this habitat has declined an estimated 78% over the last 30 years (Feyrer et al. 2011). Delta Smelt habitat availability is tightly tied to low salinity and high turbidity. Appropriate values of these factors are prerequisites to any use of habitat (Feyrer et al. 2011). Delta Smelt are primarily pelagic, but evidence suggests areas adjacent to tidal wetlands with long residence times may be favorable for spawning and juvenile rearing (Merz et al. 2011; Murphy and Hamilton 2013; Sommer and Mejia 2013). Though direct use of wetland complexes by Delta Smelt is rarely noted, they have been caught in relatively small tidal channels (Gewant and Bollens 2012). Work in the Liberty Island restoring wetland, which includes extensive shallow open-water areas as well as marsh and tidal channels, has documented Delta Smelt foraging and apparent resource partitioning from co-existing fish species, though implications for increased growth and survival are not known (Whitley and Bollens 2014). Liberty Island is within the Cache Slough Complex, where a contingent of Delta Smelt resides year-round rather than migrating for spawning and rearing (Sommer et al. 2011).

Along with relatively favorable elevations and turbidity, these resident fish play a role in making the Cache Slough Complex a preferred area for wetland restoration. New restoration sites may also create areas of high-turbidity, low-salinity waters favored by Delta Smelt, though their presence will clearly depend on many factors. In particular, temperature may also influence the extent of habitat availability to Delta Smelt. Temperatures during the summer often exceed their upper thermal limit, thereby decreasing the amount of available rearing habitat (Feyrer et al. 2011).

Less is known about Longfin Smelt use of tidal wetlands. They are anadromous, spending much of their lives in the San Francisco Bay or the Gulf of the Farallones and migrating into the estuary to spawn (Moyle 2002). They appear to use Suisun Marsh as spawning habitat, so tidal restoration in both Suisun Marsh and the western Delta could provide important rearing habitat for Longfin Smelt (Moyle 2008). There is little information on whether Longfin Smelt utilize tidal wetland habitats for rearing; however, larval Longfin Smelt have been collected in Liberty Island (Nozuka et al. 2005) and in various tidal sloughs (Sommer et al. 2004, Gewant and Bollens 2012).

Support for these target species will incidentally provide habitat for a variety of other plant and animal species. Some native species, such as Sacramento Splittail (*Pogonichthys macrolepidotus*), may benefit from increased access to shallow water habitat interspersed with emergent vegetation (Sommer et al. 2002; Moyle et al. 2004). An undesired outcome of tidal wetland restoration would be extensive invasion by non-native species that may suppress primary and secondary productivity. Colonization by one of the Asian clam species could short-circuit in situ food web support (Lucas and Thompson 2012), and certain invasive weeds could reduce turbidity to a harmful extent while harboring potential predators of native fishes (Nobriga et al. 2005; Brown and Michniuk 2007; Ferrari et al. 2014). Continued research on restoration design is needed to discover wetland configurations that would discourage colonization by undesirable species while nurturing native species. Tidal wetlands that mimic the structural heterogeneity of the Delta of the past are most likely to support native species recovery (Moyle et al. 2010; Herbold et al. 2014).

Conclusion

Time will tell if restoring tidal action to parcels of the upper estuary will affect at-risk fish species, but we will be able to discern those effects only if adequate monitoring resources are brought to bear on the correct questions. This conceptual model, like its companion models, was developed to assist in identifying the correct questions. The questions range in scale from the perspective of a tiny fish to the landscape of the estuary. Many of them will be difficult to answer, and will require collaboration across disciplines. Smart monitoring and “special studies” will provide mechanistic links between fish and habitat to reduce uncertainties and guide future restoration efforts.

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Interagency Ecological Program

Tidal Wetlands Restoration Monitoring Project Work Team

Food Web Conceptual Model

DRAFT

Prepared by: Rosemary Hartman, Larry Brown, Bruce Herbold, Adam Ballard, and Jim Hobbs

Based on the DRERIP Food Web conceptual model:

Durand, J. (2008). Delta Aquatic Foodweb Conceptual Model. Sacramento, CA, Delta Regional Ecosystem Restoration Implementation Plan

PREFACE

This Conceptual Model is part of a suite of conceptual models which is designed to guide monitoring of restoration sites throughout the San Francisco Estuary (SFE), but particularly within the Sacramento-San Joaquin Delta (Delta) and Suisun Marsh. The conceptual models have been developed based on the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) models, and are designed to aid in the identification and evaluation of monitoring metrics for tidal wetlands restoration projects. These models are designed to describe our understanding of how tidal restoration may benefit at-risk fish species.

Contents

Introduction

This is a general model of food webs within tidal wetlands and how physical features of the wetland affect the structure and function of the food web. It focusses on how the food web provides support for (or may reduce support for) threatened fish species.

Many tidal restoration sites in the Delta are currently being constructed to comply with environmental regulatory requirements associated with the operation of the Central Valley Project and State Water Project. These include the Biological Opinions for Delta Smelt, and salmonids (United States Fish and Wildlife Service 2008, National Marine Fisheries Service 2009), and the Incidental Take Permit for Longfin Smelt ((CDFW) 2009).

These regulatory requirements are based on the hypothesis that the decline of listed fish species is due in part to the decline in productivity (phytoplankton and zooplankton in particular) or alterations in the food web such that production is consumed by other species in the Estuary (Jassby et al. 2002, Baxter et al. 2010). Shallow subtidal habitat and intertidal habitat can be highly productive, so restoring areas of tidal wetlands may result in a net increase in productivity that will provide food web support for these fish species. However, other factors such as invasive bivalves that can reduce phytoplankton and zooplankton biomass and invasive predatory fish species that may compete with or prey upon fishes of concern can limit the utility of tidal wetlands for food web support (Lucas and Thompson 2012, Herbold et al. 2014).

This model builds off the previous DRERIP model (Durand 2008) and an update of the previous DRERIP model (in review).

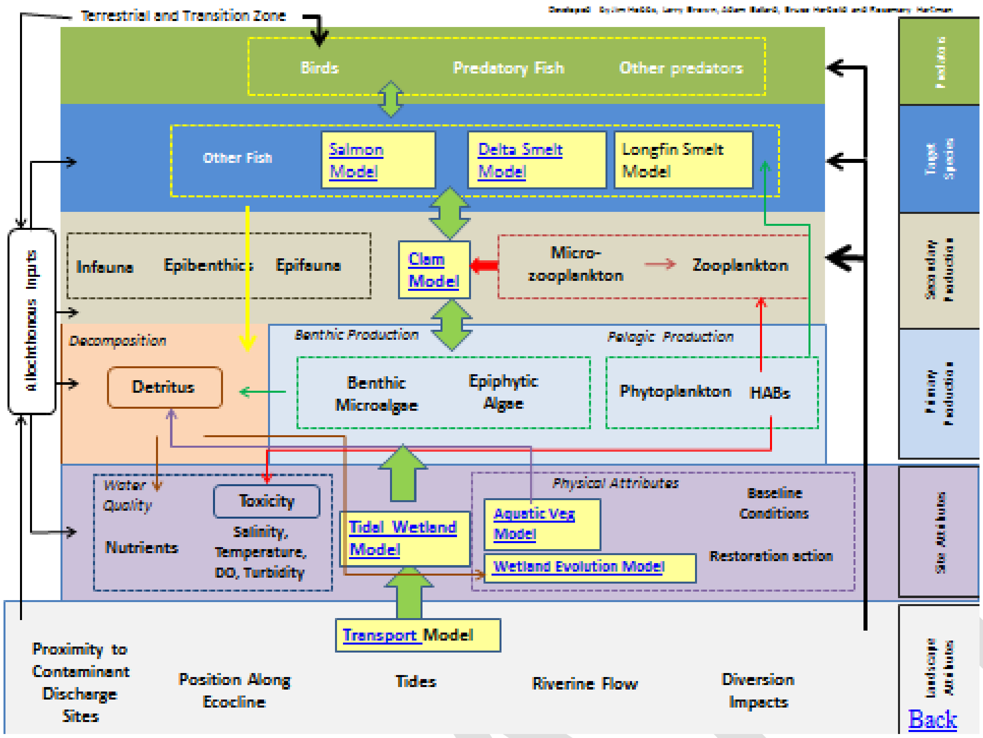

Figure 1: The model, in all its glory.

Model Structure

This model concentrates on food web support for the species of concern (Chinook Salmon, Delta Smelt, and Longfin Smelt), but also acknowledges other fish species that use tidal wetland habitat. It does not focus on invertebrates, terrestrial vertebrates, or plants except in relation to how they support the species of concern.

This model shares the "tiered" structure of the other tidal wetland conceptual models, though its goal of describing the structure of a food web causes it to approach the tiers somewhat differently from a species-specific model (eg. Delta Smelt Model) or a process-based model (eg. Wetland Evolution Model). The Food Web Model starts out with the large-scale to local-scale structure of the General Wetland model, going from Tier 1: Landscape Attributes to Tier 2: Site attributes, but higher tiers follow a traditional food chain structure. Tier 3 is Primary Production (including decomposition as another process that makes carbon available to the food chain), Tier 4: Secondary Production, and Tier 5: Target Species (the "outcome" tier). Unlike most of the other models, the food web model has a "Tier 6" above the outcome tier describing predators that may exert top-down control on target species.

This model describes the structure of the food web in tidal wetland complexes. Therefore, the tiers are referred to as "attributes" rather than "drivers". Arrows represent known relationships, both positive and negative, rather than flows of carbon or energy through the foodweb. This model uses the same conventions for boxes and arrows as the General Wetland Model and other sub-models. Attributes surrounded with dotted lines interact strongly. Arrows represent specific, describable linkages or relationships between attributes. Thickness of arrows generally describes level of hypothesized importance; however this has not been quantified systematically. Thick arrows moving between tiers that do not connect to specific factors represent relationships with all attributes in the following tier. Arrow and text color is used to aid in following arrows and does not convey meaning. Yellow boxes are links to sub-models which provide greater elaboration on the specifics of particularly important processes, structures, or species.

Body of model.

Tier 1: Landscape attributes

Riverine Flow

Inputs from the rivers of the Delta (chiefly the Sacramento and San Joaquin Rivers, but also Cosumnes, Calaveras, and other smaller rivers), are a critical control of the salinity, turbidity, inundation regime, and geomorphology of the site. The Low Salinity Zone (area from 1 PSU to 5 PSU where most Delta Smelt are found), changes in location and extent depending on Delta inputs (Brown et al. 2014). Higher riverine inputs also contribute to cooler temperatures, which limits the growth of harmful algal blooms (Baxter et al. 2013).

Riverine input also provides much of the allochthonous inputs of nutrients and materials to the site (see Allochthonous Inputs below, and the Transport Model). Freshwater flow is highly variable, both over the course of the year and between years (Kimmerer 2002). River flow controls access to the site for anadromous fish from upstream breeding habitat (Perry et al. 2009). Some research suggests that the magnitude of riverine flow controls the occurrence and size of phytoplankton blooms (California Department of Fish and Game and Orsi 1992, Jassby 2005).

Tides

Link to Transport Model

Tides move water in and out of wetlands over the tidal cycle. They determine flux of materials and allow dispersal of productivity and nutrients on a short time scale (sub-daily). Tidal flow is normally larger than riverine input in all but extremely high-flow conditions (Kimmerer 2004). Tides also allow exchanges of productivity against the direction of net riverine flow, within a tidal excursion. Tidal exchange aids in transfer of nutrients and species between habitat patches (Diggory and Parker 2011, Mitchell et al. 2012, Enright et al. 2013). See the General Tidal Wetland model and Transport Models for more information.

Position along ecocline

Link to General Wetland

Position along the ecocline includes both the distance from the ocean as well as position on the upland-deep subtidal gradient. Geographic position interacts with riverine flow to control the location and extent of habitat with appropriate water quality (salinity, turbidity, temperature, etc). Distance from riverine inputs will also help determine sediment supply, which will affect turbidity on-site, rate of marsh accretion, and proportion of organic matter in the soil (see Wetland Evolution and General Wetland Model for more information)

Diversion Impacts

Link to Delta Smelt and Salomn

Water withdrawals from the Delta reduce Delta outflow and can result in landward movement of the salinity gradient, affecting species composition and productivity (see Salinity section below) at a geographic location. Diversions may also directly entrap phytoplankton, zooplankton and/or fish of concern (Nobriga et al. 2004, Monsen et al. 2007, Kimmerer 2008), though the degree to which entrapment affects overall population of at-risk species is hotly debated (Kimmerer 2008, Grimaldo et al. 2009a, Miller 2011). Overall changes to hydrology caused by diversions have been linked to estuary-wide decreases in phytoplankton biomass (Lehman 2000a). While most large diversions will likely be outside tidal restoration sites, they may reduce access to the site, change the hydrology of the site, or alter allochthonous inputs of productivity. See Delta Smelt Model and Salmon Model for more information.

Proximity to contaminant discharge sites

Link to Contaminants Model

Proximity to contaminant discharge sites affects contaminant concentrations and thus toxicity on site. (Presser and Luoma 2006, Krupa et al. 2011). Proximity to wastewater treatment plants also affects primary production due to increases in ammonium and other nitrogen species downstream of the plants (see Nutrients below). Proximity to managed wetlands and agricultural discharges may cause increases in biological oxygen demand and lead to low DO as well as nutrient and DOC inputs (Stringfellow et al. 2008). Both agricultural discharges and urban stormwater discharges may be a major source of pesticides, often at levels toxic to zooplankton and invertebrates (Weston et al. 2014). See the contaminants model for more information.

Allochthonous Inputs

Allochthonous inputs may come from riverine, tidal, or terrestrial sources, so they flow from the landscape tier to all other tiers in the model.

Aquatic

Plankton and nekton as well as particulate organic material and nutrients are transported through tides and freshwater flow. (Dean et al. 2005, Lehman et al. 2010b). One study of organic carbon sources in wetlands in the SFE found an average of 11% of organic carbon in marsh consumers came from Bay inputs, while 13% came from riverine inputs (Howe and Simenstad 2011). This means the majority (76%) of the organic carbon consumed by the resident species studied was produced within the marsh; however, allochthonous inputs may vary in importance. Different habitat types and different regions may have vastly different degrees of allochthonous material. While Howe and Simenstad (2011) found that Napa River marshes have a relatively small degree of allochthonous input, Suisun Bay phytoplankton biomass is dominated by plankton imported from upstream (Jassby 2008). The vast majority of organic carbon in the estuary as a whole comes from allochthonous sources, though much of it is from macrophyte detritus, which is often much less bioavailable than phytoplankton production (Jassby and Cloern 2000).

Terrestrial

Nutrients and DOC from the terrestrial environment may be imported to the wetland via tributary inputs, runoff, or directly as detritus from riparian vegetation (Jassby and Cloern 2000, Oh et al. 2013). The total loading of organic sources in the estuary is primarily from detritus brought in via tributaries, with most of this eventually being traced to terrestrial sources rather than aquatic sources (Jassby and Cloern 2000). Despite its ubiquity in the system, terrestrial detritus is estimated to be a relatively small proportion of total organic matter incorporated into pelagic consumer tissue (Howe and Simenstad 2011), but it is an area requiring more study.

The magnitude of allochthonous inputs changes with season and with yearly changes in climate. Tributary loading may increase by an order of magnitude during winter and spring due to increased precipitation and resulting increases in organic material in runoff, though this effect is much lower in dry years (Jassby and Cloern 2000). Organic matter from direct agricultural runnoff may double during the winter versus the summer, though there is less inter-annual variation than in tributary loading (Jassby and Cloern 2000).

Input of animal material may be more directly incorporated into the aquatic food web. Terrestrial invertebrates living on over-hanging riparian or emergent vegetation may often become prey for wetland fish species (Howe et al. 2014). Even small mammals foraging along the intertidal zone may become prey for predatory fish. Terrestrial animals, especially birds, may enter the wetland from the terrestrial zone and consume fish, vegetation, and invertebrates (See Birds section below).

Tier 2: Site attributes

Water quality

Nutrients

Nutrients are usually not limiting in the system, however the SFE as a whole is much lower in productivity than other nutrient-rich estuaries, potentially due to light limitation (Cloern 2001). Only during blooms of phytoplankton (once a common occurrence, now rare), are nutrients reduced to the point of becoming limiting (Glibert et al. 2014a).

The chief sources of anthropogenic nutrients are agricultural drains and wastewater treatment plants (Hager and Schemel 1992). There are higher concentrations of nitrogen and phosphorus on the San Joaquin River than the Sacramento River due to lower flow and greater agricultural land use. However, the Sacramento River has higher total nutrient loading, due largely to major waste-water treatment plants (Monsen et al. 2007). The largest of these, the Sacramento Waste Water Treatment Plant on the Sacramento River near Clarksburg, provides a very large point source of nutrients, so building restoration sites directly downstream of this plant may benefit from increased productivity on site or suffer from eutrophication, depending on physical factors.

The form of nutrients present and their relative concentrations may also influence productivity and community composition. High concentrations of ammonium (NH_4^+, frequently associated with wastewater discharge) have been implicated in reductions in diatom biomass and increases in the proportion of flagellates and cyanobacteria (Dugdale et al. 2007, Jassby 2008). Phytoplankton preferentially uptake NH_4^+, even when NO_3^- is plentiful, but diatoms and chlorophytes have lower growth rates when uptaking NH_4^+ compared to NO_3^-. Therefore, high NH_4^+ concentrations may skew the community toward dominance of cyanobacteria and flagellates that are comparatively more efficient at using NH_4^+(Glibert et al. 2014b).

Increases in the proportion of all forms of nitrogen when compared to phosphorus may favor flagellates and cyanobacteria over diatoms (Glibert et al. 2011). In the SFE, long-term trends of increasing N:P ratio and dominance of reduced (NH_4^+) rather than oxidized (NO_2^{-2}, NO_3^-), forms of nitrogen have been correlated with major food web alterations. It has been hypothesized that the change in phytoplankton communities cascaded through the food web all the way to the pelagic organism decline (Glibert 2010). The effect of nutrient ratios and nutrient forms is an area of active research, and the scientific community has not come to a clear consensus.

Toxicity

Link to contaminants model

Contaminants may be directly toxic to fish, invertebrates, and primary producers. They may have sub-lethal effects. Sub-lethal effects, such as reduced growth or fecundity, may lead to reductions in population growth rates (Giddings et al. 2000, Brooks et al. 2011). Toxins may bioaccumulate in organisms over time and biomagnify as they move up the food chain, resulting in larger effects on top predators compared to their prey (van der Oost et al. 2003). This can cause changes in species composition as a result of varying susceptibilities of organisms to these toxins (See Contaminates Model for more information).

Salinity

The position and characteristics of the salinity gradient are determined by the interaction of annual precipitation, riverine flows, landward tidal movement of saline water, and geographic location. It will strongly influence species composition for almost all components in the food web (Peterson and Vayssieres 2010, Hennessy and Enderlein 2013). Some species have wide salinity tolerances, whereas others do not. Salinity tolerances may interact with predator-prey relationships to drive community composition. For example, the invasive clam *Potamocorbula amurensis* is most abundant in brackish water areas and grazing by this species has been associated with changes in both total phytoplankton biomass and community composition across the Low Salinity Zone (area of water between 0.6 PSU and 4 PSU, centered around X2, shifting with Delta outflow but generally encompassing Suisun Bay and the Western Delta). At the landward edge of the Low Salinity Zone, large diatoms dominate, while at the seaward edge both total chlorophyll a and proportion of diatoms is much lower (Lehman 2000b).

Salinity also affects both the productivity (Watson and Byrne 2009) and the decomposition rates of macrophytes (Mendelssohn et al. 1999), which may lead to changes in the importance of detritus in the food chain (see Decomposition below).

For resident animals, rapid changes in salinity or salinities at the limit of species tolerance will increase osmoregulatory energy demand (Brooks et al. 2011). If food is not limiting, species may be able to survive at the limits of species tolerance, but will reduce their distribution if they do not have adequate food to compensate for increased energy demand. More mobile species may simply move with the salinity gradient. This is a likely mechanism for pelagic or other mobile fishes, such as Delta Smelt, Longfin Smelt and Chinook Salmon.

Temperature

All species have certain temperature requirements, so extreme high or low temperatures may prevent species from occupying a restored wetland and contributing to the food web. Climate change may put many areas in the Delta above the thermal maxima for Delta Smelt and other sensitive species (Wagner et al. 2011, Brown et al. 2013). Temperature increases metabolism in fish and their demand for food (Brooks et al. 2011). Increased temperature also decreases the ability of water to hold dissolved oxygen and increases the toxicity of certain heavy metals. Increases in temperature may increase phytoplankton production (Durand 2008), but may favor some taxa over others or lead to harmful algal blooms (See HAB section below). Simultaneous increased biological oxygen demand when algal blooms die and decreased ability of warm water to hold dissolved oxygen may cause fish die-offs if water becomes too warm.

Seasonal changes in water temperature, driven by increased air temperature and sunlight will increase metabolism during summer. However, the seasonal temperature swings vary by location, with smaller temperature changes and cooler water overall in the Suisun Bay/Grizzly bay area when compared to the Delta (Kimmerer 2004).

DO

Dissolved oxygen in the water has the potential to be a major limiting factor for fish and other organisms in a wetland. The Delta rarely experiences hypoxic conditions due to high vertical mixing rates and low productivity, though temporary periods of low DO have occurred in Suisun marsh (Brooks et al. 2011), and the Stockton Deep-water Ship channel (NMFS 2009). Food limitation (hypothesized to be a major factor in recent pelagic organism declines) can exacerbate the effects of hypoxia because energy reserves are not available to compensate for increased stress (Baxter et al. 2010, Brooks et al. 2011).

Turbidity

Turbidity produced from both organic particles (algae and detritus) and suspended sediment in the water column. The Delta generally has high suspended sediment concentrations when compared to other estuaries, but these concentrations have been declining over the past fifty years. This decline is due largely to reductions in sediment loading from major rivers since the cessation of hydraulic mining and the construction of sediment-blocking damns. Sediment concentrations may also be reduced locally by presence of submerged aquatic vegetation or high biomass of benthic grazers. (see Wetland Evolution model and DRERIP sediment model for more information).

Turbidity is a major limiting factor on phytoplankton production in the SFE, where production is generally light-limited rather than nutrient-limited (Jassby 2005). Due to the ubiquity of high turbidity in the estuary, the algal species that were historically dominant (diatoms) may be adapted to low light conditions, experiencing photoinhibition in high light (Glibert et al. 2014b). While community composition, nutrients, contaminants, and temperature do contribute to regulating phytoplankton productivity, light availability may explain as much as 80% of observed patterns in net primary productivity for the SFE (Cole and Cloern 1984).

High turbidity may also be beneficial for species of concern. Delta Smelt abundance is positively correlated with high-turbidity habitat, so long-term reductions in turbidity may be restricting smelt habitat availability in the estuary (Nobriga et al. 2008). Turbidity has been hypothesized to reduce predation risk for all Delta Smelt life stages (Ferrari et al. 2014). Turbidity also may increase feeding success by decreasing stress, though it also reduces feeding efficiency at very high turbidity levels (Baxter et al. 2013, Hasenbein et al. 2013). Turbidity has also been shown to reduce perceived risk in Chinook Salmon (Gregory 1993). See Salmon and Smelt models for more information on how turbidity affects specific species.

Geomorphology

Link to general tidal wetlands model

All of the aspects of geomorphology and topographic structure in the General Tidal Wetland model will affect the food web within the wetland, especially the bathymetry and water depth, channel formation, extent of shallow habitat, refugia (temperature and predation refuges), and residence time. Residence time and water depth (controlled by tides, riverine flow, and bathymetry (see General Wetland Model), are particularly important for phytoplankton production (see Phytoplankton section below).

Vegetation

Link to aquatic vegetation model

Vegetation affects the food web both by providing habitat and by primary production. There have been numerous studies relating fish community composition to aquatic vegetation type. Non-native centrarchid predators are generally most abundant in association with thick-canopied submerged aquatic vegetation such as *Egeria densa* (Brown and Michniuk 2007, Ferrari et al. 2014)(See Predation below). Diverse vegetation architecture and emergent plant structure also provides refuge from predation for some at-risk fish species, especially salmon (McLain and Castillo 2010). Open-water species such as Delta Smelt tend not to seek cover in vegetated habitats, instead benefiting from high turbidity habitats (Ferrari et al. 2014). See the Aquatic vegetation model for more information.

Plants provide physical substrate for production of epiphytic algae and epifaunal invertebrates. These may be a significant contribution to at-risk fish diets (See Epiphytic Algae and Epifauna sections below).

Production from vegetation (either submerged, floating or emergent), is relatively indigestible in comparison to algae and phytoplankton. Therefore, carbon produced by vegetation enters the food web chiefly through the detrital loop (See Decomposition section below).

Link to wetland evolution

Wetland evolution

After tidal action is restored to an area, colonization of vegetation and vertical accretion is expected to change the bathymetry, topographical heterogeneity, and habitat structure of the site (Kirwan and Murray 2007). This will theoretically cause development of a complex channel network, allowing fish access to food resource and refuge habitat of the productive intertidal vegetated area (Visintainer et al. 2006). Change in marsh elevation will also cause changes in residence time, leading to different rates of primary productivity. See the Wetland Evolution model for more information.-

Baseline conditions and restoration action

Starting site elevation, starting community composition, location of breaches, and decisions about on-site vegetation control or planting sets the beginning site and habitat attributes for the food web species and processes. These attributes are expected to change as the site evolves. See the Wetland Evolution model for more information.-

Tier 3: Primary Production

Benthic Production

Benthic microalgae

In shallow, unvegetated habitat, production by benthic algae can be quite high, though often varies spatially depending on species composition and tidal dynamics (Guarini et al. 2002). Benthic substrate is also critical for species composition and primary productivity in benthic communities, with higher grazing rates and higher algal turnover in sandy, rather than muddy substrates (Herman et al. 2001). Of the few studies of benthic algae in the SFE, one study found microphytobenthos production in marshes around the estuary to be 0.01-0.07 g/m2/day, an order of magnitude lower than phytoplankton production, and two orders of magnitude lower than macrophyte production, but this was considered an underestimate (Cohen et al. 2014). Another study found benthic algal production in the Delta to be higher, 0.05-0.1 g/m2/day, and an order of magnitude higher than found in Cohen et al's findings (Cornwell et al. 2014). Further study is needed to assess the actual contribution of benthic algae to the system.

Epiphytic alga

There is extensive literature on the importance of epiphytic algae to sea grass communities, but less work has been done in other tidal wetland communities. One study from an Atlantic salt marsh found epiphytic algal production to be quite high (as much as 130 mg C m-2 h-1) (Zheng et al. 2004). Another study of a salt marsh in Louisiana found epiphytic diatoms to be the major food source for copepods (Galvan et al. 2008). Biomass of epiphytic algae may be more important than biomass of macrophytes in determining consumer biomass and community composition in some marsh systems (Campeau et al. 1994). To date, there has been little study of the importance of epiphytic or epibenthic algae in the SFE.

Pelagic Production

Phytoplankton

Phytoplankton biomass is highly variable across space and time in the SFE, with few of the predictable seasonal patterns present in open ocean phytoplankton dynamics (Cloern and Jassby 2008). Overall, phytoplankton production in the SFE is low compared to other estuaries, and has been in decline for the past 30 years (Jassby 2008, Winder and Jassby 2011). This decline is likely largely due to invasion of the bivalve *Potamocorbula* in the brackish part of the estuary; however, the increase in the N:P ratio may also be involved to an unknown extent (See Clam Model and Nutrient section above). Due partially to the higher benthic grazing in the brackish part of the estuary, phytoplankton biomass, cell size, and diatom abundance tends to be higher at the landward edge of the Low Salinity zone (Lehman 2000b).

Phytoplankton production is tightly coupled to light availability, which is controlled by water depth and turbidity, and residence time (Thompson et al. 2008, Lucas et al. 2009). Below a point known as the “compensation depth”, photorespiration exceeds carbon fixation by photosynthesis, reducing the amount of primary production available for consumption by higher trophic levels (Durand 2008). This relationship means that shallow-water areas are often the most productive because phytoplankton never falls below the compensation depth so primary production can occur throughout the water column. Residence time is the other key control on phytoplankton biomass. Increased residence time allows for nutrient retention, biomass accumulation, and temperature increase when air temperatures are high. As discussed in the Temperature section, benefits of long residence time caused by increases in temperature will be greatest in summer. Decreased residence time allows for nutrient and organism flux to adjacent areas, and for nutrient recharge within the wetland (Durand 2008). Long residence times maximize production, however longer residence times may result in nutrient depletion with nutrients becoming limiting. Warm temperatures and reduced nutrients may encourage harmful algal blooms, or allow invasive bivalves to reduce phytoplankton biomass (Jassby 2005, Lucas and Thompson 2012).

Light availability and air temperature, and riverine flow (major controls on phytoplankton production) are highly seasonal. Short residence times, coupled with decreased temperatures and less sunlight in the fall and winter mean phytoplankton biomass is generally lower, with spring productivity levels often 5-10 times winter levels (Jassby and Cloern 2000, Jassby 2008). This increase in spring and summer productivity is often greater during dry years than wet years due to concentration of nutrients and increased residence time (Jassby and Cloern 2000). Blooms of beneficial phytoplankton such as diatoms and chlorophytes, once seasonally common, now only occur when there is a window of opportunity with low NH_4^+ concentrations, and long residence times, often tied to reduced flow during droughts (Glibert et al. 2014a). These are only likely to occur in areas with relatively low benthic grazer (clam) biomass (Thompson et al. 2008).

Phytoplankton community composition also affects productivity. In fresher water, the community is dominated by diatoms and green algae, which have higher chlorophyll a-specific net primary productivity (Lehman 2007). Phytoplankton species identify also determines food quality for higher trophic levels. Cryptomonads and diatoms are considered preferred food sources for zooplankton, while flagellates and cyanobacteria are less optimal (Orsi 1995, Cloern and Dufford 2005, Lehman et al. 2005). Over the past 30 years, the decline in phytoplankton biomass appears to have been accompanied by a shift away from diatoms and towards less preferred flagellates and cyanobacteria associated with increases in ammonium (discussed in the Nutrients section above), invasive clam grazing (Lucas et al, BDSC presentation), or a combination of the two. This has lowered availability of high-quality food for zooplankton (Lehman 2000b, Glibert et al. 2014b). Therefore, even when primary productivity is high, there may be reductions in zooplankton populations due to a reduction in food quality or reductions in availability of phytoplankton because of competition with benthic grazers.

Harmful Algal Blooms

Link to Contaminants Model

In recent years, blooms of cyanobacteria Microcystis spp. have occurred in the Delta, especially the relatively warm, slow moving freshwater of the South Delta (Lehman et al. 2005). Microcystis is not always toxic, but when it is it may be associated with reductions in survival of zooplankton and lesions in the livers of fish (Ger et al. 2010, Acuna et al. 2012). It has been shown to reduce other phytoplankton species through allelopathy in other systems, and hypothesized to affect other phytoplankton here as well (Lehman et al. 2010a). Even blooms that are not toxic may cause drops in DO due to the increased biological oxygen demand during decomposition of dead algae after the bloom is over, and provide lower food quality for consumers than other species of phytoplankton (Lehman et al. 2013, Morris and Civiello 2013, Lehman et al. 2014). Some copepods may actively reject Microcystis, so copepod numbers may not be reduced if there is still adequate alternate food supply, but cladoceran populations are often lower when Microcystis is present (Lehman et al. 2010a).

Decomposition

While decomposition is not a source of primary production, it is included here because it allows inaccessible carbon to be incorporated into the food web. The largest source of detritus in wetlands is the highly productive emergent vegetation, which is why most marshes are traditionally considered to be detritus-driven ecosystems (Darnell 1967). Some studies of the pelagic food web in the San Francisco Estuary suggest that the relative importance of detritus-based secondary production is low compared to phytoplankton-based production (Sobczak et al. 2005); however, detritus-based production is more important in other estuaries and traditionally linked to salmon (Maier and Simenstad 2009). Detrital carbon is not easily consumed by zooplankton, so must first be processed by bacterioplankton or protists before entering the pelagic food chain (Muller-Solger et al. 2002). A large proportion of detritus may not be bioavailable even to bacteria, and only a small fraction of what is available will become incorporated into higher trophic levels.

Detritus-based production may play a greater role in certain habitats within the SFE, and the contribution of detritus to the food web is expected to increase with tidal restoration. Decreases in phytoplankton standing stock due to invasive clam grazing, and increases in the invasive copepod *Limnoithona tetraspina*, which consumes primarily protists and rotifers (see Zooplankton below), may be increasing the importance of the detrital/microbial loop for the pelagic ecosystem (Winder and Jassby 2011). In some systems there is evidence that omnivorous mesozooplankton consume detrital particulate matter directly, allowing for more efficient incorporation of detritus to the food web (Van den Meersche et al. 2009), though this has not been extensively studied in the SFE.

In wetland areas, the proportion of carbon entering the consumer pool may be more dependent on detrital inputs than phytoplankton due to the proximity of large amounts of decaying vegetation. Production by macrophytes in these systems may be an order of magnitude greater than phytoplankton or benthic algae production (Cohen et al. 2014). Even samples from open-water and deep-channel habitat indicate detritus may be 95% of the particulate organic matter, swamping phytoplankton biomass (Sobczak et al. 2002). Stable isotope analyses indicate 30-50% (depending on sampling site) of carbon in resident consumers was derived from emergent vegetation (Howe and Simenstad 2011), though this may be significantly less for transient species such as salmon and smelt (Herzka 2005). The relative importance of any particular carbon source in consumer diets is normally directly related to its abundance on the landscape (Melville and Connolly 2003). Therefore, the relative proportion of marsh-derived carbon may be proportional to the area of tidal marsh (Howe, unpublished data). If more tidal marsh is restored, importance of marsh derived carbon to the estuary as a whole is expected to increase.

Transient species may benefit less from marsh-derived production than resident species. Rapidly-growing juvenile fish acquire the isotopic signatures of the wetland in which they feed in days or weeks, whereas larger, adult fish may never equilibrate with their environment (Herzka 2005). This means that once a juvenile fish has left the wetland the corresponding signature may be drowned out by the open-water signature relatively quickly, making it difficult to quantify the importance of wetlands to the population.

Tier 4: Secondary Production

Benthic/epibenthic and epifuanal

There in an increasing appreciation of the importance of benthic and epibenthic production, even for species traditionally labeled as “planktivores” (Howe et al. 2014).

Insects –

Insects, particularly benthic/epiphytic chironomid larvae are a key part of many fish diets in the SFE, both in littoral areas (Howe et al. 2014), and open water(Grimaldo et al. 2009b). They may make up 30% of Chinook salmon diets in certain areas (Grimaldo et al. 2009b), and insects may be the first benthic/epiphytic food resource for salmon to recover after tidal restoration (Gray et al. 2002). Chironomids are more tolerant to low oxygen than many other tidal wetland invertebrates, so may be an important source of secondary production in warm, high-residence time areas, such as dead-end sloughs, that are susceptible to low oxygen conditions (Gray et al. 2002). Insects may also enter the food web through fall-out from overhanging riparian vegetation and emergent marsh vegetation, though this is usually a low proportion of total insect biomass (Tanner et al. 2002).

Epibenthic invertebrates

Amphipods are important food sources for fish in littoral habitats, making up as much as 75% percent of their diets in vegetated areas (Grimaldo et al. 2009b, Howe et al. 2014). Even in open water habitats, amphipods may make up 50% of fish diets, depending on fish species, due to dispersal of amphipods from the vegetated habitats in which they are most abundant (Grimaldo et al. 2009b), or due to shifts in habitat associations during their life history. *Corophium*, the major genus of amphipods that showed up in smelt diets, are pelagic as juveniles, then become epibenthic tube dwellers as adults. Amphipods are found to make up a small proportion of Delta Smelt diets (Slater and Baxter in press), but a larger proportion than they did in the 1970s (Moyle 1992) before the declines in mysids and calanoid copepods.

Infauna –

Link to Invasive Clam model

Clams (native or invasive) make up the bulk of the biomass in the benthic community, with some tube-dwelling amphipods, ostracods, polychaetes, oligochaetes and tunicates, though species composition varies with location and dynamics of the salinity gradient (Peterson and Vayssieres 2010). True benthic infauna (as apposed to epibenthic organisms at the surface of the sediment), are generally unavailable to the open-water feeding salmon and smelt. Therefore, benthic infauna are more important as competitors for phytoplankton resources than as sources of food themselves. Invasive clams *Potamocorbula amurensis* and *Corbicula fluminea* can greatly reduce phytoplankton biomass and have been implicated in reductions in zooplankton biomass (See Invasive Clam Model) (Kimmerer et al. 1994, Feyrer et al. 2003). Benthic infauna is a major source of food for native Sacramento Suckers, Green Sturgeon, White Sturgeon, and non-native catfish and carp.

Epiphytic organisms

A variety of invertebrates live on substrate provided by aquatic plants, including some species of amphipods, and insects (Orsi and Mecum 1996). The BREACH study of marsh habitats in the delta found submerged aquatic vegetation and floating aquatic vegetation has higher biomass and diversity of amphipods and isopods than emergent aquatic vegetation (Simenstad et al. 1999). They also found juvenile Chinook salmon to eat a large proportion of chironomids, mostly associated with vegetated habitat (Simenstad et al. 1999). Many resident fishes have an even higher association with submerged or emergent vegetation due to the high density of invertebrates coupled with refuge from predation (Rozas and Odum 1988).

Pelagic Secondary Production

Microzooplankton

Bacterioplankton, rotifers, ciliates, and other microzooplankton are understudied and may be more important than generally realized (Bernhard et al. 2012). Bacterioplankton biomass was found to be an order of magnitude less than zooplankton biomass in one study of the SFE (Sobczak et al. 2005), but it is difficult to quantify zooplankton and bacterioplankton using the same methods. Another study of microzooplankton in Suisun Bay found that they may be the dominant component of biomass >2 µm in size, and they are a significant source of grazing pressure on phytoplankton (Rollwagen-Bollens et al. 2011). Bacterioplankton are important as decomposers, making detritus available to higher consumers. Many mesozooplankton species are omnivorous, consuming bacteria and protists as well as phytoplankton. In the Delta, the copepods *Eurytemora affinis* and *Sinocalanus doerri* appear to utilize detritally-borne microbial food sources (Durand 2008), though not as frequently as diatoms, when available (Orsi 1995). Mysids, isopods and amphipods also may rely on microzooplankton, particularly during time periods when primary production is depressed, such as cooler winter months (Durand 2008). As phytoplankton biomass has decreased since the bivalve invasions, this detrital/bacterial loop may be relatively more important to the pelagic ecosystem (Winder and Jassby 2011).

Zooplankton

Zooplankton, particularly calanoid copepods such as *Pseudodiaptomus forbesi* and *Eurytemora affinis*, are the dominant food resource for most open-water fish (Grimaldo et al. 2009b). Zooplankton species abundances follow seasonal patterns as well as being tied to specific salinities (Kimmerer and Orsi 1996, Hennessy and Enderlein 2013). However, both abundance and community composition of zooplankton has changed drastically over the past 30 years. Mysids were once a dominant component of the pelagic food web, but have decreased 10-fold over the past 30 years (Winder and Jassby 2011), and are now a minor part of pelagic fish diets (Grimaldo et al. 2009b, Slater and Baxter in press). These declines may be linked to the overall decline in phytoplankton productivity and shift in dominant phytoplankton species discussed above (see Phytoplankton section). A field study of dominant copepods in the low salinity zone of the estuary suggested chronic food limitation, possibly due to introduction of invasive clams and subsequent reduction in phytoplankton (Kimmerer et al. 2014).

The introduced cyclopoid copepod *Limnoithona tetraspina* has replaced the historically dominant mysids and calanoid copepods, making up 95% of the copepod individuals in some areas, though often only half the total copepod biomass due to its smaller size (Bouley and Kimmerer 2006, Winder and Jassby 2011). The size differential between *Limnoithona* and the next most abundant copepods (usually *Pseudodiaptomus forbesi* and *Eurytemora affinis*) make it a lower quality food resource for fish species of concern (Bouley and Kimmerer 2006, Gould and Kimmerer 2010). Fortunately, *Limnoithona* does not appear to compete directly with larger copepods for diatom food resources, preferentially consuming rotifers and protists (Kratina et al. 2014). Their ability to utilize microbial resources over phytoplankton may explain their rapid increase in abundance after their introduction in a period of low phytoplankton biomass.

Zooplankton abundance and community composition changes seasonally, with increases in early summer of Sinocalanus doerii and peaking in the late summer with large populations of *Pseudodiaptomus forbesi.* This is likely linked to the increase in temperature and increase in phytoplankton availability during the summer. Because Delta Smelt and Chinook Salmon are most abundant in the Delta in winter and spring when calanoid copepods are less abundant, they may be relying more on other species when they are there. Strong correlations between copepod availability and larval smelt diet during the summer have been documented, and while adult winter diets have been less well described, they include cycolopoid copepods, amphipods, cladocerans and larval fish (Baxter et al. 2013).

Tier 5: Target species

Salmon

Link to salmon model

Salmon fry and smolts may spend varying amounts of time in wetlands before completing their migration. Wetlands provide both food resources and refuge habitat for rearing salmon. Salmon will eat both zooplankton and benthic/epibenthic/epiphytic invertebrates common in vegetated wetland habitats (Merz 2001, Gray et al. 2002). They will use both the high-turbidity open water habitat and the vegetated intertidal habitat as refuge from predation as well as foraging (Gregory 1993, McLain and Castillo 2010, Ferrari et al. 2014). See Salmon model for more information.

Delta Smelt

Link to Delta Smelt model

Delta Smelt are expected to benefit from restoration chiefly in the export of productivity (especially phytoplankton and calanoid copepods) to the surrounding environment (Nobriga 2002, Baxter et al. 2013, Slater and Baxter in press). Delta Smelt diets are comprised mainly of copepods, particularly the introduced *Pseudodiaptomus* (Slater and Baxter in press), so the predicted increase in particulate organic matter, phytoplankton, and zooplankton during wetland restoration is predicted to increase the abundance of Delta Smelt along with the rest of the POD species (Striped Bass, Longfin Smelt, and Threadfin Shad) (Baxter et al. 2010). However, Delta Smelt do consume some epibenthic or epiphytic invertebrates such as amphipods (discussed above).

Because most Delta Smelt migrate from spawning areas in freshwater (winter-spring), to rearing areas in brackish water (summer-fall), the extent to which they will benefit from increased export of food from wetlands will change with the seasons (Sommer et al. 2011, Murphy and Hamilton 2013). Given lower phytoplankton production in the winter, they may be benefiting more from detrital pathways (see Decomposition). They are expected to benefit more from wetlands in the Delta during spawning season through both food availability and increase in spawning habitat, whereas in the summer and fall they may derive more benefit from restoration sites in Suisun Marsh as a source of local food resources.

Several current wetland restoration plans suggest that unvegetated shallow subtidal areas may provide valuable Delta Smelt habitat if high connectivity is established between shallow open-water habitat and high-productivity vegetated marsh. Such connectivity should maximize benefits of wetland restoration for Delta Smelt (see MAST delta smelt model for more information).

Longfin Smelt

Longfin Smelt are ecologically somewhat similar to Delta Smelt, though they tend to occupy higher salinities. Therefore, they are expected to benefit from wetland restoration in similar ways, chiefly in the export of productivity (Baxter et al. 2010). Longfin Smelt are currently understudied so their response to wetland restoration is even more uncertain than that of Delta Smelt.

Other native fish

While most are not currently listed as threatened or endangered, wetland restoration is expected to benefit native fish species, such as Tule Perch and Splittail, through increases in habitat and on-site and exported food resources (Grimaldo et al. 2012, Herbold et al. 2014). As with species of concern, the magnitude of such benefits will be affected by the abundance of non-native species. It is important to monitor species that are currently not threatened because human population growth, anthropogenic impacts, and climate change may cause them to become threatened in the future (Moyle et al. 2013).

Competitors

While most native fish assemblages have evolved over tens of thousands of years to balance competitive interactions, the novel fish community may include competitive interactions that could contribute to at-risk fish population instability. As discussed above, introduced clams pose a serious competitive threat to zooplankton in the SFE. Introduced planktivorous fish and invertebrates that may compete with directly with Delta Smelt (and potentially prey on early life-history stages), include Black Sea jellies (*Maeotias marginata*), Oriental shrimp (*Palaemon macrodactylus*), Siberian Prawn (*Exopalaemon modestus)*, Mississippi Silversides, juvenile Striped Bass, Threadfin Shad and American Shad, to name a few of the most abundant taxa (Feyrer et al. 2003, Moyle et al. 2014). Proving detrimental effects of competitive interactions is difficult, so while dietary and habitat overlap between Threadfin shad and Delta Smelt makes shad the most likely competitor for smelt, the strength of competition has not been well quantified (Feyrer et al. 2003).

Tier 6: Predators

Predatory fish (especially non-native ones)

Many introduced fish are hypothesized to exert significant predation pressure on at-risk species in the system. Centrarchids in particular are voracious predators and expected to benefit from the shallow water, vegetated habitat created during wetland restoration. However, these are economically valuable game fish, so their control and/or eradication may be contentious. Control of any fish in a large open system like the SFE would be impractical even if it were politically feasible.

Largemouth bass in particular are more commonly found in submerged aquatic vegetation with dense canopies, such as *Egeria densa*, and feed primarily on fish and invertebrates common in SAV (Nobriga and Feyrer 2007). Fortunately, thicker vegetation may lower predation success of this species as habitat complexity provides refugia for their prey (Rozas and Odum 1988, Ferrari et al. 2014). This may be especially helpful for juvenile salmonids, which may use vegetated habitat as a predation refuge (Kocik and Taylor 1996, Beland et al. 2004).

Striped Bass, introduced as a game fish in 1879, are particularly well-studied. Young of year Striped Bass were part of the Pelagic Organism Decline along with Delta Smelt, Longfin Smelt and Threadfin Shad, so are under many of the same pressures (Baxter et al. 2010). Striped bass share open-water habitat with Delta Smelt, so have the potential to be a major factor in the decline of the species. While age-0 striped bass declined at the same time as Delta Smelt and the other POD species, age 1-3 Striped Bass are the most likely predators of smelt (Baxter et al. 2010). However, there are few studies testing the extent to which Striped Bass prey on Delta Smelt , and no correlations have been found in adult or juvenile Striped Bass abundance and Delta Smelt abundance(Nobriga et al. 2013). An analysis of Striped Bass diets versus prey availability predicted that rare species, such as Delta Smelt, are less likely to be incorporated into Striped Bass diets (Nobriga et al. 2013). While predation from Striped Bass may be a factor in the decline of Delta Smelt, it is not the only factor (Baxter et al. 2013).

While Striped Bass and Largemouth Bass are the predatory fish most commonly referred to as potential controls on smelt populations, other introduced aquatic species such as catfish are a growing concern. Introduced jellyfish, shrimp, and small fish such as and Mississippi Silversides are all potential predators on larval Delta Smelt (Baerwald et al. 2012). Unfortunately, the frequency of predation on at-risk fishes for many of these species is still unknown.

Native predatory fish, chiefly the Sacramento Pikeminnow, prey on species of concern (Nobriga and Feyrer 2007), so restoration action that favors native species may also favor these native predators. It is unclear whether pikeminnow exert significant predatory control on fish of concern in the SFE, but they are perceived as a major source of predation on salmonids in the upper Sacramento watershed, where various measures to control their populations at damns have been used historically (Moyle 2002).

Birds

Major piscivorous bird groups include herons and egrets (Ardea), diving ducks, and cormorants. Diving ducks have been shown to locally decimate clam populations, allowing phytoplankton and zooplankton to recover (Poulton et al. 2002). Wading birds forage in shallow open-water and partially vegetated habitat, where they may prey on both native and non-native fish.

While there have been few analyses of the effects of birds on fish of concern in the SFE, there have been several experiments and meta-analyses of other systems, with mixed results. One experiment excluding birds from a Midwestern stream found large increases in prey abundance, (Steinmetz et al. 2003); however, an analysis of bird predation on migrating salmonids in the Columbia River found a negligible impact on juvenile salmonid populations (Wiese et al. 2008). Most attempts to evaluate the impact of piscivorous birds on fisheries have been hampered by the difficulty in scaling up a small-scale diet analysis to the large, and often poorly estimated, populations of both birds and fish of concern (Harris et al. 2008).

Other predators

The most common mammalian piscivore in tidal wetlands of the SFE is the American river otter. Otters are opportunistic predators, generally catching fish in proportion to their abundance and inversely proportional to their swimming speed (Cosby 2013). While otters may consume large numbers of salmonids during migration (one study estimated two otters ate over 8000 juvenile salmonids in a six-week period) (Dolloff 1993), this is most likely to occur when salmonids are present in high densities (4 N/m2). When salmonids are present in lower densities, otters are more likely to prey on slower fish such as cyprinids or sculpins, and crustaceans (Cosby 2013). The only known study of otter diet in the SFE found crayfish to make up the majority of their diet (Grenfell 1974). Given the relatively low density of fish of concern and the high availability of slower, resident fish and crustaceans, otters are unlikely to have a large predatory impact on Chinook Salmon, Delta Smelt, or Longfin Smelt.

Conclusion:

How might this model inform monitoring of restoration sites? (this section may have to wait until later in the workteam process.)

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Interagency Ecological Program

Tidal Wetlands Restoration Monitoring Project Work Team

Aquatic Vegetation Conceptual Model

DRAFT

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PREFACE

This Conceptual Model is part of a suite of conceptual models developed by the IEP Tidal Wetland Monitoring Project Work Team to guide monitoring of tidal wetland restoration sites within the Sacramento-San Joaquin Delta (Delta) and Suisun Marsh. The conceptual models have been developed based on the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) models, and are designed to aid in the identification and evaluation of monitoring metrics for tidal wetland restoration projects. These models describe our understanding of how tidal wetland restoration may benefit at-risk fish species.

Contents

Introduction

Tidal wetlands are defined by emergent vegetation uniquely adapted to sheltered intertidal zones, transitional habitats that connect the terrestrial habitat with open water (Mitsch and Gosselink 1993). However, tidal wetlands include much more than marsh plains populated with emergent, hydrophytic plants. They are mosaics of varied interconnected habitats, which may include shallow open water, subtidal and intertidal channels, vegetated marsh plains, mudflats, salt pannes, and upland transitions (Mitsch and Gosselink 1993). In the Sacramento-San Joaquin Delta of California (Delta), 97% of historic tidal wetlands have been lost due to land reclamation (Whipple et. al. 2012), which may be limiting resiliency of native fish populations, particularly Delta Smelt (*Hypomesus transpacificus*), Longfin Smelt (*Spirinchus thaleichthys*), and Chinook Salmon (*Oncorhynchus tshawytscha*). Multiple mandates are now in place to restore tidal wetlands throughout the Delta and Suisun Marsh (USFWS 2008, NMFS 2009, CDFG 2009BCDC 2011, CalEcoRestore 2015). Aquatic plants play an influential role in structuring ecosystem processes and habitats within these systems.

Wetland aquatic vegetation (AV) is categorized into emergent, submerged and floating macrophytes, and macroalgae (Mitsch & Gosselink 2000). Delta communities for all AV types are a mix of native and non-native species (Appendix A). Emergent vegetation such as sedges, rushes, and some broadleafed forbs exist along the shoreline and are dominated by cattails *(Typha spp.)* and “tules” (*Schoenoplectus* spp.), but invasives such as common reed (*Phragmites australis)* and giant reed (*Arundo donax)* dominate in some areas (Hickson et al. 2007). Species extending from the shoreline across the water include submerged and floating vegetation communities highly dominated by invasive species. The submerged aquatic vegetation (SAV) community is dominated by non-native Brazilian waterweed, *Egeria densa* (Santos et al. 2011). *Egeria* is a notorious ecosystem engineer because of its ability to alter its surrounding physical environment (Jones et al. 1994) and is a highly invasive species (Yarrow et al. 2009). The floating aquatic vegetation (FAV) community is dominated by two non-native species: water hyacinth (*Eichhornia crassipes*) and water primrose (*Ludwigia spp*) (Khanna et al. 2012b). The invasive forms of aquatic vegetation are highly prolific, causing major changes in large-scale physical processes (Hestir 2010b, Hestir et al. 2015), which in turn have improved habitat suitability for invasive fish species while allowing continued habitat degradation for native, endangered fishes (Brown and Michniuk 2007).

The vegetation community changes across the salinity gradient, with lower species diversity in Suisun Marsh than in the Delta. Emergent vegetation in the Marsh is also dominated by *Schoenoplectus* spp. with a *Phragmites* and *Arundo* as common invaders (Whitcraft et al 2011, Vasey et al 2012). The submerged vegetation is dominated by the native *Stuckenia* rather than the mix of invaders found in the Delta (Boyer and Sutula 2015). There is no floating vegetation tolerant of the higher salinities (Boyer and Sutula 2015).

Certain principles and goals lead the development of this tidal marsh aquatic vegetation conceptual model:

- Tidal wetland restoration efforts must consider the environmental factors that influence and control aquatic macrophyte establishment and growth in order to discourage colonization by undesirable species while allowing establishment of favorable species that encourage wetland ecosystem processes (e.g., peat accumulation) to occur.
- The model encompasses invasive and native aquatic macrophyte species, and emergent, floating, and submerged growth forms of aquatic vegetation (AV). Algae, including phytoplankton and periphyton, are discussed in the Tidal Wetland Food Web Conceptual Model.
- The model depicts the broad relationships between the landscape surrounding a restoration site, habitat characteristics for AV on a restoration site, as well as the feedbacks between AV and physical and biological aspects of the habitat.
- The model informs monitoring programs for AV at tidal restoration sites in the Delta and Suisun marsh, but does not discuss fully saline marshes typical of the San Francisco Bay.
- The model builds on existing Delta research and references previous models such as those developed to support Delta restoration under the DRERIP.
- The description of the model is not intended to be an exhaustive literature review of Delta AV, but instead includes salient publications that help predict various AV trajectories and ecosystem responses at Delta tidal wetland restoration sites.

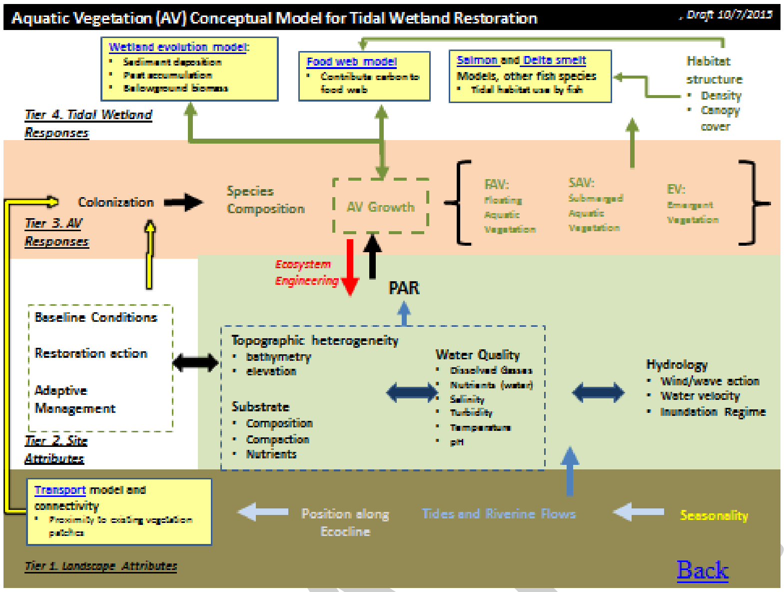

Figure 1. Aquatic vegetation conceptual model diagram.

Model Structure

Aquatic macrophyte cover and species composition in any given location are the product of multiple interacting environmental factors that exist at multiple scales. To depict these factors in a conceptual model, we organized the model into multiple tiers that reflect different spatial scales (Figure 1).

The inspiration for this tiered model is a similarly organized conceptual model for Delta Smelt biology, presented in IEP-MAST 2015, “An updated conceptual model for Delta Smelt: our evolving understanding of an estuarine fish” (IEP-MAST 2015). Inspiration was also derived from the DRERIP modeling approach, draft vegetation conceptual models (Anderson 2008), and a recent review on SAV and FAV completed for the Central Valley Regional Water Quality Control Board (Boyer and Sutula 2015).

This model and its companion models share several conventions for the accompanying diagrams. Site attributes within dotted lines have strong interactions and are categorized by type (e.g., water quality, substrate). Arrows between attribute categories represent general linkages, though for the sake of simplicity, not all linkages between specific attributes are represented by arrows. Thickness of arrows generally describes hypothesized importance;

however, arrows do not indicate certainty or strength of effect, as is the convention in the DRERIP models. Thick arrows moving between tiers that do not connect to specific factors represent relationships with all attributes in the following tier. Arrow and text color is used to aid in following arrows and does not convey meaning. Yellow boxes are links to sub-models which provide greater elaboration on the specifics of particularly important processes, structures, or species.

The AV Conceptual Model has 4 tiers:

Tier 1, Landscape Attributes: These are factors acting on the scale of the entire San Francisco Estuary and include aspects of habitat that vary with the geographic placement of a tidal wetland.

Tier 2, Site Attributes: These are factors acting on the scale of a specific tidal wetland. These include water quality and physical structure attributes of the tidal wetland as well as local hydrodynamic processes.

Tier 3, Aquatic Vegetation Responses: This tier describes the response of the aquatic vegetation community to both landscape and site attributes, including the feedback loop between aquatic macrophytes and site attributes (AV functions as an ecosystem engineer).

Tier 4, Tidal Wetland Responses: This final tier describes outcomes for tidal wetland ecosystem processes that are influenced by AV. It is at this tier that the AV model links to other sub-models within the tidal wetland conceptual model suite.

Tier 1: Landscape Attributes

Seasonality

Many processes that are important to vegetation change seasonally. Riverine flows are highest in the late winter and early spring. Temperature is highest in the summer (see Temperature, tier 2). Photoperiod, angle of the sun, and cloud cover change seasonally, causing reduction in amount of light reaching the vegetation during the winter (see PAR, Tier 2). These factors lead to seasonal peaks in vegetation biomass, which may differ by species and growth type (see AV Growth, tier 3).

Tides and Riverine Flows (Landscape-scale hydrodynamic processes)

Tides and river flows in the Delta are two major factors that control landscape-scale hydrodynamic processes. Tidal processes counteract river inflows, temporarily changing the direction of flow during flood tide. Tides also cause sub-daily fluctuations in water stage, interacting with site elevation and bathymetry to result in a given inundation regime at local scales (see Bathymetry and Inundation Regime, Tier 2), which in turn influences the AV species that can establish at a particular site.

Riverine flows influence AV directly by controlling the freshwater flow regime (rate and degree of freshwater input on both daily and seasonal temporal scales) and indirectly by influencing water quality factors such as salinity, sediment inputs, and nutrient concentration. The relative importance of these direct and indirect influences on AV community composition varies with degree of riverine inflow and wetland-river connectivity. In wetlands with frequent exposure to high flows, the direct effect of flow surpasses the indirect effects of water quality on community composition, as adaptations to withstand flooding events or to recolonize quickly afterwards are favored. In wetlands with a more stable, low flow regime, indirect effects of water quality may be more important in structuring community composition. Rooney et al. (2013) describe these differences in AV community composition as flood response 'guilds' that may help to explain differences between wetlands. In addition to the typical flow regime of a given wetland, periodic extreme high flow events (floods), may uproot SAV, push FAV out of the system, and increase the extent of fringing emergent vegetation (Lacoul and Freedman 2006).

In the Delta, freshwater flows come largely from the Sacramento River, as well as relatively small inputs from the San Joaquin, Calaveras, Cosumnes, and Mokelumne Rivers (See Transport and Tidal Wetlands Overview Models). These riverine inputs vary with respect to volume, sediment load, AV species composition (as propagule source), and productivity and thus are likely to have varying effects on AV colonization and establishment in the Delta, including tidal wetland restoration sites. Connectivity between wetland restoration areas and freshwater inputs will influence the potential AV colonizer species, and local hydrodynamics and water quality within restoration areas will influence the AV species that become established.

Sea level rise and changes to freshwater flows caused by precipitation changes and water management have already resulted in shifts to the vegetation community around the estuary (Watson and Byrne 2012), and this pattern is expected to continue with climate change. With increasing land area restored throughout the Delta, salinity may intrude further inland, exacerbating this trend (RMA 2013). Changes to plant communities may be a more powerful indicator of long-term trends in salinity or sea level rise than direct measures of water level because they change more slowly than tidal and seasonal variability in water levels.

Position on the Ecocline

In this set of conceptual models, we have defined two "ecoclines." First, a longitudinal ecological gradient from the Golden Gate to the headwaters of the rivers. Secondly, a lateral gradient from the center of the river channel to the furthest upland zone perpendicular to the first ecocline. The geographic location of tidal marsh within the Delta landscape dictates its proximity to various freshwater inputs, distance from the ocean and other tidal marshes, position relative to the estuarine turbidity maximum, as well as the degree of tidal forcing. The turbidity maximum, usually occurring somewhere between Carquinez Straight and Mallard Island between 1-6 psu, will shift landward or seaward depending on tidal stage, bottom topography, and river inputs (Schoellhamer and Burau 1998, Schoellhamer 2000). These landscape-scale factors will influence AV-relevant physical and water quality characteristics (particularly the salinity and sediment regime) at the local scale, as well as the types and species of AV that are available to enter and become established at a particular tidal wetland.

Transport and Connectivity

The position of a tidal wetland along the ecocline, together with tidal and riverine influences, informs the degree of connectivity of a particular wetland to existing AV patches and thereby the species likely to colonize newly restored wetlands. This hydrologic connectivity is essential for understanding and predicting colonization, as initial colonization of created wetlands typically occurs via above-ground dispersal of propagules rather than vegetative spreading from other areas (Capers 2003). This concept of connectivity of wetlands to AV patches is also dependent on propagule size for individual AV species. For example, water hyacinth can reproduce from fragments as small as one inch (Penfound and Earle 1948), while other species may require larger fragments or specific plant parts for colonization. Species with seeds that spread easily by wind, such as *Typha* or *Phragmites*, may spread to areas without direct hydrologic connectivity to the site (Lacoul and Freedman 2006). For more details on processes important to transport of materials within the estuary, see the Regional Transport model.

Tier 2: Site Attributes

Baseline Conditions

Antecedent conditions of a wetland restoration site may influence the colonization, establishment, and growth of macrophytes even after the site is returned to tidal action. Prior land uses, particularly agriculture, may have led to subsidence of the restoration site and resulting elevations/inundation regimes that are unsuitable for emergent vegetation establishment. Soil compaction, which often accompanies subsidence, hinders underground vegetative growth in general and specifically excludes species not adapted to stressful environments (Sloey et al. 2015). In addition to these physical alterations of topographic heterogeneity and substrate (both more thoroughly discussed below), agriculture and other dry-land uses deplete the seed bank of wetland species, making a restoration site dependent on import of seeds and vegetative fragments ("seed rain") for any natural establishment of vegetation (Zedler 2000, Diggory and Parker 2011). In contrast, a restoration site that has pre-established wetland vegetation will also have a viable seed bank, though it will also often have a high proportion of non-native propagules (Zedler and Kercher 2004).

Restoration Action

The actions taken to restore a tidal wetland vary considerably, according to the goals and resources of the project. One project's restoration actions may constitute a levee breach followed by allowing nature to take its course, while another project may involve moving significant amounts of soil as channels are dug out or elevations are raised to the intertidal level. Biotic and chemical interventions to influence the eventual plant community of the restored wetland may or may not be undertaken. Restoration actions intended to influence the vegetation community of the site generally fall into four categories: 1) physical site structuring; 2) pre-construction invasive vegetation control; 3) planting, and 4) post-construction vegetation control. Although this conceptual model is part of an effort to guide monitoring of restoration sites rather than their design, it is important to understand what restoration actions have been taken in order to predict vegetation responses.

Physical restoration plan

If emergent marsh is a goal of tidal wetland restoration and the site is subsided, site grading and/or the addition of fill material to bring elevations into the intertidal range may be important design elements. In some cases, this is achieved through the beneficial re-use of dredge material, as is being done in the Montezuma Wetlands Restoration Project in Suisun Marsh. Attempts to exclude invasive submersed vegetation from a site may also include earth moving, as channels are sized to promote peak tidal velocities higher than the tolerances of establishing propagules (Koch 2001).

Pre-construction invasive vegetation control

Existing wetlands that are slated for restoration are often plagued with non-native invasive plants, such as *Phragmites*, *Eichhornia*, and *Egeria*. Invasive species tend to alter habitat structure, reduce biodiversity, and change nutrient cycling in a wetland (Zedler and Kercher, 2004), so eradicating or controlling these plants prior to opening the site to tidal action is desirable. Strategies most often used include drying the site, mowing, grazing, burning, and applying herbicides, or some combination thereof. However, none of these methods extract the seed bank, and most highly invasive species also reproduce vegetatively from small fragments, particularly in open disturbed environments (Zedler and Kercher, 2004). Additionally, invasive vegetation pervades the estuary (Underwood et al 2006, Santos 2011, Khanna et al 2012), so re-introduction following the establishment of hydraulic connectivity of the restoration site is highly likely.

Planting

Unlike FAV and SAV, emergent vegetation is often a desired goal of restoration design. Although wetland vegetation can establish rather quickly in a restored wetland without human intervention ("self-design," Mitsch et al. 2012), planting may be included in restoration plans in order to more quickly establish emergent vegetation, increase plant diversity, and/or pre-empt colonization by invasive species. In a meta-analysis of invasive plant control experiments, Kettenring and Adams (2011) found that successful control of invasive species is often not sustained unless desirable native species are planted to occupy the newly open niche.

Work in brackish restored tidal marshes in the Napa River demonstrated that seed rain is capable of providing enough propagules to revegetate emergent vegetation in newly restored wetlands, but species diversity was lower in restored sites than in reference sites (Diggory and Parker 2011). Rare species were poorly represented in seed rain, even within marsh areas where they were present, indicating dispersal limitation. Restoration practitioners concluded that planting is needed if high species richness is a goal (Zedler 2000, Diggory and Parker 2011). Experimental comparison of planted and self-design wetlands over 15 years resulted in a similar pattern of lower diversity in unplanted restoration sites (Mitsch et al. 2012). If planting is to be done, work on Liberty Island showed that transplants of adult *Scheonoplectus* and *Typha* establish more successfully than rhizomes, particularly in compacted soils (Sloey et al. 2015). Other restoration actions, such as addition of nutrients, construction of tidal channels, and planting seedlings in tight clusters can increase the establishment and growth of vegetation in restoration sites (Obrien and Zedler, 2006).

Invasive Species Management

Changes to the restoration site after it is built may create conditions that favor certain species over others at different times; thus, invasive vegetation control is well-suited to an adaptive management approach. The California Department of Boating and Waterways is authorized to regularly spray herbicides for control of *Eichhornia*, spongeplant (*Limnobium laevigatum)*, and *Egeria* through CA Harbors and Navigation Code Section 64. Other invasive species, such as *Ludwigia spp.*, *Potamogeton crispus*, and *Myriophyllum spicatum* are currently being evaluated using a formal risk analysis tool , which will determine if formal control programs should be developed for these species in the future (Gordon et al. 2012, V. Cook-Fletcher, CDFW, personal communication). Chemical treatment of other invasive vegetation by project proponents may require additional permitting. Mechanical control is also used to some extent and may be feasible on a confined restoration site (Greenfield et al. 2006). On the landscape scale, active research into the introduction of biocontrol agents may help combat the invasion of *Egeria*, *Eichhornia*, and *Arundo (*Paul Pratt, USDA pers. comm., see Consumers, below)

Hydrology

Inundation Regime

Landscape level hydrologic processes (see Tier 1, above) and elevation (see Elevation, below) influence the site's local inundation regime. In tidal freshwater systems, the frequency and duration of inundation is thought to be the dominant factor influencing aquatic plant species distributions (Odum et. al. 1984). Species composition varies from the upland edge to the open water. Emergent, submerged, and floating vegetation tend to be distributed along an inundation gradient and finer scale zonation patterns are noted within these communities for specific vascular plant species (Figures 2 and 3, see Elevation below, and Species Composition, Tier 3). For example, a study of vegetation establishment in Liberty Island in the North Delta found inundation regime to control the success of experimental plantings (Sloey et al. 2015).

Water Velocity and Wave Stress

Water velocity at a site is influenced by wind waves, riverine/stream discharges, daily tidal cycles, and the interaction of these factors with the site's topography. For example, a site with a constricted breach will experience a very different velocity regime when compared to a total levee removal design. Large restored marsh systems with distributary channels that mimic natural marshes allow for drainage and provide enhanced tidal exchange. Sites with dendritic channels can funnel the water into the tidal plain and create velocities that influence where seedlings can colonize substrates (Hoover 1983). Further, current velocities vary over seasonal and daily timeframes, creating alternate scour and sedimentation events that affect plant establishment and growth. Characteristics such as fetch, shoreline configuration, and wind strength and direction affect the wave climate and turbulent flows experienced by colonizing aquatic vegetation (Knutson et. al. 1981).

Water velocity and wind-wave stress influence emergent and submerged vegetation colonization rates (Hudon et al 2000). Propagules must be able to stay in place long enough to put down roots, which is often impossible in high water velocities. At Liberty Island, emergent vegetation is becoming established more rapidly in sheltered locations away from the downstream outlet (Sloey et al 2015). Similarly, SAV establishment in the Delta appears limited above 0.49 m/s (Hestir 2010). Extremely high flows may uproot submerged vegetation, but act chiefly to flush senescent material out of vegetation beds and make it available to detritivores in the rest of the system (Lacoul and Freedman 2006). Individuals of the same species growing in different microhabitats may exhibit different morphologies, growing taller, thicker and at higher biomass in sheltered areas (Thouvenot et al. 2013). However, more epiphytic algae also grow in areas with low wave stress, causing light limitation and reduced macrophyte growth (Strand and Weisner 2001).

Floating vegetation, such as *Eichhornia*, is similarly affected by high velocities. FAV often anchors itself to the edge of emergent vegetation along the banks, but may be flushed out of the system in high flow years (Khanna et al 2012, Boyer and Sutula, 2015).

Vegetation also affects water velocity (see Ecosystem Engineering, below), so plant establishment creates a feedback loop which slows water velocities and facilitates additional plant colonization.

Monitoring of velocities within the site may elucidate relationships between site topography and plant communities. If tidal channels clog with invasive floating or submerged vegetation, velocities may be too low. If emergent vegetation fails to establish on the marsh plain, velocities or wave stress may be too high. Either of these situations may call for corrective management actions.

Topographic Heterogeneity

Bathymetry:

Topographic heterogeneity may be key to development of functional restoration sites, due to the influence of topography on both water velocity and soil drainage. Sites with low topographic relief may fail to establish vegetation years after restoration (Brooks et al 2015). Fine-scale topography and bathymetry within a given elevation zone, particularly drainage patterns and the presence of tidal creeks, may be as important as position on the ecocline in determining plant diversity, due to the effect of drainage patterns on salinity (Morzaria-Luna et al 2004, Culberson 2001). In brackish and saline tidal marshes, drainage patterns within marsh plains, including presence of channels and creeks, may cause localized areas of lower salinity where rainwater regularly washes out the channels (Figure 2). In contrast, the areas further from the channel may accumulate salts in the soil, causing the vegetation to be dominated by halophytes (Culberson 2001). However, in areas with high drainage, soil salinity will decrease further from the waterline, resulting in the most salt-tolerant plants on the edge of the channel (Odum 1988). In freshwater marshes, root zone inundation, rather than salinity, has a dominant influence on zonation patterns (Figure 3, more below).

Elevation and Depth

The areal extent of the intertidal zone is determined by elevation, slope and tidal range. Inundation regime interacts strongly with elevation to produce the observed emergent vegetation zonation pattern within the intertidal zone. At the upland edge of the intertidal zone, plant community composition is determined primarily by competition for light, whereas at the lower edge of the intertidal zone, community composition is determined by abiotic factors (Culberson 2001, Crain et al. 2004, Bornette and Puijalon 2011, see Competition, below). Species with greater ability to withstand root-zone anoxia caused by prolonged flooding will dominate the lower end of the intertidal zone, particularly in freshwater wetlands (Grace 1999, Lenssen et al. 1999, Lenssen et al. 2003).

In salt marshes, depth interacts with salinity in driving the observed intertidal plant zonation (see Salinity, below, and Species Composition, Tier 3), while in freshwater marshes zonation is much less pronounced, emergent plants occupy the entire depth of the intertidal area, and there is a strong degree of overlap between communities (Figures 2 and 3, Odum 1988). As depth increases, emergent plants are inhibited by increasing anoxia of the root zone and decreasing light availability.

At subtidal depths, submerged plants replace emergent plants as the dominant macrophyte growth type. Depth interacts with turbidity to control zonation of SAV, with fewer plant species tolerating the low light availability of deeper depths (Chambers and Kaiff 1985, Hudon et al. 2000, Madsen 2013, see PAR below). Aquatic macrophytes are rarely found at depths greater than 4 meters in any system (Bornette and Puijalon 2011), and in the Delta, Durand (2014) found a low probability of *E. densa* colonization below 5 m. Areas with higher turbidity may further reduce this maximum depth (see Turbidity, below).

Truly floating vegetation is not limited by depth (Boyer and Sutula, 2015), though rooted forms with floating leaves, such as *Ludwigia*, are limited by the same factors as submerged plants until their leaves reach the water's surface (Thouvenot et al 2013),

Substrate

Composition

The physical and chemical properties of the substrate are important factors in determining whether emergent and submerged rooted plants colonize and grow. Soil properties such as soil texture, structure, consistency, bulk density, and permeability influence vegetation colonization and growth (Bornette and Puijalon 2011). Delta soils are composed of mineral sediments from riverine inputs and of peat derived from decaying marsh vegetation.

For emergent plants growing in the intertidal zone, drainage and aeration of the root zone are often the most important factors for plant establishment. Poor drainage in brackish water marshes caused by dense clay soils, may lead to buildup of salt in the soil and osmotic stress in the plants. Higher percentages of sand in the soil may increase drainage and decrease likelihood of salt stress (Zedler 1988). Even in freshwater marshes, salts may buildup in areas of the high marsh with poor drainage. Furthermore, low oxygen in the root zone of waterlogged marsh soils can limit establishment of emergent plants lower in the intertidal zone (Bornette and Puijalon 2011). To combat this, most plants adapted to wetland habitats have aerenchyma (spongy tissue in the stems that allows gas exchange between roots and shoots) or other adaptations for growing in waterlogged soils.

Because substrate for submerged plants is always waterlogged, substrate stability is more important than drainage for colonization of these species. Larger grain size in the sediment (sand, rather than silt), will cause lower bed stability and may lead to less SAV establishment (Boyer and Sutula 2015), However, there is a feedback loop between SAV establishment and silt, since SAV slows water velocities, causing silt to drop out of suspension and allowing more SAV establishment (see Ecosystem Engineering, below).

Compaction:

Sediment compaction and a lack of substrate heterogeneity at the onset of restoration may slow tidal marsh vegetation establishment and create marshes that never develop the heterogeneity of natural tidal marshes (Brooks et al. 2015). This is particularly important for restoration of flooded islands where soils may have been compacted by years of agriculture. For example, on Liberty Island, a study of emergent vegetation found greater vegetation expansion in areas with a deeper layer of non-compacted soil (Sloey et al. 2015). The dominant emergent plant species was also influenced by soil compaction, with *Schoenoplectus spp*. more able to tolerate the stressful combination of high soil bulk density and long inundation time than *Typha spp*. (Sloey et al. 2015).

Nutrients in sediments:

Nutrients and their availability in wetland sediments are highly variable. The oxidation state of soils affects rates of decomposition and the form and availability of various nutrients. Oxygen depletion in soils leads to lower redox potential. Under these conditions, soil nitrogen tends to be in the form of ammonium, and the availability to plants of phosphorus and various metals is increased (Mitsch and Gosselink 1986, Bornette and Puijalon 2011).

The low redox potential of anaerobic soils can also lead to potentially toxic conditions. For example, reduced forms of iron and manganese are more soluble (and thus mobile) and are potentially toxic to plants. Sulfide is produced by anaerobic sulfate-reducing microorganisms when oxygen is not available; sulfides are highly toxic and can be found at sufficient levels to exclude plant establishment. Some toxicity can be prevented through oxygenation of the rhizosphere, particularly in species with aerenchyma (Lambers et al. 1998).

Similar to soil oxidation state, the availability and form of nutrients is also affected by pH. Low pH in the rhizosphere can increase the availability of some required nutrients (e.g., zinc, manganese, and boron), but can also mobilize potentially toxic ions, such as aluminum (Lambers et al. 1998).

Emergent plants acquire most of their nutrients from the sediment, while submerged plants may acquire nutrients from both the water column and the sediment (Lacoul and Freedman 2006). Relative root versus shoot absorption differs by nutrient and plant species, and even within a species, studies differ as to whether SAV prefers nutrients in the water column or sediment (Barko et al. 1991, Boyer and Sutula 2015). Further study is necessary to uncover how these dynamics play out in the Delta.

Floating plants rely entirely on nutrients in the water column; however, nutrient release from sediment, particularly with low dissolved oxygen layers that can occur beneath FAV, may make sediment nutrients available to FAV in the water column. (Bornette and Puijalon 2011).

Water Quality

Turbidity

Broad-scale patterns of turbidity within the estuary are explained by the site's location relative to the estuarine turbidity maximum (see Position on the Ecocline, above). The turbidity at a site is controlled by the amount of sediment, dissolved organic matter, and phytoplankton suspended in the water column, which varies with position along the lateral and longitudinal ecoclines. Sediment input from riverine and tidal deliveries of highly turbid water is augmented by resuspension of sediments from the bottom by waves in shallow open water areas (See Wetland Evolution Model and Schoellhamer et al. 2012 for more on suspended sediment).

Turbidity influences the establishment and growth of submerged aquatic vegetation in the water column by reducing penetration of light available to colonizing species (Bornette and Puijalon 2011, see PAR below). In general, environments with higher turbidity have fewer submerged aquatic plants, both in terms of number of species and biomass. Turbidity and depth interact to affect light availability, so submerged aquatic plant zonation will be determined by both turbidity and depth (Canfield et al. 1985, Tanner et al. 1993, Skubinna et al. 1995). In extremely turbid environments, turbidity may be more important than depth in determining plant biomass (Scheffer et al. 1992). High turbidity environments often favor floating vegetation, species with floating leaves, or caulescent forms of submerged vegetation which are able to produce erect stems that grow towards the water surface (Bornette and Puijalon 2011).

Emergent and submerged vegetation also affect turbidity as they become established (see Ecosystem Engineering, Tier 3).

Water Temperature

Water temperature in the Delta is controlled chiefly by air temperature, though water's high specific heat means the realized water temperature is much less variable in aquatic systems than terrestrial systems (Bornette and Puijalon 2011). In the Mediterranean climate of the Delta, the water does not freeze in winter and becomes quite warm in the summer, generally staying within the range of 10-25 C (Wagner et al. 2011).

While increases in temperature in terrestrial systems may lead to water stress, in aquatic ecosystems many submerged plants show optimal growth rates at temperature 28-35 C (Bornette and Puijalon 2011). Many of the invasive plants in the Delta are tropical species, so warm water may greatly increase their growth rates (Wilson et al. 2005, Yarrow et al. 2009a). In future climate change scenarios, there is likely to be an increase in both maximum and minimum temperatures (Wagner et al. 2011), potentially favoring tropical invasive weeds over native plants and increasing total vegetation biomass (Bornette and Puijalon 2011). It is important to note, however, that increased water temperature will interact with other environmental factors to affect the plant response. For example, one study found *Egeria* to decrease in growth at warm temperatures (26 and 30 C) when salinities were also elevated (Borgnis and Boyer 2015).

Most species of emergent plants, and some submerged plants, die back seasonally during cold weather. While hard frosts in the Delta are rare, they are one of the few natural control measures on growth of *Eichhornia* (Wilson et al. 2005). Recent expansion of *Eichhornia* in the Delta may be tied to low occurrence of frost during recent years (Boyer and Sutula 2015).

Salinity

The position along the ecocline, tides, riverine flows, water management, and evaporation combine to affect each site's salinity characteristics and in turn the types of vegetation that will colonize and develop at the site. Before major dams were built upstream, the salinity interface migrated as far upstream as Courtland along the Sacramento River (California Department of Water Resources, 1993); however, summer releases of freshwater from dams reduces the landward migration of the salinity interface during most years. Tidal marshes can be classified as brackish near Suisun Bay and become increasingly fresh as you move from Suisun Bay to upstream riverine systems.

Average salinity determines the dominant plant species and overall species richness of the site (Vasey et al 2012, Watson and Byrne 2009) and the gross primary production of those plants (Spalding and Hester 2007). Competition often interacts with salinity to drive the species composition at any given site along the estuarine salinity gradient (see Competition below). Salt water wetlands bordering the San Francisco Bay may have <20 species of vascular plants, brackish marshes of Suisun have 27-165 species, and the freshwater marshes of the Delta may have over 117 species (Tuxen et al. 2011).

Local changes to salinity within a site will drive plant zonation on a smaller scale. Culberson (2001) found local soil salinities could explain the regional emergent tidal marsh vegetation patterns in the Estuary, creating narrow zonation patterns in more saline marshes with wider bands along channels in more freshwater environments (see Figure 2, Species Composition tier 3).

In the submerged aquatic plant community, the dominant species will shift from *Zostera spp.* and *Stuckenia* in salt and brackish marshes to *Egeria, Potamogeton, Ceratophyllum*, and *Myriophyllum* in fresh water marshes (Boyer and Sutula 2015, Borgnis and Boyer 2015). The dominant floating plants in the Delta, *Ludwigia spp.*, *Hydrocotyle umbellata* and *Eichhornia*, occur only in fresh water (Wilson et al. 2005, Hestir et al. 2008, Khanna et al. 2012b).

At tidal restoration sites, local salinity may change once the site is restored, depending on previous land management, causing a shift in plant communities. Poor water circulation may result in pockets of high salinity dominated by halophytes (Culberson 2001). Sudden increases or decreases in salinity can change biogeochemical cycling of the wetland (Neubauer et al. 2013) and cause stress in plants acclimated to freshwater habitat (Sutter et al. 2014). Monitoring patterns of vegetation establishment in relationship to their salinity tolerance may be a better indicator of long-term salinity regime than discrete measurements of salinity.

Dissolved Gasses

All plants must have access to oxygen and some form of inorganic carbon to conduct photosynthesis and respiration. Emergent and floating plants generally receive most of the oxygen and carbon they need as gasses from the air through their leaves. Low oxygen availability in the sediment limits some emergent plants, but most submerged and emergent plants have specialized anatomy (aerenchyma) which enables diffusion of oxygen to the root zone (see Substrate above; Lenssen et al. 2003, Yarrow et al. 2009a).

Submerged plants may be limited by dissolved gasses in some situations. The concentration of dissolved CO_2 is a more frequent controlling factor over submerged aquatic plant growth than dissolved oxygen. Gasses dissolve 10^3-10^4x more slowly in water than air, resulting in rapid depletion of CO_2 in the vicinity of submerged leaves during photosynthesis (Lambers et al. 1998). CO_2 is the preferred form of inorganic carbon for photosynthesis, but CO_2 in the water column is in equilibrium with carbonic acid. Drawdown of CO_2 by photosynthesis will lead to an increase in pH and thus an increase in the availability of bicarbonate as an alternate carbon source (Boyer and Sutula 2015). Many submerged plant species are adapted to this and can facultatively switch from C3 to C4 metabolism and/or use bicarbonate as an alternate carbon source under conditions of low dissolved CO_2 (Yarrow et al. 2009a, Madsen 2013).

Nutrients in the water column

All plants require nitrogen and phosphorus, which they may acquire either through their roots or their leaves. Emergent vegetation acquires most of their nutrients from the sediment, so is less affected by water column nutrient concentrations than submerged or floating vegetation (see sediment nutrient section Lacoul and Freedman 2006).

Submerged vegetation may acquire nitrogen and phosphorus from the water column or the substrate, though water column absorption may be preferred by *Egeria densa* (Boyer and Sutula 2015). In the Delta, nitrogen and phosphorus loadings are relatively high and generally considered not limiting to primary production (Cloern 2001). Many aquatic macrophytes preferentially utilize ammonium; however growth may be inhibited at very high ammonium levels (Bornette and Puijalon 2011). Current issues in the Delta of increased availability of ammonium over nitrate may be important in determining SAV growth as well as phytoplankton species composition and growth (Glibert et al. 2014). However this is likely not the main driver of expansion in SAV in the last decade, as ammonium concentrations have been relatively constant as SAV has expanded (Boyer and Sutula 2015).

Floating vegetation must acquire nutrients from the water column; water column nutrient availability is one of the primary drivers of FAV growth, since light and substrate are not limiting. *Eichhornia* is most often limited by nitrogen, and increases in nitrogen will stimulate plant growth (Boyer and Sutula 2015 and references therein), though some cases of phosphorus limitation have also been observed (Wilson et al. 2005).

Monitoring nutrients present in the water column and sediment may elucidate relationships between nutrient concentrations and observed species composition at a restoration site.

pH

pH of the water chiefly affects SAV through its effect on the $C0_2$-bicarbonate balance (see dissolved gasses, above), with submerged plants growing in pH > 5.4 needing to make use of bicarbonate as well as CO2 (Lacoul and Freedman 2006). pH in the Delta is normally slightly basic (pH = 8 $\underline{+}$ 0.4, Monsen et al. 2007).

pH of the soil is generally more important than pH of the water column for emergent plants (Lacoul and Freedman 2006), since higher soil pH is associated with lower bioavailability of metals (Jackson et al. 1993).

PAR: Photosynthetically Active Radiation

All photosynthetic plants require light. The observed relationships between aquatic plant growth, zonation, and community composition with turbidity, depth, competition, and day length are all indirectly a result of those factors influencing light availability. In plant physiology, the total amount of light reaching the leaves of the plant is less important than the amount of light from the part of the electromagnetic spectrum (400-700 nm) which is available for photosynthesis, referred to as "photosynthetically active radiation" (PAR).

Light reaching the vegetation is driven by amount and timing of solar radiation (see Seasonality, tier 1), shading from competitors (see Competition, tier 3), growth of epiphytes (see food web, teir 4), water depth (Elevation, above), and turbidity (Water Quality, above). Which factor, if any, is the primary driver of observed plant zonation is often difficult to distinguish in the field and may change seasonally and spatially.

PAR is easily measured with PAR sensors, which are common components of weather stations and other field measurement equipment, though it may also be indirectly measured using data on drivers of PAR, such as shading, day length, and water depth.

Tier 3: Aquatic Vegetation Responses

Colonization

The colonization of vegetation in restored wetlands depends on initial site conditions, proximity to existing vegetation patches, availability of propagules, planting or seeding treatments, and responses of aquatic vegetation to site characteristics within the context of the landscape. Ideally, conditions at tidal restoration sites will promote the natural colonization of restored sediment surfaces in ways that mimic natural tidal marsh evolution. However, predicting colonization patterns is difficult because the Delta is highly degraded, there is no consistent monitoring record of aquatic vegetation, reference tidal marsh sites are rare, and our knowledge of restoration is still fairly nascent.

Means of reproduction may greatly influence colonization probability for a species. Many aquatic plant species reproduce asexually, which facilitates their dispersal and colonization throughout aquatic systems (Santamaria 2002). For many non-native species in the Delta, asexual reproduction is the sole or primary means of spreading.

Emergent marsh plant species typically reproduce via aboveground stolons or underground rhizomes, though species are also capable of reproducing sexually from seed. In freshwater marshes of the Delta, *Typha spp.* and *Schoenoplectus* spp. produce rhizomes, which enable rapid colonization and expansion. Similarly in Suisun Marsh, non-native *Phragmites australis* can rapidly spread within a disturbed site and produce thick stands of vegetation; rhizomes can be quite deep and difficult to remove, enabling rapid regeneration of stands after control efforts.

Most submerged and floating species in the Delta reproduce primarily by vegetative means, including fragmentation, production of turions, and/or rhizomatous growth. *Eichhornia*, while also capable of sexual reproduction, primarily spreads vegetatively through production of ramets along floating stolons (horizontal stems). *Eichhornia* can produce a large number of seeds that can remain dormant for extended periods of time, but germination appears to primarily occur following dry-down events (Gettys 2014). *Ludwigia* also reproduces primarily through vegetative growth, very rarely establishing from seeds in its invasive range (Thouvenot et al. 2013).

Of submerged species, *Egeria densa* is dioecious but not known to produce seeds in the Delta, as female plants have not been observed outside of South America (Pennington 2014). As a result, populations within the U.S. are believed to have low genetic diversity. *Egeria* does not produce tubers, turions, or rhizomes, but rather relies on stem and root crowns for colonization and survival. Other non-native submersed species are capable of sexual reproduction; *Myriophyllum spicatum* can produce a significant number of seeds, but successful colonization of new plants from seed in nature has not been documented and sexual reproduction is not thought to be significant for the species under natural conditions (Madsen 2014). *Potamogeton cripsus* also produces seed, but has an extremely low germination rate (0.5%) so that reproduction is mainly due to vegetative turions (Woolf 2014).

Colonization of marsh vegetation is highly dependent on proximity to existing sources of seeds or other propagules, the size of source populations, and dispersal dynamics (Smith and Warren 2012). Generally, planting or seeding is considered unnecessary for common plants with large seed or propagule sources (Williams and Faber 2001) but planting may increase overall species diversity and the necessity of planting differs greatly by species (Zedler 2000, Mitsch et al 2012). The large populations of several floating and submersed non-native aquatic species in the Delta mean that large sources of propagules exist in close proximity to restoration sites.

There is often considerable uncertainty in exactly how vegetative communities will develop in response to restoration actions due to variability in multiple site factors and their interactions. Flow and substrate stability are principal drivers affecting establishment of vegetation at a site (Boyer and Sutula 2015). Elevation influences tidal inundation depth and duration (See Wetland Evolution model and Topography, above)), which in turn affects species ability to colonize a site. Light availability at the sediment surface (PAR) is very important to establishment of submerged species (Boyer and Sutula 2015). Other key site characteristics, including salinity and hydrology, affect survival and colonization, often through interactive effects. Actions or designs which may promote colonization by native over non-native species at restoration sites have not been explicitly investigated in the Delta.

Previous restorations in the Delta indicate that, given appropriate elevations and tidal inundation periods, emergent vegetation will rapidly establish on marsh plains (Sloey et al. 2015). Surveys conducted for the BREACH I & II studies found that restoring marshes are typically dominated by bulrushes (*Schoenoplectus acutus* and *S. californicus*) intermixed with *Typha* sp. (cattails) and many understory forbs. Diversity appeared to correlate with topographic complexity, which was higher in ancient reference marshes. Thus, reference sites tended to be much more species rich; however, whether reference sites can be considered to be a reliable template for predicting marsh evolution and vegetative complexity is unknown as no long-term studies of vegetative community changes through time following restoration have been conducted in the Delta (Simenstad et al. 2006).

Growth

Tidal wetlands are some of the most productive ecosystems on earth, in part because aquatic plant species generally exhibit higher growth rates than their terrestrial counterparts (Mitsch and Gosselink 1993). Some invasives are particularly fast growing and can overtake newly restored areas in less than a season. For example, *Eichhornia* is one of the fastest growing plants in the world; a mat of 10 plants has been found to expand to 650,000 in a single growing season (Penfound and Earle 1948). Abiotic factors regulating this rapid growth are covered in Tier 2, above, while biotic factors regulating growth are covered in Competition, and Tier 4, below.

The basic form for most emergent plants in tidal wetlands, such as *Typha* and *Schoenoplectus* spp., is clonal with ramets comprising leaf systems and their associated rhizomes, roots, and inflorescences. The clonal structure allows for rapid growth once an individual is established and can enable survival during extended periods of stress (Santamaria 2002), including chemical control efforts (e.g., *Phragmites*, Ailstock et al 2001). The large root system of most emergent plants means that underground biomass is considerable and can contribute significantly to the accumulation of plant material over time. Well-established beds of emergent plants can raise marsh elevations by forming large root mats (Rooth et al. 2003, Drexler et al 2011, see Wetland Evolution model).

Floating species typically form dense mono-specific mats due to their ability to grow rapidly and spread vegetatively along floating stems (stolons). Rapid growth of species like *Eichhornia* means that floating mats can double in size every 7.4 days (Gutierrez et al. 2001), and *Ludwigia* can double in as little as 15 days in the field (Thouvenot et al 2013). Despite being more limited by light and CO_2 concentrations than emergent or floating species, submerged macrophytes are also capable of rapid growth rates [e.g., 0.5" per day in *Egeria* (Pennington 2014), 4" per day in *Potamogeton crispus* (Woolf 2014)]. Doubling time in *Egeria* can be as low as 12 days (Pistori et al. 2004).

The growth forms of submerged species can be quite variable, both within a species and between species. For example, *Egeria* produces a thick canopy with multiple orders of branching at the upper end. In contrast, the native *Stuckenia pectinata* exhibits relatively sparse leaf growth which minimizes self-shading (Boyer and Sutula 2015). Growth forms can also vary seasonally for some species. For example, *Egeria* exhibits slower growth in winter and plants lack canopy branches, have shorter internodes, and can even have a prostrate habit on the bottom (Haramoto & Ikusima 1988). One dominant native, *Ceratophyllum demersum*, lacks roots and is typically found tangled within the canopies of other submerged species, such as *Egeria* (Boyer and Sutula 2015). For a discussion on the effects of canopy structure on plant communities, see "Competition" below. Canopy form can also influence the value of the plants as habitat for invertebrates and fishes.

Decomposition

Decomposition of plant material may provide an important source of production for higher-order consumers. While few aquatic fish or invertebrates consume living plant material, decomposing plant material is more bioavailable due to colonization by fungi and bacteria (Boesch and Turner 1984). Aboveground dry biomass of *Schoenoplectus* species during the midpoint of the growing season recorded in areas of the Delta may be as high as 920-2,300 g/m^2, an order of magnitude higher than submerged or floating vegetation (Miller and Fujii 2010). Biomass turnover is approximately twice the standing biomass at the midpoint of the growing season (Miller and Fujii 2010), meaning that 2,000-4,000g/m^2 of senescent plant material may be available to detritivores every year. Decomposition of *Schoenoplectus* may range from 1-4% biomass per day, depending on temperature, meaning the carbon input rate for these species is on the order of 20-40 g/m2/day. Although this carbon is less bioavailable than phytoplankton carbon (Sobczak et al 2002, Muller-Solger et al 2002); ,these rates are an order of magnitude higher than the highest rates of phytoplankton primary productivity (0.1-1 g/m2/day) typically measured in the Delta (Jassby et al. 2002).

Decomposition rates which are too high may be detrimental to the ecosystem. The large biomass of aquatic species can have dramatic effects on water chemistry following high mortality events, such as control treatments or freezing temperatures. Control treatments that are not followed with biomass removal result in large amounts of biomass left to decompose in the water, causing nutrient release and anoxic conditions that may result in local fish-kills, particularly in areas with limited flow (Reddy and Sacco 1981,Greenfield et al. 2007, Wilson et al 2007).

Competition

Competition often interacts with salinity to drive species composition at any given site along the estuarine salinity gradient. Very few plants can tolerate high salinities. Plants that do survive in high salinities, such as the emergent species *Spartin*a, will often grow more vigorously at low salinities when isolated from competition, but will be out-competed when competitors (such as *Schoenoplectus spp*) are present (Crain et al. 2004). In the submerged community, *Stuckenia* will grow more vigorously at low salinities, but is outcompeted by *Egeria* in the Delta; in brackish reaches of Suisun marsh, *Stuckenia* is dominant (Borgnis and Boyer 2015). In freshwater and low-salinity regions of the Delta, growth peaks of *Egeria* in both the late spring and fall and its ability to overwinter give it a competitive edge over other species (Santos et al. 2012).

Dense canopy-forming vegetation favors mono-specific stands, leading to lower biodiversity through competition and exclusion (Roberts et al. 1999). As described earlier, the particular growth form or branching architecture of certain species results in intra- and inter-specific competition for light. As a result, controlling one species can lead to dramatic population increases of other species once the competitive pressures are removed (Khanna et al. 2012).

Ecosystem Engineering:

Aquatic plant species, either directly or indirectly, can affect the availability of resources to other species and in doing so modify, maintain, or even create novel habitats; such organisms are referred to as “ecosystem engineers” (Jones 1994). Ways in which vegetation can affect the abiotic environment include attenuation of water velocity, impacts on water quality and dissolved oxygen, enhancement of sediment deposition, and decreased light availability, among others.

Across the marsh plain, mean flow speed and turbulence intensity are inversely related to emergent vegetation stem density and to distance from the creek edge (Wood and Hine 2007). Flow energies decrease by about one order of magnitude when flows encounter the vegetated marsh surface and continue to decrease as vegetation density increases. Stands of submerged aquatic vegetation also significantly reduce water velocity, leading to sedimentation of suspended particles, increased water clarity, and increased thermal stratification (Kosten et al. 2009, Hestir 2010, Hestir et al. 2010; Hestir et al. 2015).

Reductions in flow speed coupled with energy decay provide a hydrologic mechanism for sediment deposition patterns commonly observed in marsh systems. In the Delta, decreases in turbidity have been correlated with increases in SAV cover, potentially caused by the expansion of the invasive *Egeria densa* (Hestir et al. 2015). This can facilitate subsequent colonization of vegetation in a positive feedback loop (Yarrow et al. 2009). For a full discussion of the role of vegetation's effects on hydrodynamics, see the Wetland Evolution Model.

Emergent vegetation canopy cover may reduce water temperature, as one study found surface temperatures in emergent vegetation to be 1.4 C lower than in submerged vegetation (Crepeau and Miller 2014). Dense canopies of floating or submerged vegetation can also shade out and affect growth and survival of other species in the community (see Competition, above). However, reduction in flow can also increase local surface water temperatures through reduction in mixing (Boyer and Sutula 2015).

Dissolved oxygen may be high within a canopy of floating or submerged vegetation near the water's surface, but will decline or become anoxic below the canopy. This is due to reduced mixing within the plant canopy, decreases in light and associated increases in the respiration: photosynthesis ratio of organic matter below the canopy (Frodge et al. 1990, Caraco and Cole 2002). Control methods of nuisance aquatic weeds may also cause low dissolved oxygen levels due to decomposition of shredded or poisoned plant material (Greenfield et al. 2007). In a restoration site, monitoring areas with extremely dense aquatic vegetation for low dissolved oxygen may help identify areas that limit fish occupancy.

AV Type and Species Composition

The abiotic and biotic factors that influence vegetation colonization and growth lead to complex patterns of zonation that are typical of tidal marshes (Figures 2 and 3). Spatial distribution of plants across estuarine salinity gradients is driven primarily by the degree to which species can tolerate salinity and, as noted in Suisun Marsh, the transition to a halophytic plant community can be pronounced. Much of the Delta is maintained as fresh water with little seasonal variation through water management practices; however, Suisun Marsh has a north-south salinity gradient resulting from fresh inflows from local runoff and creeks and an east-west salinity gradient as a result of Delta freshwater inflows and operation of the salinity control gates. As noted in Salinity, above, freshwater marshes tend to have higher plant diversity than brackish or saline marshes.

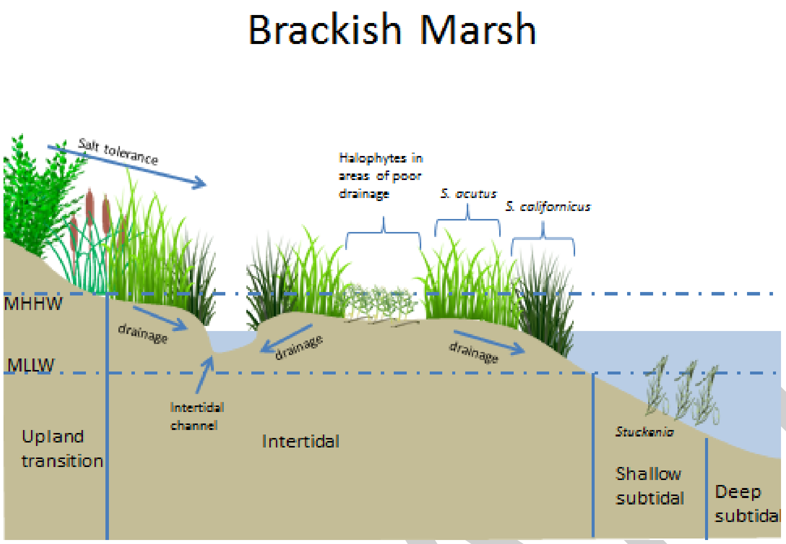

Figure 2: General factors controlling vegetation zonation in brackish marshes

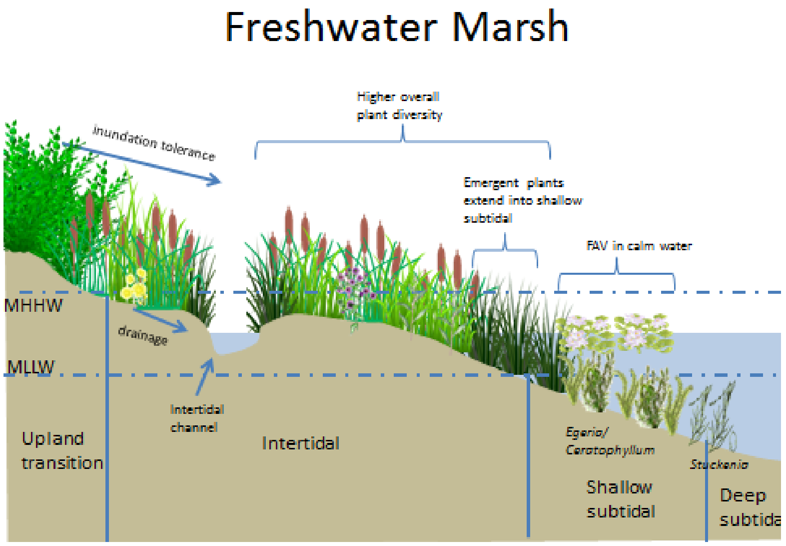

Figure 3: General factors controlling vegetation zonation in freshwater marshes

Comment [CL1]: This figure seems to suggest that Stuckenia will likely be deeper than Egeria. Do we know that to be true? I think Egeria might easily be in deeper areas, even "deep subtidal" as long as there is enough light.

Comment [SS2]: Per Anitra: add myriophyllum, Stuckenia not prevalent in freshwater marsh (but see Kathy Boyer's recent work!)

Comment [RKH3]: Stukenia tended to be deeper than Egeria in Kathy Boyers work, I think, but I'll have to look it up again.

Within marshes, vegetation zonation also develops with respect to drainage and elevation patterns (See Bathymetry and Substrate above). Lower intertidal zones are dominated by species which tolerate frequent inundation, such as tules (*Schoenoplectus californicus* and *Schoenoplectus acutus* var. *occidentalis*) and cattails (*Typha* spp.). Areas along sloughs and at higher intertidal zones on fringing marshes or marsh plains support vegetation which can only tolerate less frequent flooding by spring tides.

As sea level rises, marshes that cannot accrete sufficient sediment to maintain elevations will drown to become mudflats and higher elevation regions will become inundated, a process known as marsh migration; brackish areas will become more saline and freshwater marshes in more seaward transition zones will become brackish (Schile et al. 2014). This successional progression will result in changes to the vegetative communities of much of the San Francisco Estuary.

A long history of agriculture and other anthropogenic activities has resulted in many introductions of non-native species. Pristine brackish to freshwater tidal marsh reference prototypes are rare in the Delta, but certain remnant wetlands (Rush Ranch) and restored sites (Ryer Island, Liberty Island) are less impacted by invasive vegetation and provide examples of what might be expected from restoration actions. Though reduced in size, these areas are characterized by mature marsh communities that have persisted in the face of the Delta's large-scale changes (Simenstad et al. 2006).

Delta tidal marshes are composed of a mix of emergent, submerged, and floating native and non-native species. Dominant species comprising these vegetation types are described briefly below. A more complete list of plant species found in the Delta and Suisun Marsh is listed in Appendix A.

Emergent plants:

Within Suisun's brackish marshes, lower intertidal zones are dominated by emergent species including tules (*Schoenoplectus californicus* and *Schoenoplectus acutus* var. *occidentalis*), cattails (*Typha* spp.) and common reed (*Phragmites australis*). Higher intertidal zones of Suisun's brackish marshes commonly include saltgrass (*Distichlis spicata*), pickleweed (*Sarcocornia pacifica*), Baltic rush (*Juncus arcticus* subsp. *balticus*), three-square bulrush (*Schoenoplectus americanus*), marsh gumplant (*Grindelia stricta* var. *angustifolia*), sea arrow-grass (*Triglochin maritima*), slender arrow-grass (*T. consinna*), fleshy jaumea (*Jaumea carnosa*), and perennial pepperweed (*Lepidium latifolium*).

In freshwater marshes emergent vegetation establishes in the intertidal zone and slightly below, up to 0.2 m below mean lower low water (Simenstad et al. 2000). Tules and cattails dominate the freshwater marshes of the interior Delta, with *S. californicus* typically occupying lower elevation and more exposed sites than *S. acutus* (Hickman 2007).

In the high marsh, several invasive species may be of management concern for restoration. Perennial pepperweed (*Lepidium latifolium*) is a highly invasive herbaceous perennial that is found to adapt readily to natural and disturbed wetlands. As it establishes and expands, the plants create large monospecific stands that displace native plants and animals (for example, in Suisun Marsh; Howald 2000). *Phragmites* and *Arundo* have also demonstrated negative effects for fish and wildlife on the East Coast of the US (as reviewed in Lambert et al. 2010). They are major invasive species in the Delta, but their effects on wetland habitat on the West Coast are less known.

Submerged plants:

The native *Stuckenia pectinata* is typically the only submerged plant species found within the open Suisun Bay (Boyer and Sutula 2015), though the non-native *Potamogeton crispus* and native *Ceratophyllum demersum* have been noted to have recently spread in the artificially freshened reaches of Montezuma Slough (Grewell et al. 2012). Within the freshwater areas of the Delta, *Stuckenia pectinata* is thought to be outcompeted by other freshwater species, in particular the non-native *Egeria densa* (Borgnis and Boyer 2015).

Freshwater areas of the Delta are now dominated by non-native submerged vegetation species, including *Egeria densa* and to a lesser extent *Potamogeton crispus* and *Myriophyllum spicatum* (Hestir et al 2015); restoration sites will likely be very near to sources of propagules for colonization. *Ceratophyllum demersum* was the most frequently observed native submerged plant species in recent studies within the central and eastern Delta (Santos et al. 2011, Boyer et al. 2013).

Floating plants, and rooted plants with floating leaves:

Two non-native species, *Ludwigia* and *Eichhornia,* both dominate the FAV assemblage in the Delta (Khanna et al. 2012). Water primroses (Ludwigia spp) and the native *Hydrocotyle* root in the sediment in areas of calm, shallow water and spread laterally via creeping stems and floating leaves. The spatial coverage of non-native water primroses (*Ludwigia* spp.) has dramatically expanded within the last five years, particularly throughout the northern Delta (Boyer and Sutula 2015). Unlike *Hydrocotyle, Ludwigia* has been observed to climb over other emergent plants (Boyer and Sutula 2015). *Hydrocotyle umbellata* has recently declined considerably in its coverage (Boyer and Sutula 2015).

Eichhornia is a true floating species, with roots hanging into the water, not anchored in soil. It is now the most dominant floating species in the Delta (Khanna et al. 2012). Although it is commonly observed in areas of calm, shallow water, it can become easily dislodged and is frequently found floating in deep, open waters; this fragmentation mechanism facilitates its dispersal to new areas. Other true floating species include the native water fern*, Azolla filiculoides,* and duckweed (*Lemna spp*) (Hickson et al 2007).

Tier 4: Tidal Wetland Responses

Wetland Evolution

Landscape and site conditions influence the evolution of wetlands (see Wetland Evolution Conceptual Model), but there are two-way interactions between surface elevation and plant establishment, growth, and decomposition rates (Reed 2002; Williams and Orr 2002, Culberson 2001). The presence of vegetation is likely to modify elevation and soil conditions over time through trapping of sediment and accumulation of organic matter eventually increasing soil surface elevations. Increased elevations and decreased soil density caused by establishment of plants will make the substrate easier for future plants to colonize and reduce future change in elevation (Simonsted et al. 2006; Sloey et al. 2015, Marriotti and Fagherazzi, 2010, Mudd et al. 2010; also see Wetland Evolution model for more information).

Habitat Structure

Aquatic macrophytes are ecosystem engineers (see above); in addition to altering elevations, their structures can transform shallow water zones into complex habitats for invertebrates and the fishes that feed upon them. As it provides substrate for epiphytic algae and invertebrates, refuge from predation, and high food supply, vegetated areas often have an order of magnitude higher animal biomass than open-water (Heck and Crowder 1991). However, emergent, submerged and floating vegetation each provide very different habitat structure. Species within each vegetation class exhibit further differences. For example, Howe et al. (2014) found variation in neustonic macroinvertebrate community composition was explained partially by species composition of emergent vegetation adjacent to channels. The mixture of plant communities within a restoring wetland is a prominent component of the value of the wetland to listed fish species and will change seasonally and annually as the wetland develops.

The structure of emergent vegetation provides habitat for epiphytic algae and invertebrates. Multiple studies across all types of emergent marshes have found positive correlations between invertebrate biomass and vegetative cover (De Szalay and Resh 2000, Rozas and Odum 1998, Heck and Crowder 1991). Recent studies in Suisun Marsh point to the positive influence of emergent marsh vegetation in providing enriched food resources for Delta fish species, with high abundances of phytoplankton, mysid shrimp, and native fishes in relatively undisturbed marsh sites (O'Rear and Moyle 2013, Schroeter et al. 2015).

However, these relationships are not true in all situations. Van der Wal and Herman 2012 did not find significant effects of marsh plant cover and vegetation type on macrobenthic biomass, taxon diversity, and community structure. Organic enrichment of the sediment and mechanical hindering of macrofaunal activity by plant roots are possible mechanisms for the variability seen in macroinvertebrate responses in salt marshes. On the east coast, *Phragmites australis* has invaded *Spartina* marshes causing hydrological and physical changes (increases in elevation and reductions in edge) on the marsh surface. Resident fish are negatively affected (Able and Hagan 2000, 2003; Hunter et al. 2006), possibly due to lower carbon exports and by impeding fish movement (Weinstein and Balleto 1999).

Submerged aquatic vegetation provides food and substrate for a large biomass of aquatic invertebrates (Heck and Crowder 1991, Manatunge et al 2000). Some of these invertebrates consume the macrophytes themselves, while others graze on epiphytic algae or prey on smaller invertebrates living in the vegetation (Voigts 1976, De Szalay and Resh 2000, Pickard and Benke 1996). These vegetation-associated invertebrates provide an important source of food for fish; however, vegetation may also decrease predation efficiency (Manatunge et al 2000). Species composition and biomass of invertebrates may vary between macrophyte species in the Delta (Boyer, unpublished data).

Submerged aquatic vegetation also provides egg laying substrate for many fishes and habitat for many larval fishes (Grimaldo et al. 2004). SAV also mediates predation on fish species by providing cover (Heck and Thoman 1981, Rozas and Odum 1988). Vegetated littoral habitats in the Delta tend to favor higher biomass of fishes; however, they are dominated by introduced species such as largemouth bass and inland silversides (Brown and Michniuk 2007, Nobriga et al. 2005). Grimaldo et al. (2012) noted consistent patterns of higher abundances of fishes in edge habitats in interior Delta marshes over several years, suggesting that they provided favorable rearing conditions: however, most of these fishes were alien species. Conrad et al. (2016) observed the highest densities of juvenile Largemouth Bass at intermediate SAV densities, suggesting that at these densities juveniles find a balance of increased refuge from predators while maintaining an ability to locate and forage for invertebrate prey. Nobriga et al. (2005) found fish biomass was generally higher in SAV-dominated habitats than the turbid, open habitats; however, unvegetated areas were favored by native fish species such as Delta Smelt, Chinook Salmon, and Splittail (*Pogonichthys macrolepidotus*).

Floating vegetation also provides structural habitat, but its effect on invertebrate and fish communities may take many forms. *Eichhornia* can support high densities of invertebrates, particularly amphipods, but community structure of invertebrates in the roots of this invader is significantly different from that of native floating vegetation (Toft et al 2003). In a review on the effects of *Eichhornia* on fish in other areas, Villamagna and Murphy (2010) found that the effects depended on whether the fish community was based on a littoral and epiphytic or pelagic and planktivorous feeding strategy. Fish response to *Eichhornia* was non-linear, with intermediate levels of *Eichhornia* favoring the highest abundance of fish species (Villamagna and Murphy 2010). Edges of *Eichhornia* mats likely provide fish food and habitat; however, large dense mats without edges would likely reduce fish habitat due to decreases in dissolved oxygen. In the Delta, Toft et. al (2003), also supported this concept by showing native fish species tended to favor native invertebrates found in native *Hydrocotyle* over invasive *Eichhornia*, which may be related to *Hydrocotyle'*s sparser growth pattern.

Food Web Linkages

Plants as carbon sources:

Besides creating habitat structure for invertebrates that are an important source of fish food, aquatic plants also contribute to the food web directly through the detrital loop (Howe and Simenstad 2014, Bergamino and Richoux 2015, Howe et al. 2014). While detritus from tidal marsh vegetation is much less bioavailable than phytoplankton or algal carbon (Muller-Solger et al. 2002, Sobczak et al. 2002), production from emergent vegetation may be more than an order of magnitude greater than phytoplankton (Cohen et al. 2014). For a full discussion of the role of detrital carbon in the food web, see the Food Web Model.

Gut content and isotope analysis of marsh fish indicate marsh-derived detrital carbon transfer through macroinvertebrates to higher trophic levels (Levings et al 1991, Toft et al. 2003; Visintainer et al. 2006; Cohen and Bollens 2008; Whitley and Bollens 2014). In lower Columbia River tidal marshes, all size classes of juvenile salmon were linked to wetland-derived food webs, and they fed directly on insect and amphipods associated with vegetation (Bottom et al. 2011). Stable isotope analysis supports this result, finding the majority of carbon being derived from vascular plant detritus (Maier and Simenstad 2009). Stable isotope data from a variety of consumers shows that the percentage of marsh-derived carbon in higher trophic levels is proportional to the relative area of marsh on the landscape (Grimaldo et al 2009, Howe and Simenstad 2011, E. Howe, unpublished data)

Aquatic Vegetation Consumers

The food web exerts control from above on vegetation directly through grazing, and indirectly through alteration of the substrate. Invertebrate grazers include sap feeders and leaf miners such as Delphacidae, Cicadellidae, Aphididae, Chironomidae, and Ephydridae (Howe et al 2014). Larger herbivores include crayfish, which are abundant in the Delta, but poorly studied (Herbold and Moyle 1989). Studies in other systems have found crayfish can greatly reduce biomass of submerged macrophytes or inhibit their establishment through herbivory (NystrÖM and Strand 1996, Rodríguez et al. 2005, Matsuzaki et al. 2009).

Vertebrate herbivores, particularly fish and birds, may also affect the vegetation community. Asian carp (*Cyprinus carpio*) are abundant in the Delta and have been implicated in reductions of submerged vegetation in other areas (Crivelli 1983, Matsuzaki et al. 2009). Geese and ducks may inhibit the establishment of both submerged and emergent vegetation (Baldwin 2004). If vegetation fails to establish as expected, monitoring these and other aquatic herbivores may provide clues as to the cause.

If a restoration site is combating invasive weeds, particularly voracious herbivores may be used as biocontrol methods. Carp have been used as biocontrol agents for invasive aquatic weeds (Chilton and Muoneke 1992), though their effect on Delta vegetation is not known. In the Delta, two weevils have been introduced for biocontrol of *Eichhornia* (*Neochetina bruchi* and *N. eichhorniae*), but so far have not had wide-spread success (Spencer and Ksander 2004). Research into biocontrol methods is ongoing.

Fish Linkages

Worldwide, many species of fish use the increased food supply and protection from predators provided by aquatic vegetation as an important nursery and foraging area (Heck and Thoman 1981, Rozas and Odum 1998, Castellanos and Rojas 2001). However, the effect of aquatic vegetation on native fish species targeted by restoration is highly variable and under considerable debate. Several studies in interior Delta sites demonstrate that marsh ecosystems may have more invasive fish than native fish and are not primary habitats for salmon or smelt (Nobriga et al. 2005; Brown and Michnuik 2007; Grimaldo et al. 2004, 2012, Conrad et al, 2016). However, some recent studies on naturally restoring marshes and restored marshes in the western Delta are suggesting marshes are important for native fish as well (Grenier 2004; Grimaldo et al. 2009; Howe and Simenstad 2011). In the Delta, we hypothesize that an increase in vegetation structural complexity favoring epiphytic algae and invertebrate communities may benefit native fish such as juvenile Salmon and Splittail that feed in littoral tidal and floodplain areas. It may ultimately benefit Delta Smelt and Longfin Smelt through exported food resources to subtidal zones (See Salmon and Delta Smelt models).

The benefit of emergent vegetation to salmonids in tidal wetlands has been extensively studied, particularly in the Pacific Northwest (Levings et al. 1991, Bottom et al. 2011, David et al. 2014, Goertler et al. 2015, see Salmon Model). These papers and many others have found greater food resources, higher rates of feeding, faster growth rates, and larger size of outmigrants in areas with restored tidal wetlands. There have been far fewer studies on the relationship between salmonids and submerged or floating vegetation, but one experiment in the Delta found lower rates of predation for juvenile Chinook salmon in SAV beds (Grimaldo et al. 2000), and salmonids often prefer habitats with SAV for cover in other systems (Halvorsen et al. 1997). Other studies found salmonids less likely to occupy SAV in the Delta (McLain and Castillo, 2009), so this is an area for further research.

The interactions between smelt and wetland vegetation are poorly understood. Delta smelt generally prefer to inhabit open water areas (Baxter et al 2015), and in captive environments have been observed to actively avoid vegetation (Ferrari et al. 2014). However, Delta Smelt are found to consume high-energy larval insects and amphipods, which are likely to be sourced from aquatic vegetation, when occupying open water adjacent to emergent vegetation (Whitley and Bollens 2014). Recent data has also demonstrated Longfin Smelt spawning in marsh channels (Grimaldo pers. comm). Despite the food-web subsidy derived from wetlands, smelt may be negatively affected by submerged vegetation due to increases in water clarity and increased predator abundance caused by expansion of SAV (Hestir 2015, Ferrari et al. 2014).

Conclusion:

Ecological restoration often focuses on establishment of desired vegetation and control of undesirable vegetation, assuming the remainder of the ecosystem will follow. Therefore, monitoring success of restoration sites is often synonymous with monitoring vegetation. Current restoration projects in the upper San Francisco Estuary which aim to benefit salmon and smelt must monitor vegetation as the foundation of the community, but then go on to better understand the hypothesized linkages between vegetation and the rest of the food web. Early research suggests tidal wetlands with emergent vegetation may be an under-recognized part of life history diversity for migratory fish species. However, invasive floating and submerged vegetation is also likely to grow in wetland habitats, and the impact of these forms of vegetation on the native fish of the Delta is still understudied. Proper monitoring of fish, invertebrates, and vegetation together may help restoration practitioners design and manage sites that best support native fish of California.

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Interagency Ecological Program

Tidal Wetlands Restoration Monitoring Project Work Team

Ecosystem Element Conceptual Model

Invasive bivalve control on wetland productivity

Prepared by: Rosemary Hartman, Larry Brown, Jan Thompson and Francis Parchaso

Based on the DRERIP *Corbicula* and *Potamocorbula* conceptual models: https://www.dfg.ca.gov/erp/current_models.asp

PREFACE

This Conceptual Model is part of a suite of conceptual models which is designed to guide monitoring of restoration sites throughout the San Francisco Estuary (SFE), but particularly within the Sacramento-San Joaquin Delta and Suisun Marsh. The conceptual models are designed to aid in the identification and evaluation of monitoring metrics for tidal wetlands restoration projects. These models are designed to structure scientific information such that it can be used to inform sound resource management and public policy.

The text for the Invasive Bivalve Model is largely adapted from the DRERIP *Corbicula* and *Potamocorbula* models. The introductory material from these models has been abbreviated and changes have been made to fit our new model structure and describe how the original models may be applied to monitoring of restored tidal marshes. For a more thorough description of the organisms' biology, see the original models.

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Overview

The San-Francisco Bay-Delta Estuary (hereafter SFE) is considered the “most invaded estuary in the world”(Cohen and Carlton 1998). It is also an estuary with unusually low primary productivity when compared to other temperate estuaries, even though relatively high nutrient levels do not generally limit phytoplankton production in the estuary (Cloern 2001, Jassby et al. 2002). The limited phytoplankton production has generally been attributed to the shallow depth of the photic zone caused by high turbidity (i.e., light limitation).Tidal restoration in the SFE is often conducted with the goal of increasing primary and secondary production by creating shallow, high residence time tidal wetlands, which are predicted to export productivity to at-risk fish species in the pelagic environment. However, the introduction and subsequent expansion of invasive benthic filter-feeders, most notably the bivalves *Potamocorbula amurensis* (overbite clam, hereafter *Potamocorbula*) and *Corbicula fluminea* (Asian clam, hereafter *Corbicula*), may reduce any such benefit by removing any increased phytoplankton production before it can be exported or used by pelagic consumers (Lewis et al. 1970, Lucas et al. 2002b, Lopez et al. 2006, Lucas and Thompson 2012). The effect of invasive clams is not limited to reducing in plankton resources available to at-risk fish species; clams also bioaccumulate contaminants that may cause serious toxic effects in their predators (Stewart et al. 2004). Therefore, restoration actions often aim to minimize establishment of invasive clams.

Potamocorbula and *Corbicula* are spatially separated with a small region of overlap in the SFE. *Potamocorbula* dominates in the saline environments of Suisun Marsh and the San Francisco Bay and *Corbicula* dominates in the fresh waters of the Sacramento-San Joaquin Delta (Delta). The two species have many similarities in how they affect the wetland food web, however there are some key differences in their biology and ecology that impact their distribution and effects. This model describes the factors that lead to colonization and growth of the clams, but focuses on describing the effects clam grazing has on the wetland food web rather than on the clams themselves. For a more thorough description of clam biology, see the DRERIP *Corbicula* and *Potamocorbula* conceptual models (Thompson and Parchaso 2012, 2013).

Natural History

Corbicula fluminea

Corbicula has been present in the estuary since at least 1945 (Hanna 1966). However, the species did not spread rapidly, and it was only when it began encrusting water supply systems in some locations did it receive attention (Eng 1979). It appears to be food limited in the system (Prokopovich 1969, Foe and Knight 1985, Thompson and Parchaso 2013). Its population density is patchy throughout the Delta, with food availability, physiological tolerances, dispersal capabilities, and predator densities all hypothesized to influence distribution (Lopez et al. 2006). The following is a brief description of *Corbicula*’s biology and life history as relevant to management of tidal wetlands.

Taxonomy: Determination of species within the *Corbicula* genus is not straightforward, with some species showing morphologic but not genetic differences and *vice versa* (Thompson and

Parchaso 2013). Confusion over species identification has led to conflicting reports of physiological tolerances from *Corbicula* species in different regions in North America, so any studies of *Corbicula* not from the SFE may not apply to the current population in SFE. Based on previous work on *Corbicula* in the SFE (Siripattrawan et al. 2000, Thompson and Parchaso 2013), we will assume that *Corbicula* found in the SFE is Lee et al's (2005) *Corbicula fluminea* "morph A".

General Life History: *Corbicula* is a simultaneous hermaphrodite that may cross-fertilize or self-fertilize their eggs (Kraemer and Galloway 1986). The ability to self-fertilize makes it a particularly successful invader because a single colonist can establish a population (Sakai et al. 2001). *Corbicula* generally reproduces when water temperatures exceed 10°C, between April and November in the SFE (Eng 1979, Foe and Knight 1986). The trochophore, veliger, and pediveliger stages are brooded in the gills of the adult and the pediveligers are released through the siphon and settle out of the water column within 48 hours (Thompson and Parchaso 2013). Adults have a relatively short brooding period, so may have several broods throughout the reproductive season (Heinsohn 1958, Foe and Knight 1986). The ability of *Corbicula* to produce multiple broods per season means recruits may be available to establish new populations as soon as water conditions are favorable, facilitating its invasion of new areas (McMahon 1983, 2002).

Corbicula most likely reaches reproductive maturity at 6-10mm, which it can achieve at 3-12 months of age (Thompson and Parchaso 2013). Adults in North American populations live 3-4 (Eng 1979, Brown et al. 2007). *Corbicula* lives relatively close to the surface of the sediment or on top of harder sediments. It feeds both by filtering plankton out of the water column and by pedal-feeding in the sediment (Thompson and Parchaso 2013).

Potamocorbula amurensis

Potamocorbula was discovered in 1986 and hypothesized to be a recent invader who came via ballast water (Carlton et al. 1990). After its discovery, *Potamocorbula* invaded very rapidly, becoming the most common bivalve in the Estuary within two years (Nichols et al. 1990, Thompson et al. 2008). The following is a brief description of their biology and life history as relevant to management of tidal wetlands.

Note on taxonomy: *Potamocorbula amurensis* (Schrenck, 1861) was first identified in the SFE with the genus *Potamocorbula* (1990), but revisions of the genus during the 1990s reclassified the species as *Corbula amurensis* (Coan 2002). Throughout this document we will use Huber's (2010) reclassification as *Potamocorbula*. This is the genus currently recognized by the World Register of Marine Species (http://www.marinespecies.org/aphia.php?p=taxdetails&id=397175&allchildren=1).

General Life History: *Potamocorbula* has several important differences in life history from *Corbicula*. *Potamocorbula amurensis* is dioecious (separate male and female individuals), with broadcast spawning and external fertilization instead of being hermaphroditic and self-fertilizing like *Corbicula*. It generally spawns for 1-2 months in summer and 1-2 months in fall (Thompson and Parchaso 2012), though spawning may occur throughout the year in certain circumstances, particularly during droughts (Parchaso 1993).

Non-feeding trochophore larvae formed after spawning are planktonic and do not swim. They develop into suspension-feeding, mobile veliger larvae within 48 hours of fertilization. They swim actively at first, then intermittently and settle at day 17-19 (Nicolini and Penry 2000). The planktonic larval stage allows *Potamocorbula* to spread very rapidly throughout the environment. Juvenile clams begin reproducing after two months, or once they are about 5mm in length (Parchaso and Thompson 2002), after which they can reproduce twice a year (or continuously in certain conditions) for 2-2.5 years (Thompson and Parchaso 2012). Their short life span, high reproductive output, and pelagic larval stage allow them to colonize rapidly and dominate new areas quickly. Adults live near the surface of the sediment, often with part of the shell exposed, where they filter-feed phytoplankton and microzooplankton out of the water column (Carlton et al. 1990).

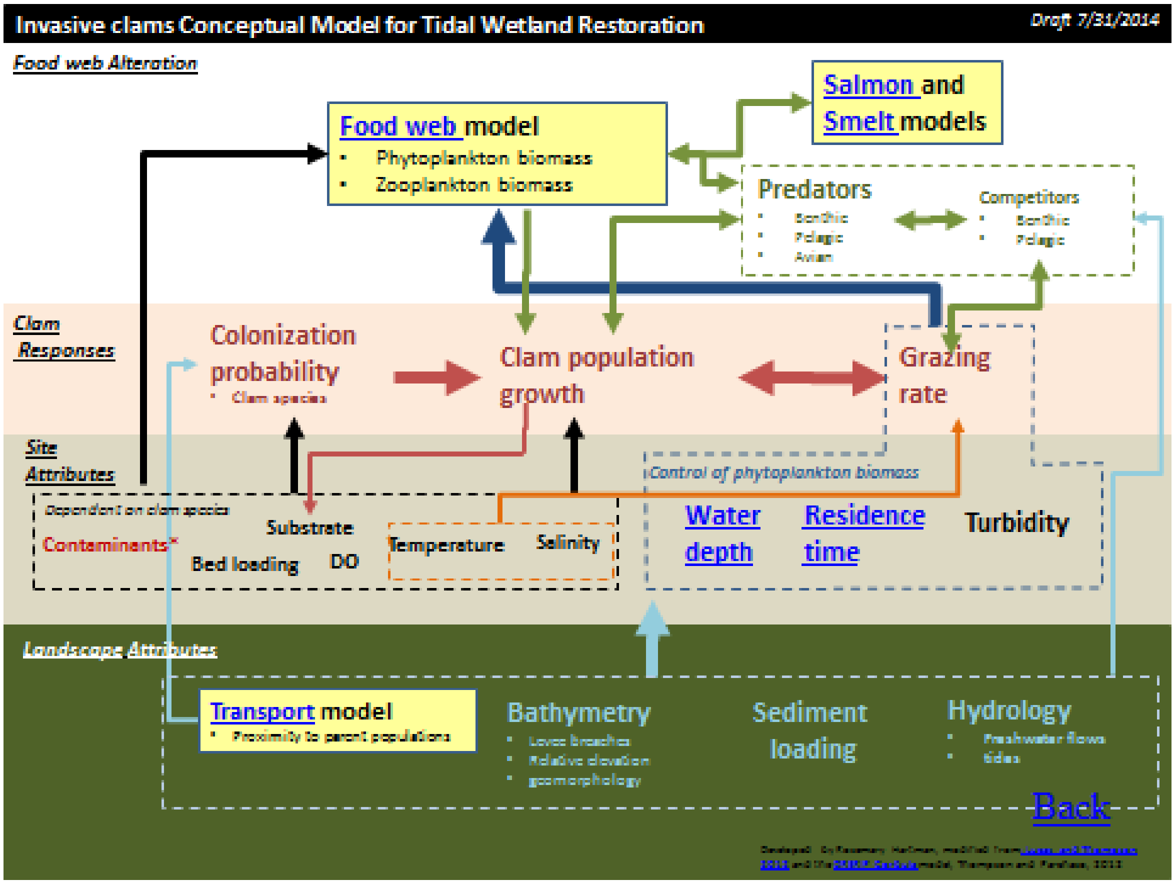

Figure 1. Conceptual model diagram of how invasive clams affect tidal marsh food web production.

Model structure

The factors that determine the effect of invasive clams on the food web are a combination of attributes acting at the landscape and site-specific scales (Figure 1). These habitat attributes may act on the food web either by affecting clam colonization probability, population growth rate, or grazing rate. The tiers move from landscape factors, to local factors, to clam responses and finally food web responses. Water depth, residence time, and turbidity interact to control clam grazing rate. This interaction is particularly important in driving phytoplankton biomass, so these factors have been surrounded by a box that crosses between tiers.

As in the General Wetland model, attributes surrounded with dotted lines interact strongly. Arrows represent specific, describable linkages or relationships between attributes. Thickness of arrows generally describes level of hypothesized importance; however this has not been quantitatively described. Thick arrows moving between tiers that do not connect to specific factors represent relationships with all attributes from the preceding tier to the next tier. Arrow and text color is used to aid in following arrows and does not convey meaning. Yellow boxes are links to companion models which provide greater elaboration on the specifics of other important processes, structures, or species.

For the Invasive Clams Model, there are four tiers (Figure 1): 1) Landscape Attributes, which are factors acting on the scale of the entire San Francisco Bay and Central Valley ecoregions, 2) Site Attributes, which are physical and hydrologic properties and processes within the focal wetland, 3) Clam Responses, which are the establishment, population growth, and ecosystem function (grazing) of the invasive clams, and 4) Food Web Alteration, the "outcome" tier which describes the effect of clam grazing on the productivity and food web responses of the wetland ecosystem.

Link to the Transport Model

Tier 1: Landscape attributes

Transport

While connectivity and transport from source populations is important to the spread and establishment of both *Potamocorbula* and *Corbicula*, there are important differences in environmental tolerances and dispersal mechanisms of larval stages between the species, generating differences in their distributions.

For *Potamocorbula*, freshwater flows control transport of larvae as well as salinity distributions. The strong net flow of water down-estuary during the spring, and the resulting positions of the salinity gradients associated with tributary streams and rivers allow recruits to move throughout the bay (Thompson and Parchaso 2012). In spring larvae may be carried over 40 km into the South San Francisco Bay (Thompson et al 2008). Potamocorbula larvae are most likely to establish in the upstream areas of the estuary during the fall, when flows are lower and salinity is higher. However, they are capable of recruiting upstream in the spring of dry years. The ability of larvae to disperse over large distances and re-establish populations that may have been extirpated (due to hydrologic variability, predation, etc.) contributes to the success of *Potamocorbula*'s invasion in the estuary. Dispersal ability of larvae makes planning of restoration sites more difficult because any active control measures for *Potamocorbula* may have to be ongoing, instead of occurring only before the project begins.

In contrast to *Potamocorbula*, *Corbicula* pediveligers are only found in the water column for the first 48 hours after release (Thompson and Parchaso 2013). After 48 hours, larvae settle on the surface of the sediment and disperse by either being 'lifted' into the water column by a mucus thread or by bouncing along the bottom as bed load. These dispersal mechanisms are used until the young clams are 10-14 mm in length, with larger adults also dispersing as bed load if there is sufficient flow (Prezant and Chalermwat 1984, Kraemer and Galloway 1986, Mouthon 2009).

Though *Corbicula* larvae are not as mobile as *Potamocorbula* larvae, the mucus threads on the juveniles allow them to move in response to environmental stresses (Williams and McMahon 1986). Adults tend to be less susceptible to environmental stress than juveniles due to their larger size, among other factors, so the ability to move with water quality is most important in younger life stages (Mouthon 2009). Regular migration from nearby populations may be necessary to recolonize areas with temporary changes to water quality, for example die-offs due to low dissolved oxygen. In some environmental conditions, dispersal by juveniles and adults from source populations can maintain populations that cannot maintain themselves (Thompson and Parchaso 2013).

Bathymetry

Link to the General Tidal wetland model

Overall bathymetry of the restoration site affects water depth and residence time, which interact with grazing rate to control phytoplankton productivity (see Control of Phytoplankton Biomass). Due to interactions between water depth, residence time, and grazing rate, it is difficult to predict exact outcomes (Lucas and Thompson 2012). Restoration managers may want to emphasize restoration sites with deeper initial conditions or rapid transitions between shallow tidal wetlands and open-water habitats; however, clam density, hydrologic residence time, and mixing of the water column will all influence the overall grazing rate. (Also see Water Depth and Grazing Rate sections under the Site Attributes and Clam Response tiers.)

Sediment Loading

Link to the Wetland Evolution model

Sediment loading is the major factor controlling turbidity and suspended sediments in the SFE. Turbidity interacts with water depth, residence time, and grazing rate to control phytoplankton productivity (See Turbidity under Site Attributes and Grazing Rate under Clam Responses). Sediment loading also determines what substrate is available for clam colonization, and affects bathymetry and hydrology through the process of wetland evolution (See Wetland Evolution Model). Sediment loads are chiefly controlled by freshwater flow in upstream rivers and sediment available for transport. Sediment loading is usually greatest during storm events (See General Tidal Wetland Model)

Hydrology

Landscape-scale hydrologic attributes determine the broad-scale distribution of the two species of clams throughout the SFE; primarily through the effect of freshwater flow on salinity (see Salinity under Site Attributes).

Potamocorbula is generally restricted to brackish and saline habitats, so its incursion upriver varies with season and hydrology. Spatial studies, done during several years by DWR and the USGS with varying sample designs, have shown differences in recruitment in wet and dry years based on the position of X2 (the distance from the Golden Gate bridge to the point in the estuary where salinity is equal to 2psu, an indicator of the position of the salinity gradient) *Potamocorbula* populations move up the estuary as salinity and X2 increases (Thompson and Parchaso 2012). Older *Potamocorbula* that became established in brackish habitats in dry years may persist in freshwater in wet years for months until brackish conditions are re-established. Because of the high reproductive output of adults and long dispersal distances of larvae, prolonged outflow events would be needed to reduce the estuary-wide population of *Potamocorbula* (Thompson and Parchaso 2012). Therefore, sustained reduction of *Potamocorbula* at brackish water restoration sites would require controlled floods during most if not all years, which may not be a feasible management strategy for long-term control. Freshwater conditions might also favor establishment of *Corbicula*, discussed next.

Corbicula is generally found in areas of the estuary where salinity is <2, so its distribution is also closely tied to freshwater flow. During high flow years, *Corbicula* are found as far west as Grizzly Bay, and in the upper reaches of Suisun Marsh. During low flow years they may be reduced in abundance or completely absent from these regions as the salinity field moves eastward during the fall (Thompson and Parchaso 2013). While freshwater flow may control *Corbicula's* distribution, increasing salinity would be an unpopular management strategy for reducing *Corbicula's* distribution. If flow management strategies reduce freshwater flows in the future in response to increased pressure for human use of fresh water, *Corbicula's* distribution to the Central and Eastern Delta may be reduced as the salinity increases, but an increase in salinity will also lead to an increase the distribution of *Potamocorbula* (Thompson and Parchaso 2013). Climate change may make *Corbicula*'s distribution less consistent between years due to the expected increase in climatic variability and frequency of extreme conditions (Cloern et al. 2011), but their ability to rapidly re-colonize areas with appropriate water quality means that management strategies implemented now may not prevent future impacts.

Tier 2: Site Attributes

Link to Contaminants Model

Contaminants

While there has been little direct evidence of contaminants controlling clam populations, there have been many studies suggesting clams experience sub-lethal toxic effects from a variety of pollutants common in the SFE. The importance of contaminants in controlling clam population is not clear, and likely contaminants have the largest impact when combined with other stressors. It will be important to study contaminants in clams during restoration monitoring because there are many documented effects of bioaccumulation of contaminants in clams affecting the rest of the food web (see Bioaccumulation section below).

Numerous common contaminants in the SFE have been found to affect *Potamocorbula* physiology, and some may be severe enough to affect population size. Field and lab studies in the SFE have shown that high concentrations of heavy metals in *Potamocorbula* tissues, especially cadmium and chromium, led to atypical spawning and tissue lesions in liver and reproductive organs (Teh et al. 1999, Clark et al. 2000). Another study found a 40% reduction of reproductive individuals under elevated silver concentrations (Brown et al. 2003).

Corbicula's population dynamics may also be influenced by contaminants. A few studies have hypothesized that exposure to pollutants may be driving *Corbicula*'s distribution (e.g. Brown et al. 2007, Sousa et al. 2008), and others have linked differences in condition and growth rate to contaminants in the field (Cataldo et al. 2001, Bouldin et al. 2007). Lab experiments have shown *Corbicula* is able to regulate and dispose of some contaminants, so low-level doses are not certain to impact their population growth or distribution (Cherry et al. 1980). However, some have documented lethal and sub-lethal effects of a variety of heavy metals, including Cd, (Inza et al. 1998b, Villar et al. 1999), Cu (Villar et al. 1999, Croteau et al. 2004), Hg (Inza et al. 1998a), Sn (Sebesvari et al. 2005), and Se (Fournier et al. 2005). Metal concentrations within *Corbicula* tissues in the SFE may be high enough to influence individual survival (Luoma et al. 1990). Poisons and heavy metals have been used to control *Corbicula* populations in some water control structures; however, using contaminants to influence clams in the wild is a poor choice due to non-target effects.

Substrate:

Potamocorbula occurs in all sediment types except hard rock surfaces (i.e. silt, clay, hard-pack clay, sand, gravel, peaty mud, and shell hash) (Thompson and Parchaso 2012). *Potamocorbula* generally occurs in the mid-intertidal to subtidal zone, though it occasionally forms large populations in high intertidal areas as well (Thompson and Parchaso 2012). Clams may live with one-half to two-thirds of their shell exposed to the water column.

Corbicula can live in all water depths and on all substrate types, including concrete and peat, and can be found within *Egeria* beds and tule stands. *Corbicula* is believed to be an opportunistic species that rapidly invades newly developed water conveyance canals (DWR 1967, Eng 1979) and dredged river channels (Kraemer 1979b, Paunović 2007). *Corbicula* will burrow shallowly in sediment, or may attach to solid substrates (e.g. concrete) with byssal threads until individuals exceed 5 mm in length, when their byssus disappears (Thompson and Parchaso 2012).

While *Corbicula* are opportunistic with respect to substrate, there are some habitats in which it has higher success than others. It appears to prefer sandy, well-oxygenated substrates over substrates with larger or smaller particle sizes (Belanger et al. 1985). It is also successful in submerged aquatic vegetation where the combination of filter feeding and deposit feeding may be particularly beneficial (Wickel 2011).

Clams can act as ecosystem engineers by changing the sediment. For example, once established in a concrete canal *Corbicula* causes accretion of mud and organic matter, sufficient that adults are capable of maintaining their position even after they lose their byssal threads (Prokopovich 1969, Eng 1979). *Corbicula's* activity may also bioturbate the sediment sufficiently to change nutrient concentrations in the sediment. One study found total oxygen and microbial activity increased and resulted in an increased flux of soluble reactive phosphorus due to clam bioturbation (Zhang et al. 2011*)*.

Dissolved Oxygen:

Low levels of dissolved oxygen (DO) tend to be less stressful for bivalves than other aquatic animals due to their ability to reduce their metabolism and close their shells in times of poor water quality. However, extended periods of low DO may cause massive die-offs, especially when accompanied by high temperatures and stressful salinities.

There is little DO tolerance data available for *Potamocorbul*a, but one study found them in areas with oxygen concentrations from 3.2 to 6.3 ml L^{-1} (Kamenev and Nerasov (2012)). Other studies do not list specific values, but describe *Potamocorbula* as having a high tolerance for low DO (McEnnulty et al. 2001, Sato and Koh 2004).

Corbicula is not as tolerant to low DO as *Potamocorbula*. *Corbicula* cannot survive prolonged periods of hypoxia (pO2 <23.8 torr, 20% saturation), and cannot survive short periods of hypoxia (5-10 days) under increased temperatures (Johnson and McMahon 1998). Even if *Corbicula* does not die off, growth is impaired below a threshold of between 1-3 mg/L (Belanger 1991). When temperature rises, oxygen concentrations drop and the combined conditions can lead to large scale die-offs (Cherry et al. 2005). Temperatures >30ºC decrease oxygen uptake in *Corbicula* (McMahon 1979a), and numerous die-offs have been linked to low DO and/or high temperatures (Cherry et al. 2005, Ilarri et al. 2010). In addition, *Corbicula* does not have a capacity to regulate oxygen consumption with increasing hypoxia, as is common in some freshwater bivalves (McMahon 2002). While low DO conditions are rare in the well-mixed shallows of the SFE, discharges from nearby managed wetlands or waste-water treatment plants may cause decreases in DO that could lead to die-offs in restoration sites.

Temperature

While temperature does not appear to limit distribution of either species within the SFE (Thompson and Parchaso 2012, 2013), temperature is an important factor because it affects grazing rate (See Grazing Rate below).

Neither distribution nor reproduction of *Potamocorbula* appears to be temperature limited within the range of water temperatures commonly observed in the estuary (5-25º C). If the species identification is accurate, *Potamocorbula*'s native range is from 22º to 55º N latitude in the western Pacific and thus it is unlikely that the temperature regime in SFE would be limiting. *Potamocorbula* occurs between 7.5-15.8º C in Amur river (the warmest part of a relatively cold estuary) and are believed to be limited by the cold temperatures there (Kamenev and Nekrasov 2012).

Temperature tolerances of *Corbicula* exceed water temperatures observed in the system;the broadest range for adult *Corbicula* reported is 0-34º C (Rodgers et al. 1979). Heat shock studies have confirmed a maximum temperature for *Corbicula* of 38ºC (Nascimento et al. 1996), though as discussed above, high temperatures make individuals less tolerant to low DO (see Dissolved Oxygen). The minimum temperature where populations were observed in the SFE was 6 ºC (Eng 1979) and the maximum where they were observed was 34ºC (Heinsohn 1958). Reproduction generally requires water temperatures above 10ºC, so low winter temperatures at some locations may limit reproduction but does not appear lethal to adult *Corbicula*.

Temperatures in > 38ºC or <2ºC for a prolonged period will kill *Corbicula*. However, sustained low temperatures <2ºC have not been observed in the SFE and high temperatures exceeding 38ºC would likely only be observed where water residence time is long, air temperature and irradiance are high, and the water is shallow. Non-lethal effects may occur at much lower temperatures since temperature increases the ventilation rate and thus increases the exposure of the animals to adverse environmental conditions (Doherty and Cherry 1988). For example, metabolic suppression observed starting at 30ºC in the lab may contribute to massive die-offs where temperatures were above 30ºC, but below the lethal limit of 38ºC (Mattice and Dye 1975).

Salinity:

The broad-scale distribution of both species is determined primarily by salinity. Larvae of both species are more sensitive to salinity than the adults, so salinity conditions during reproduction and settlement determine where clams establish in any particular year, while adults may be distributed over a larger area due to the previous years' salinity regimes (See Hydrology above for more on salinity controls on distribution in the SFE). The difference in larval tolerance (Potamocorbula > 2 and Corbicula < 2) largely controls the observed distribution.

The salinity range for adult Potamocorbula is 0.1-32 (Carlton et al. 1990, Werner and Hinton 2000) but recruits have a narrower salinity tolerance, between 2 and 30 (Nicolini and Penry 2000). This range is broad enough to encompass most of the northern estuary in most seasons and years, but restricts recruits from establishing in freshwater. *Potamocorbula* is the dominant species in the salinity range of 1-18 in the SFE and occurs in salinities that range from 0-23.5 in other estuaries (Kamenev and Nekrasov 2012). *Potamocorbula* and *Corbicula* overlap at a salinity range of about 2, but *Potamocorbula* is replaced by *Corbicula* when they are located upstream of the tidal excursion of bay water. This is similar to the distribution for the same species on the Yangtze River where they overlap but the benthic communities are dominated by *Potamocorbula* downstream and *Corbicula* upstream (Chao et al. 2012).

While *Corbicula* adults can tolerate salinities outside of their observed distribution, they are frequently only found in fresher water due to larval salinity requirements. Adults have occasionally been reported in the field at salinities as high as 10 (Heinsohn 1958, Siegfried et al. 1980) and 17 (Evans et al. 1979). Laboratory studies found individuals could survive even higher salinities for short periods of time, but would stop siphon activity or reduce metabolism and die after prolonged exposure to salinities greater than 10-12 (Evans et al. 1979). Reproduction is only successful at salinities less than 4 (Heinsohn 1958, Foe and Knight 1986), and populations in the SFE are rarely found in regions with an average salinity greater than 2 (Thompson and Parchaso 2013).

Water Depth:

The water depth in any particular location within the restoration site is determined by overall site bathymetry and hydrology. Water depth influences clam distribution through desiccation in the intertidal zone, and also has important interactions with residence time and grazing rate to control phytoplankton biomass.

Potamocorbula generally occur in the mid-intertidal to subtidal zone, though occasionally form large populations in the high intertidal as well (Thompson and Parchaso 2012). *Corbicula* are also found in the intertidal zone and can withstand between 6-27 days exposed to the air, dependent on weather conditions (McMahon 1979b). Drying of *Corbicula*-invaded areas may be a possible management strategy in restoration projects, since they are often extirpated from shallow areas during low-flow years.

Beyond the intertidal zone, no relationship has been found between clam establishment and depth for either species (Siegfried et al. 1980). However, water depth does have a large effect on clams through its effect on phytoplankton biomass (see Control of Phytoplankton Biomass below). Phytoplankton production is expected to increase if the water is shallower due to increased light availability. However, If the water column is well-mixed, the benthic bivalves can deplete the water column faster if the water is shallow, leading to lower net phytoplankton production (Lucas and Thompson 2012). Therefore, the relationship between water depth and phytoplankton production may be highly variable in clam-invaded areas.

Residence time:

Residence time interacts with water depth, turbidity, and grazing rate to control phytoplankton biomass (see Control of Phytoplankton BIomass below). Because of this, traditional restoration theory that prescribes "slower is greener" may no longer apply in clam-invaded ecosystems. If grazing rate exceeds phytoplankton growth rate, then longer residence times will result in lower phytoplankton biomass (Lucas and Thompson 2012). If planners want to maximize phytoplankton export, restoration plans with decreased or varied residence times may be beneficial in areas where clams are expected to invade.

Turbidity:

Turbidity may affect clams growth and grazing rate either directly through increased energy expenditure during filtration or indirectly by affecting light penetration and phytoplankton productivity. (see Control of Phytoplankton Biomass below, and General Tidal Wetland model).

High turbidity due to suspended sediments may increase energy required for bivalves to filter water. High particulate concentrations will cause bivalves to produce more pseudofeces in order to eliminate indigestible particles in the water. When the pseudofeces production rate is high, energy expended to produce pseudofeces may become sufficiently high that the bivalves stop feeding (Gunther et al. 1999). There have only been a few studies on the response of *Corbicula's* filtration rate to suspended sediments, and fewer studies of the same on *Potamocorbula*. Laboratory studies showed no effect of suspended sediments up to 150 mg/L on *Corbicula* filtration rate (Mattice 1979, Foe and Knight 1985). A report of population die-offs during a high-turbidity (410 mg/L) flood in the Ohio River (Bickel 1966)suggests turbidity affects clams at a some threshold between 150 and 410 mg/L.

In other systems, filtration of the water column by *Corbicula* has reduced overall turbidity. For example, in the Potomac River filtration by *Corbicula* caused a reduction in chlorophyll *a* (Cohen et al. 1984); the resulting increase in water clarity brought about startling ecosystem changes (Phelps 1994). Submerged aquatic vegetation that had been absent for 50 years returned, fish that lived in the vegetation returned with a 7-fold increase in population numbers, and bird populations also increased. However, in the SFE, turbidity is controlled largely by wind and tidal forces resuspending sediment, rather than by phytoplankton. The SFE also has higher sediment loads in comparison to estuaries where water clearing was reported after *Corbicula* invasion.

Tier 3: Clam responses

Colonization Probability:

Because both clam species may establish on a wide range of substrates, colonization probability is more related to freshwater flows and proximity to source populations than substrate (See Salinity, Transport, and Hydrology, above)

Potamocorbula colonization is more likely in brackish and salt water (salinities 2-30) (Thompson and Parchaso 2012). Microhabitat hydrodynamics will also affect colonization probability; if turbulence near the bottom prevents larvae from attaching to the bottom then juveniles cannot settle out. Large densities of adult *Potamocorbula* may increase turbulence near the bed and prevent recruits from settling (Crimaldi et al. 2002). Furthermore, adult *Potamocorbula* may filter-feed on their own larvae, as they do on other microzooplankton (Kimmerer et al. 1994, Thompson and Parchaso 2012). The current wide distribution of adults, the dispersal ability of planktonic larvae, and the wide variation in flows in the SFE means that *Potamocorbula* is able to colonize new habitat rapidly, as was seen when the species was first discovered (Nichols et al. 1990).

Corbicula colonization is largely determined by low salinity, high dissolved oxygen, and proximity to nearby source populations. As discussed above (see Hydrology), this means it often colonizes areas further down-estuary in high-outflow years where salinity increases (Thompson and Parchaso 2013). Within a given site, it may be patchily distributed, possibly due to low juvenile dispersal distance (e. g. McMahon 1999). In lentic habitats, clams are most frequently found in in areas with high primary productivity. In lotic habitats, clams are most frequently found in areas where water velocity is high enough to deliver food and DO but low enough not to erode adults (Thompson and Parchaso 2013). Because *Cobribula* has wide environmental tolerances, is not limited by substrate (McMahon 1991), and may continue to move as juveniles and adults, it can quickly invade newly disturbed habitat faster than native bivalve species, such as freshwater mussels, can establish (Thompson and Parchaso 2013).

Clam Population Growth

If salinity and water quality are within a species' environmental tolerance, population growth for both species is most commonly food-limited. The relationship between food supply and population growth is better understood for *Corbicula* than *Potamocorbula*.

The effect of food supply on *Potamocorbula* populations is complex and requires further investigation. Reproductive output may be linked to food availability as well as salinity (Parchaso and Thompson 2002), but this relationship has not been well described, partially due to a lack of data on other food sources other than phytoplankton (Werner and Hollibaugh 1993, but see Greene et al. 2011, Kimmerer and Thompson 2014). There is variation in both individual metabolic rate and population growth rate between years, and a strong seasonal difference in growth rates between winter and summer (Parchaso 1993, Miller and Stillman 2013), which may be due to differences in food as well as temperature and salinity (Thompson and Parchaso 2012).

Reproduction and population growth in *Corbicula* are believed to be controlled primarily by food availability, and possibly by food quality (Hoese 1967, Doherty et al. 1987, Cataldo and Boltovskoy 1999, Rajagopal et al. 2000, Denton et al. 2012). While there have not been specific field studies showing a relationship between food availability and *Corbicula* population growth in the SFE, many correlational studies have linked seasonal peaks in spawning to food availability (Heinsohn 1958, Foe and Knight 1986), and lab studies have demonstrated slower individual growth under food-limited circumstances (Foe and Knight 1985). While food supply does appear to limit *Corbicula* population growth in some areas of the SFE, manipulating food supply is not considered a useful strategy for management since lowering food production for *Corbicula* will also lower food availability for at-risk fish species.

Density-dependent population regulation has been shown in both species. *Potamocorbula* may avoid settling out of the water column in areas with high adult density (Crimaldi et al. 2002) and adults may filter larvae out of the water column (Kimmerer et al. 1994). There is less evidence for density dependence in *Corbicula*; however, food limitation was hypothesized as the cause of low recruitment of *Corbicula* in the Altahama River in Georgia (Sickel 1979) due to adult clams outcompeting juveniles.

Grazing Rate

Multiple factors affect bivalve grazing rate (adapted from Thompson and Parchaso 2013).

1. Grazing rates are a function of biomass. Higher total clam biomass will increase total grazing rate (Thompson and Parchaso 2013).
2. Species identity: *Potamocorbula* may pump up to four times faster than Corbicula per gram of biomass.

3. All metabolic functions are a function of temperature, so bivalves will increase their filtration rate as temperature increases up to some threshold. Multiple studies have tested for a temperature threshold in *Corbicula.* These studies have found thresholds anywhere from 24°C (Mattice 1979) to >30°C (Lauritsen and Mozley 1989). The relationship between filtration rate and temperature in *Potamocorbula* is still unknown.
4. High water column vertical mixing rate increases grazing rate because it prevents water directly adjacent to the benthic community from becoming food depleted. In the SFE, high winds and tidal action generally keep the water column well mixed (Lucas and Thompson 2012).
5. Other aspects of water quality, particularly salinity and turbidity, can affect filtration rate via the same mechanisms that affect growth rate.
6. Cell size will also limit filtration rate if the cells are too large or small for a particular species to filter (Reid et al. 1992)

Control of phytoplankton biomass: Grazing Rate + Turbidity + Water Depth + Residence time

In order to predict whether a given area will have a net export of phytoplankton productivity, it is important to understand the relationship between grazing rate, water depth, residence time, and turbidity (Figure 2, control of phytoplankton sub-model). In restoration, shallower water and long residence times have been traditionally hypothesized to maximize phytoplankton production because the photic zone can include the entire water column. However, as discussed in the Water Depth section above, shallow water also allows the benthic community to more effectively graze phytoplankton out of the water column. Longer residence times not only allow phytoplankton to maximize time spent in the shallow, photic zone of the wetland, but also allow benthic grazers maximum time to filter the entire water column. Turbidity, along with water depth, decreases light availability for phytoplankton growth while only slightly decreasing grazing rates if at all (see Turbidity section above). Because the relationships between phytoplankton growth rate and turbidity, residence time, water depth, and grazing rate are non-linear, small changes in magnitude and balance between these factors can alter the final outcome in surprising ways.

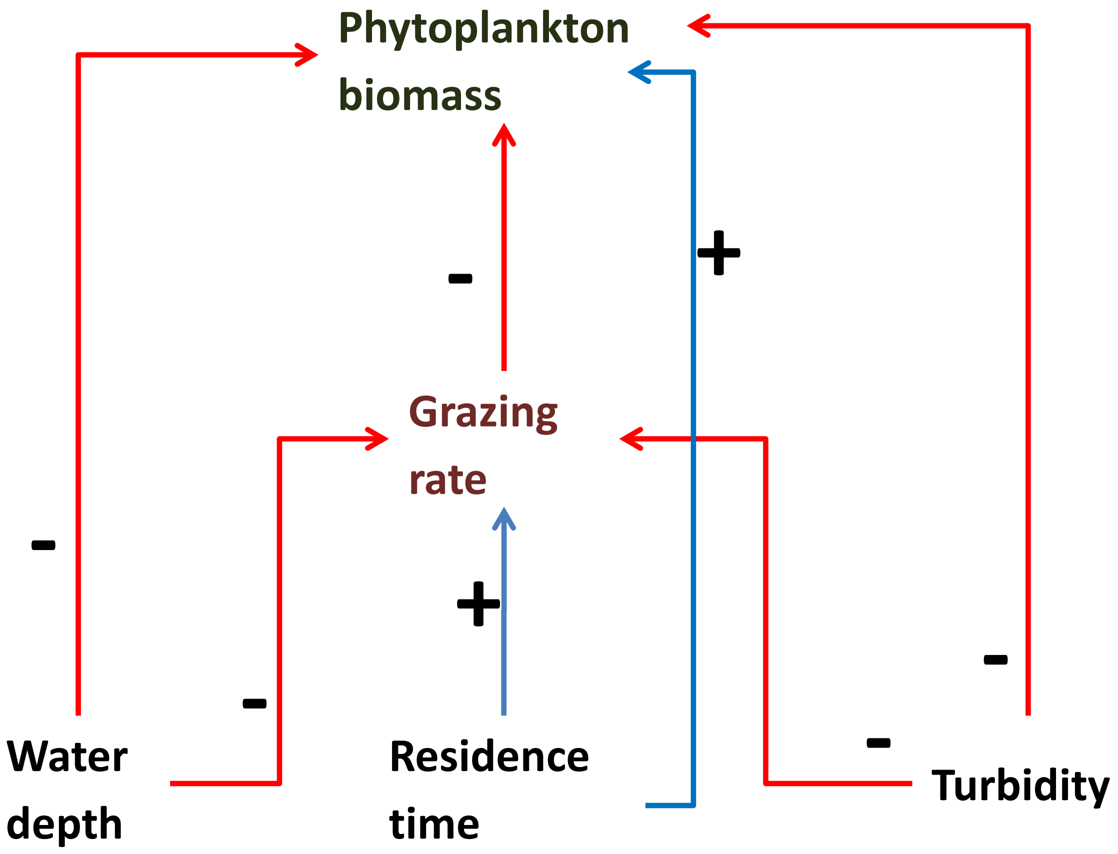

Figure 2: Submodel describing how residence time, turbidity, and water depth interact with grazing rate to control final phytoplankton biomass in a wetland restoration site. Red arrows indicate negative relationships and blue arrows indicate positive relationships.

Lucas and Thompson (2012) describe a simple mathematical model that explores the effects of combinations of water depth, residence time, grazing rate, and phytoplankton growth rate on net growth of phytoplankton biomass. They demonstrate that slight differences in combinations of these factors may lead to a variety of phytoplankton productivity outcomes, and shallower, slower water bodies are not always the most productive ones. This simple model result is supported by field studies which found that, while gross phytoplankton productivity was highest in shallow areas, net production was most likely controlled by invasive clam presence (Lopez et al (2006), and a deeper restoration site (Mildred Island) had net phytoplankton export, while a shallower site (Frank's track) was a phytoplankton sink, most likely due to differences in benthic grazing rates (Lucas et al (2002a). Using models of benthic grazing rates in combination with hydrodynamic and phytoplankton growth models may be important to find an optimal combination of factors that allows new restoration sites to export phytoplankton productivity.

Tier 4: Food web alteration

Link to the Food Web model

Phytoplankton and zooplankton biomass

As discussed above, invasive bivalves can control phytoplankton biomass if the turbidity, water depth, and residence time are appropriate. They may also control zooplankton biomass either through competition for food (e.g. phytoplankton) or through direct filtration of microzooplankton, including early life stages of meso- and macrozooplankton and meroplankton such as bivalve larvae. To date, there is more support for *Potamocorbula* controlling zooplankton biomass than there is for *Corbicula*. However, few studies have investigated this with *Corbicula* in the SFE, so it cannot be ruled out as a potential effect. Both species should monitored in wetland restoration sites, since reduction in zooplankton biomass will limit food for at-risk fish species.

Potamocorbula can filter and assimilate both phytoplankton and bacteria (<1.2 µm) from the water column (Werner and Hollibaugh 1993). High clam production in areas with low phytoplankton productivity (Thompson 2005) suggests clams often depend on carbon sources other than phytoplankton. Several species of zooplankton have declined coincident with the decline in phytoplankton in the SFE, at least partially due to food limitation (Kimmerer 2002, Feyrer et al. 2003). Some of the declines in copepods may also be due to direct filtration of juvenile zooplankton by *Potamocorbula* (Kimmerer 2006).

There have been no studies in the SFE that directly implicate *Corbicula* in reduction of any zooplankton populations either indirectly or from direct filtration. Some studies indicate *Corbicula* selectively retains phytoplankton rather than zooplankton in their stomachs (Hill and Knight 1981). Adults are capable of filtering microzooplankton, but it is unknown if microzooplankton are actually assimilated or are passed through (Reid et al. 1992, Scherwass and Arndt 2005).

Predators

Strong shells limit the susceptibility of both species to predation (for example, they may be passed through the digestive system of sturgeon alive (Kogut 2008)), but their shallow position in the substrate means these clams are easier prey for birds to access than many native bivalves in the SFE. In some cases, predation by migratory birds may be sufficient to effectively control populations of *Potamocorbula*. While the addition of easy to access prey may be an energetic advantage to these predators, predators may also be harmed by this abundant food resource because *Potamocorbula* bioaccumulates of contaminants (Poulton et al. 2002).

Known aquatic consumers of *Potamocorbula* in the SFE include the Dungeness crab (*Cancer magister*, Carlton et al. 1990, Stewart et al. 2004), the Sacramento Splittail (*Pogonichthys macrolepidotus*, Deng et al. 2007), White Sturgeon (*Acipenser transmontanus*, Adams et al. 2007, Urquhart and Regalado, 1991, Kogut 2008), and the Green Sturgeon (*Acipenser medirostris*) (Adams et al. 2007). Most bottom feeding fish in the area probably consume *Potamocorbula*, due to the clam's wide distribution and dense populations (Thompson and Parchaso 2012). Many birds consume adult *Potamocorbula*, including the Greater and Lesser Scaup (*Aythya marila* and *A. affinis*, Poulton et al. 2002) and the Surf Scoter (*Melanitta perspicillata*, Hunt et al. 2003).

There is little data on predation on *Corbicula* in SFE but it is known from studies of other systems that they are consumed by many bottom-feeding fish, large invertebrates, diving ducks, mammals, and sometimes by humans. Documented fish predators on *Corbicula* that are present in the SFE include Starry flounder (*Patichthys stellatus*) (Haertel and Osterberg 1967), redear sunfish (*Lepomis microlophus)*, bluegill (*Lepomis macrochirus*), sturgeon (*Accipenser spp*.), channel catfish (*Ictalurus punctatus*), common carp (*Cyprinus carpio*), blue catfish (*Ictalurus furcatus*), yellow bullhead (*Ameiurus natalis*), and largemouth bass (*Micropterus salmoides*) (McMahon 1983). Invertebrate predators include the Chinese Mitten Crab, (*Eriocheir sinensis)* (Rudnick and Resh 2005) and the red swamp crayfish (*Procambarus clarkia)* (Covich et al. 1981). Diving ducks and other birds are also probable predators (Thompson and Sparks 1977). To date, there have been no studies showing large-scale top-down control of *Corbicula* in the same way that has been shown for *Potamocorbula*.

Bioaccumulation of contaminants

Filter-feeders can have negative effects on higher trophic levels by bioaccumulating toxic chemicals and transferring them to predators. Any restoration action that releases contaminants may mean that those contaminants bioaccumulate in clams and get transferred to their predators.

Biomagnification of selenium has led to toxic levels in the livers of many *Potamocorbula* predators (White et al. 1987, 1988, 1989, Urquhart and Rigelado 1991, Linville et al. 2002), leading to teratogenic deformities and reproductive problems (Stewart et al. 2004, Teh et al. 2004). Selenium biomagnification is greater in benthic food webs than pelagic food webs due to the greater potential for bioaccumulation in bivalves than in zooplankton (Stewart et al. 2004). Selenium biomagnification has been implicated in population-level declines of White Sturgeon, a common predator of *Potamocorbula*, in the SFE (Presser and Luoma 2006).

Corbicula can bioaccumulate a range of metals (Ag, As, Cd, Cr, Cu, Hg, Ni, Pb, Se, Zn) with accumulation factors in the 1-2 range (Gunther et al. 1999). *Corbicula* can also bioaccumulate methyl mercury and make it available to higher organisms. Selenium is generally found in lower concentrations in *Corbicula* (1.4-4.8 mg kg^{-1}) than in *Potamocorbula* (3.7- 20 mg kg^{-1}), but is present in levels high enough to be of concern in both species (Johns et al. 1988, Lee et al. 2006).

Both bivalve species are known to bioaccumulate PCBs and chlorinated pesticides, and they likely transfer these contaminants to predators (Pereira et al. 1992, Brown and Luoma 1995, Gunther et al. 1999, Pereira et al. 1999), but trophic transfer of these contaminants has not been studied in the field (Thompson and Parchaso 2012).

Competitors

There is some evidence for density-dependent population growth in *Potamocorbula*, via both intraspecies and interspecies competition (see Population Growth above). High densities of tube-dwelling amphipods or native bivalves may reduce recruitment, possibly by competition for food or by disturbance of water flow near the sediment (Friederichs et al. 2000).

Corbicula competes for food and space with threatened and endangered freshwater bivalves in laboratory studies, but there is no agreement whether competition occurs frequently in the field (Strayer 1999). Many studies have shown that native bivalves and *Corbicula* have non-overlapping distributions and that it is environmental disturbance that eradicates the native bivalves, resulting in invasion by and dominance of *Corbicula* (eg. Sickel (1973, 1986), Kraemer

(1979a), McMahon (1991), Vaughn and Spooner (2006), Darrigran (2002)). There are some native freshwater mussels in the Delta and its tributaries, but they are poorly studied and not as diverse or numerous as freshwater mussels are in the eastern US where competition between native bivalves and *Corbicula* has been studied (Williams et al. 1993, Cherry et al. 2005, Haag 2012).

Although it has not been proven that competition with *Corbicula* is responsible for extirpation of native bivalves and other benthic species, invasive clams can stress other species through indirect mechanisms. Native bivalves can be more resistant to low dissolved oxygen and high temperature events than *Corbicula.* Large die-offs of *Corbicula* during low DO or high temperature events, have been observed to produce sufficiently high ammonia concentrations to kill juveniles and adults of native bivalves (Cherry et al. 2005).

At-Risk Fish Species:

Link to the Chinook Salmon and Delta Smelt models

Reduction in abundance of phytoplankton and zooplankton caused by clam grazing has had important effects on the food web of the estuary. These changes in the food web are hypothesized to be a major factor behind the pelagic organism decline of the early 2000s, when Striped Bass, Threadfin Shad, Delta Smelt, and Longfin Smelt all declined in population size (Sommer et al. 2007, Baxter et al. 2010) (See Salmon and Delta Smelt Models). However, food limitation was not the only factor behind those declines.

Conclusion: Restoration Implications

Increasing primary productivity is identified as one of the goals of much of the planned restoration in the Delta. For example, at the Fish Restoration Program’s Prospect Island Project, objective #1 in the restoration plan is “Enhance primary and secondary productivity and food availability for Delta Smelt and other native fishes within Prospect Island and surrounding Delta waterways.“ (From draft Prospect Island Restoration Plan). If invasive bivalves reduce standing biomass of phytoplankton and zooplankton before it can be consumed by native fishes, either on-site or after it is exported to surrounding waters, the restoration project will not attain this goal. Using this model, managers may be able to design restoration projects to minimize clam grazing. Tidal restoration is likely to directly change water depth, residence time, and turbidity. Monitoring the interplay of grazing rates with these factors may allow managers to detect conditions that could lead to low productivity before clams invade, plan for adaptive management actions, or plan for long-term management of restoration sites, such as regular drying of a site to reduce clam populations.

Control methods for invasive bivalves after they establish are limited. Industrial water-supply system managers have developed numerous techniques including biocidal chemicals, UV radiation, mechanical removal, thermal shock, desiccation, freezing, and oxygen deprivation (Sousa et al. 2014). However, most of these are not viable strategies in open water systems because of non-target effects. Therefore, once clams have become established in a restoration site, they are likely to remain there, unless the site is managed to undergo regular drying.

Despite the problems caused by invasive bivalves, species diversity in the estuary has remained relatively robust, and while relative abundances of the species involved have changed, there have been few extinctions. Multiple factors may account for this robustness, including its complexity (Kimmerer 2002), presence of microbial biomass not currently well understood (Thompson and Parchaso 2012), and surprising compensatory processes, such as behavioral changes to other filter feeders in the estuary (e.g., migration of Northern Anchovies out of the northern estuary following reduction in food resources there (Kimmerer 2006)). Due to the difficulty in eradicating clams once they invade, monitoring the impacts of clam invasions and learning to encourage resilience of native species may be more productive than attempting to remove all clams from a restoration site.

Corbicula and *Potamocorbula* are invasive bivalves currently impacting the ecosystem of the SFE, but other invasive bivalves may be of concern in the future, particularly the zebra mussel (*Dreissena polymorpha*) and the quagga mussel (*Dreissena bugensis*) which have caused both ecosystem and water supply problems in other freshwater systems. Those mussel species are currently wreaking havoc on the Great Lakes, the Mississippi River, and the Hudson River (Whittier et al. 2008, Gallardo et al. 2013). In January 2007, the quagga mussel was discovered in the lower Colorado River system and the Colorado River Aqueduct, near and within California waters (Cohen 2008) and Lake Mead (McMahon 2011), and in 2008 zebra mussels were found in San Justo reservoir in San Benito County(Emerson et al. 2015). *Dreissena* species spread easily, particularly by recreational boat bilge water and hull fouling. A recent analysis of ecological requirements of *Dreissenia* found that 50% of sites measured in the Delta had high or medium probability of supporting either zebra or quagga mussels based on their calcium and temperature thresholds (Cohen 2008). Most of California is considered “high risk” of invasion in comparison with the rest of the US (Whittier et al. 2008). An invasion of either one of these species at a restoration site has the potential to significantly impede restoration goals.

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Interagency Ecological Program

Tidal Wetlands Restoration Monitoring Project Work Team

Contaminants Conceptual Model

DRAFT

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PREFACE

This chemical contaminant conceptual model for tidal wetland restoration is an effort to summarize information on what is currently known about contaminant effects on restoration and the effects of restoration on contaminants. This model incorporates information from multiple Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) conceptual models (Alpers et al 2008; Werner et al 2008a; Werner et al 2008b; Windham-Myers and Ackerman 2012; Presser and Luoma 2013), data from the Delta Regional Monitoring Program, and the recent literature. Previous conceptual models provide a comprehensive summary on specific contaminants of concern, and users should consult the more detailed DRERIP models for contaminant specific investigations. DRERIP models have been developed for mercury (Alpers et al 2008; Windham-Myers and Ackerman 2012), selenium (Presser and Luoma 2013), pyrethroids (Werner and Oram 2008), and chemical stressors (Werner et al 2008).

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Introduction

The ~~Delta and~~ SF Estuary includes urbanized and agricultural landscapes that are highly impacted by anthropogenic activities. Current and legacy contaminants are widespread and many waterways are impaired (http://www.waterboards.ca.gov/water_issues/programs/tmdl/integrated2010.shtml) . However, ~~, yet~~ the fate, transport and effects of contaminants are ~~very~~ complex~~,~~ and difficult ~~challenging and expensive~~ to monitor. Tidal wetland restoration ~~actions~~ may result in structural or functional changes at a site that increase or decrease a contaminant's bioavailability or impacts. Effects can be directly toxic or be ~~indirectly~~ transmitted ~~across multiple~~through trophic levels (e.g., phytoplankton, zooplankton, fish, birds, or humans). Impacts may be very species specific or affect broad clases of organisms~~, and are variable among species~~. Lethal effects are more likely to occur at lower trophic levels. In contrast, higher trophic level species are more likely to experience indirect or sublethal impacts~~, whi~~ch may reduce ecological fitness through impaired growth and reproduction, changes in behavior, or increases in the organism's susceptibility to disease (Werner et al. 2008).

In the environment, stressors do not act~~occur~~ individually, but rather ~~there exists~~as a complex interplay between organisms and the changing quality of their physical habitat, food quality, competition, contaminant presence, and nutrient cycling (Cairns et al. 1995). This Conceptual Model is an effort to depict the important factors in a tidal wetland that govern contaminant fate, transport and consequent direct and indirect effects on Delta Smelt and salmon. The ~~following~~ descriptions and examples are provided to help the reader~~/the practitioner~~ understand what reactions and~~/~~ transformations contaminants may undergo in the restored tidal wetland environment and their potential effects on associated aquatic organisms.

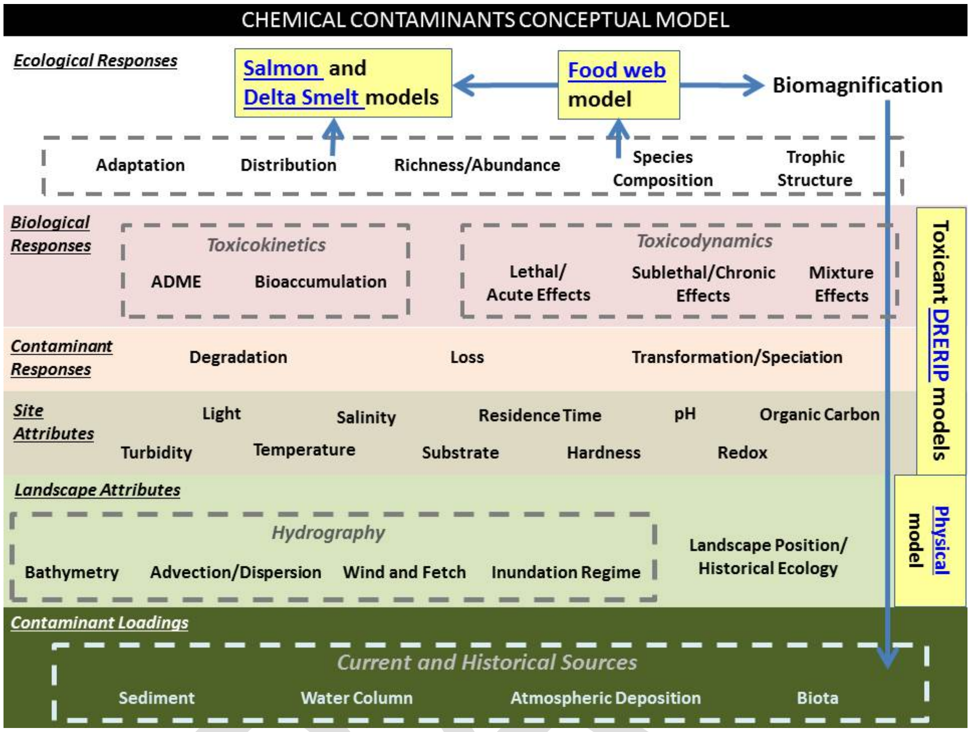

Figure 1. Chemical Contaminants Conceptual Model diagram. This model depicts a general framework of how contaminants react in a tidal wetland environment. The model links ~~makes reference~~ to other ~~detailed~~ tidal wetland conceptual models (including the Salmon and Delta Smelt Models, Food Web Model, Wetland Evolution Model, Transport model, and the various DRERIP models for contaminants and stressors). The model and this d~~iagram that should be considered when~~ developing can guide development of a site specific contaminant monitoring ~~programs for tidal wetland restoratio~~n in the Delta.

Model Structure

The Chemical Contaminants Conceptual Model is made up of six tiers. The base of the model depicts sources of contaminant loading, while next two levels represent environmental attributes that affect contaminants (concentrations), and the top three levels represent responses including chemical, biological and ecological responses. Each successive tier is dependent on and directly related to the previous tier, with additional relationships between ~~and among~~ tiers ~~also~~ indicated. External models are referenced where applicable.

Body of Model

Tier 0: Contaminant Loadings

Contaminants are chemicals that have deleterious effects in the environment. They may be natural or anthropogenic, organic or inorganic. Thousands of contaminants are present in the Delta. Sources include mining, agricultural discharge and overspray, direct application to surface water, industrial point sources, stormwater runoff, urban runoff, and waste water discharge. Transport to a tidal wetland occurs by several processes including passage via the water column, sediment transfer in the water column, and atmospheric deposition (via wet or dry deposition). Contaminants may also be present at a site due to past activities on the land.

Legacy contaminants are chemicals that have been discharged in high volumes in the past, degrade slowly and are generally sediment-associated. These contaminants are prevalent in the Delta and SF Estuary, including pesticides, metals, industrial chemicals, and flame retardants, among others. ~~These~~ Though concentrations have decreased in the Bay-Delta over the past 20 years, some still remain high enough to cause deleterious environmental effects (Conaway et al. 2007, Conner et al. 2007, Davis et al. 2007, Melwani et al. 2013, Sutton et al. 2015).

Current resident land uses, upstream of or adjacent to restoration sites, also determine the types and concentrations of aquatic contaminants. ~~Recent detections of p~~Pesticides and contaminants in Delta water bodies are frequently sourced to urban and agricultural discharges. Concentrations and loadings of these compounds are greatest during storm runoff events (Weston et al. 2012, Orlando et al. 2015). Herbicides and fungicides are generally higher~~st~~ in concentration than insecticides in the Bay-Delta ~~relative to insecticides~~ (Orlando et al. 2105). However, due to their greater potential for toxic effects ~~from~~ insecticides~~, these~~ may be of greatest interest for water quality monitoring. In water samples collected from tributaries to the Delta, toxicity to a wide range of laboratory test species has been associated with pyrethroids and fipronil, products that are commonly used i~~o~~n urban landscapes (Jorgenson et al. 2013, Weston et al. 2014a, Weston 2013).

Sediment

A significant fraction of contaminants in the Bay-Delta are associated with sediment, thus sediment transport moves~~provides a pathway for~~ contaminants ~~movement~~ into, within, and out of sites in the ~~Delta and~~ Estuary (Schoellhamer et. ~~a~~Al. 2012). Legacy contaminants, including DDT, mercury, PCBs, and a wide range of metals are especially prevalent in Delta sediments, ~~including DDT,~~

~~mercury, PCBs, and a wide range of metals~~ (Conaway et al. 2007, Conner et al. 2007, Davis et al. 2007, Oros et al. 2007). Delta waterbodies are legally impaired (Clean Water Act, Section 303d) as a result of contamination from one or more of these chemicals. Impacts of c~~C~~urrent-use hydrophobic pesticides, such as pyrethroids and fipronil, and certain pharmaceuticals/personal care products ~~are also of concern, but~~ require more research (Thompson et al. 2007, Palalma et al. 2012).

Resuspension of contaminated sediments ~~can~~ liberates sediment-bound contaminants and can lead~~ing~~ to toxicity and bioaccumulation (Roberts 2012). Mechanisms of sediment disturbance can be anthropogenic (dredging and dredge disposal, trawling, vessel movements and propeller wash) or natural (tides, storms and bioturbation) processes. These disturbances can result in episodic or continuous exposures to contaminants. Restoration construction ~~activities~~ can~~also~~ cause temporary sediment disturbance and resuspension and, thus, may ~~have significant~~temporarily increase ~~effects on~~ contaminant loads ~~during the course of construction.~~

Water column

Many contaminants enter the Delta via ~~the water column by~~ storm water runoff, waste water effluent, and other water discharges. For current use chemicals, such as pesticides, PAHs, metals and a variety of pharmaceuticals and personal care products (PPCPs), this may be the predominant loading pathway into the environment. Recent literature indicates that storm water runoff is the largest contributor of contaminant effects in the ~~Delta and~~ SF Estuary (Oros et al. 2007, Anderson et al. 2007, Weston et al. 2014), though a wide range of chemicals are discharged throughout the year from non-point sources such as agriculture, and point sources such as waste water treatment facilities (Orlando et al. 2015). The physicochemical properties of a chemical (i.e. organic carbon partition coefficient), and some environmental variables (dissolved organic carbon, pH, electrical conductivity, etc.) dictate a chemical's fractionation between the dissolved and particulate-bound forms (See Site Attributes section below). The dissolved fraction of contaminants ~~that is present~~ in the water column is generally the most bioavailable and therefore ~~incurs~~ poses the greatest risk to aquatic species.

Atmospheric deposition

Atmospheric deposition (wet and dry) also contributes to the load of contaminants in the ~~Delta and~~ SF Estuary. Chemicals, such as PAHs, that are discharged as gas into the atmosphere are most likely to result in air-water transfer by dry deposition (e.g., soot), or wet deposition via rainfall (Oros et al. 2007), however a wide variety of current use, as well as legacy pesticides and contaminants, are transferred this way (Connor et al. 2007, Bradford et al. 2012, Mast et al. 2013). Atmospheric transfer can occur over long distances. Pesticides from applications in the Central Valley are found in the High Sierra and Yosemite Valley (Fellers et al. 2004, Mast et al. 2012), and trace levels of PCBs sourced to industrial areas in North America or Asia have been found in the Arctic (Kirk and Gleason 2015). Deposited loads may, therefore, not necessarily depend on proximity to contaminant sources, and will be difficult to control at restoration sites.

On a local scale, pesticide overspray can contribute very high levels of a given pesticide in a short period of time. This can occur in water bodies adjacent to agriculture or that incur direct application of pesticides themselves (e.g. for aquatic weed control) (Riley et al. 2004, Siemering et al. 2008).

Biota

Contaminant loading also occurs via biota. Low trophic level aquatic invertebrates or algal species ~~up~~take up significant amounts of hydrophobic contaminants; mobile invertebrates~~, and where species are mobile,~~ can then transport contaminants~~ can be transported~~. Consumption by higher trophic level species results in biomagnification; the accumulation of a chemical as it passes up the food chain, increasing ~~whereby the~~ tissue concentrations ~~increase~~ at each higher trophic level (if~~when~~ elimination occurs at a slower rate than uptake).

Tier 1: Landscape Attributes

The interactions among the structural components of a tidal marsh ecosystem and its ~~the~~ hydrology~~ic components~~ are important controls of biological and ecological processes~~ associated with the marsh~~. Water fluxes across the terrestrial, marsh, and aquatic interfaces transport ~~chemicals –~~both desirable chemical constituents like nutrients and undesirable ones like contaminants~~– that are then taken up by sediments, primary producers, and consumers~~. Sediment biogeochemistry, site bathymetry, winds and tidal flow can influence contaminant loading and fate. For detailed information on these processes, refer to the Transport Conceptual Model. A brief description of important landscape attributes are provided below.

Link to Transport Model

Advection and Dispersion

Transport of chemicals occurs by advection via stream and tidal flow, and by local dispersion within the water column. Advective transport generally occurs over greater distances than dispersal (and other forms of Fickian transport) and will be greatest during times of high runoff when accumulated sediments and contaminants are mobilized by storm flows. Often the first storm flow in a water year has particularly high loads of contaminants (Orlando et al. 2015, Weston et al. 2014). Advective transport tends to be unidirectional and varies with seasonal and annual hydrological conditions. Tidal dispersion is greater at more seaward sites and is bidirectional and dispersive, unless control structures are present. In drier seasons or in drier years transport will decrease in importance, thereby raising the importance of the dispersive effects of the tides. Tidal forces vary on predictable daily, monthly and seasonal patterns.~~be driven to a greater extent by tidal forces.~~

Bathymetry

The topography of the underwater terrain will affect the influences of flows, tides, winds and other factors on contaminant fates. Bathymetry influences water and sediment movement, and ocean-estuary exchange (Barnard et al. 2013), and therefore affects the turbulent diffusion of contaminants at a site (Hemond and Fechner 2014). As such, complex bathymetry may result in patchy distribution of sediment and associated contaminants. Bathymtery also controls the degree and duration of tidal inundation which can have multiple effects on contaminaters, see below.

~~Wind and Fetch~~

~~Wind includes wind direction, magnitude, and duration. Fetch (distance over open water) will determine how winds produce waves and mix the water column. Mixing of the water column is also driven by tides and stream flow. Such mixing serves to resuspend materials from the bottom (particularly sediment, as mentioned above) and make them more available for movement via advection and dispersion. In the Delta winds are often greatest and most consistent in summer months.~~

Inundation Regime

The inundation regime is ~~defined as~~ the frequency, duration, and depth of flooding by ~~surface~~ waters in a tidal wetland~~marsh~~, and is driven largely by elevation, freshwater flow, tidal cycles, and seasonal precipitation cycles (Kneib et al 2008; Siegal et al 2010a, b). In restored tidal wetlands, the size and location of breaches and the geomorphology of the site will affect inundation regime. ~~The effects of inundation regime can be modulated through the complexity of the physical structure at the aquatic interface and geomorphology of the marsh plain. It is a key feature of the tidal wetland, and influences the characteristics and dynamics of the biological community. Further, w~~Wetting and drying cycles at sites, such as flood plains and rice fields, can result in anaerobic conditions that promote speciation of mercury to the more toxic monomethylmercury (Negrey et al. 2015) and production of harmful greenhouse gases.

Wind and Fetch

Wind direction, magnitude, and duration affects contaminant concentrations. Fetch (distance over open water) determines the strength of wind waves and mixing of the water column. Mixing of the water column is also driven by tides and stream flow. Such mixing resuspends materials from the bottom (particularly sediment, as mentioned above) and makes them available for movement via advection and dispersion. In the Delta winds are often greatest and most consistent in summer months.

Landscape Position and Historical Ecology

The position of a site on the landscape in terms of its proximity to the bay or the rivers and its distance from the main channel will control much of the contaminant characterization it will receive from its surroundings. ~~and t~~The historical uses of the site~~conditions on the site~~will ~~affect soil and sediment composition and other factors that~~ play a dominant role in contaminant effects when the site is opened to tidal inundation~~fate at a site~~.

Tier 2: Site Attributes

The concentration of a contaminant ~~at a given point in the water column~~ is a product of the input to and output from the system (i.e. sources and sinks), including forces of advection and dispersion discussed in the previous section, as well as chemical reactions and phase distribution (e.g. dissolved v~~.~~s. sediment-bound states) (Hemond and Fechner 2014). Abiotic and biotic chemical reactions, such as precipitation, transformation, degradation and metabolism are largely affected by the site conditions: temperature, turbidity, light, salinity, hardness, pH, water residence time, and microbial activity. Mass transport between phases is affected predominantly by the physicochemical properties of a contaminant (e.g. organic carbon partitioning coefficient) and the physical environmental (e.g. atmospheric pressure, organic carbon concentration, etc.).

Temperature

The kinetics of chemical reactions~~,~~ and, thus, the fate of contaminants in a system, is strongly dependent on temperature. Rates of metabolism and degradation of organic compounds increase with temperature, as does the rate of speciation of inorganic chemicals (Werner et al. 2008, Hemond and Fechner 2014). For example, the rate of bacteria-facilitated transformation of mercury to methylmercury increases with increased temperature (Benoit et al, 1998; Mason et al, 1995; Alpers et al., 2008). On the other hand, high water temperature encourages growth of aquatic plants, phytoplankton and algae, which, through the effect of shading, inhibits the photodegradation of contaminants and other UV-dependent transformations (Werner et al. 2008).

Temperature can also affect the toxic effects of contaminants. Pyrethroid pesticides, for example, are more toxic to vertebrate and invertebrate species at lower temperatures (Weston et al. 2009).

Turbidity and Organic Carbon Concentration

Turbidity is a measure of water clarity, which is strongly correlated with particulate matter concentration. Particulate matter can be sedimentary materials, like clay, silt or sand, or living organisms such as algae or plankton. Many contaminants readily sequester to particulate matter, therefore increased turbidity decreases the bioavailability of contaminants such as PAHs, PCBs, and many organic insecticides and metals (Oros and Ross 2004; Oros and Werner 2005, Werner and Oram 2008). Some bound contaminants are still bioavailable to detritivores ~~that are exposed~~ via diet (de Perre et al. 2014, Hare et al. 2003, Maul et al. 2008). Sediment particle size and organic matter content influence the bioavailability of sequestered contaminants, with uptake being most closely associated with fine organic-rich sediments. Partitioning of organic compounds to organic matter and sediment depends on the chemicals' organic carbon adsorption coefficient (K_{oc}) or the octanol-water partition coefficient (K_{ow}), solubility in water, volatility (Henry's law coefficient), and half-life (substrate-dependent) (Werner 2008).

Comment [BH1]: Adsorb?

High turbidity also reduces UV penetration in the water column and benthic habitats, and will therefore inhibit aquatic plant, algae and phytoplankton growth, and hinder UV-dependent processes, such as photodegradation or photoactivation. For example, the toxicity of polycyclic aromatic hydrocarbons (PAHs) is enhanced in the presence of UV-light (Monson et al, 1999), therefore high turbidity would reduce the toxic potential of PAHs.

Salinity, pH, and Hardness

Salinity, pH and hardness are measures of inorganic chemical concentrations in water (sodium chloride or other salts, hydrogen ions, and calcium and magnesium, respectively). As such, these measures affect the complexation and speciation of contaminants, such as pesticides, ammonia, and various metals, which influence their bioavailability, uptake and toxicity to aquatic life (Werner 2008). Environmental and internal pH drives speciation of compounds between their ionized and unionized forms, and strongly affects their bioavailability. Ionized compounds are more hydrophilic than their unionized counterparts and therefore more readily bind and adsorb into biological tissues (Klaasen 2008). Ammonia toxicity is also strongly dependent on pH, as well as hardness (and temperature). High pH ~~corresponds to low concentrations of free hydrogen and ~~favors the more toxic ammonia (NH_3) species, over ammonium (NH_4^+). Salinity and hardness can also affect the toxicity of metals. In general, the toxicity of metals is negatively correlated with salinity and hardness, however the toxicity of inorganic mercury increases with salinity due to enhanced solubility (Alpers et al. 2008).

Salinity and hardness also affect sediment redox potential.

Water Residence Time

Water residence time in a tidal wetland also significantly affects contaminant fate and transport. Increased residence time facilitates contaminant adsorption and sequestration in sediments and biota, and in low turbidity environments can also expedite UV degradation (Gregoire et al. 2009). Nevertheless higher residence times may lead to greater temporal exposures of contaminants to resident fish species, due to less frequent tidal dilution. Decreased residence time, on the other hand will increase transport of contaminants off the wetland.

Link to Toxicant DREIRIP Models

Tier 3: Contaminant Responses

Contaminants ~~undergo~~ degradation, transformation, speciation and loss ~~in a system~~ depend~~ing~~ on site attributes (detailed above) and the contaminant's physical-chemical properties.

Degradation

Contaminant degradation in the aquatic environment occurs by microbial activity, hydrolysis, and photolysis, among other mechanisms. Microorganisms (microbes), including archaea, bacteria, and fungi, mediate most degradation processes in the environment. Microbial ~~degradation favors transformations that are energetically favorable, and~~ transformations occur via enzymes that act as catalysts~~ for a given reaction.~~ Photolysis occurs from ~~the energy from~~ sunlight disrupting chemical stability. Direct photodegradation can only occur in chemicals that are capable of absorbing the wavelengths of light energy, such as the double bonds between carbon atoms (e.g. alkenes or aromatic rings) (Hemond and Fechner 2014). Hydrolysis occurs by interactions of chemicals with water molecules, and is often catalyzed by free H^+ or OH^- (thus sensitive to pH). Alkylhalides and esters are two chemical classes that are likely to undergo hydrolysis (Hemond and Fechner 2014).

Most degradation pathways reduce the toxicity of contaminants, however occasionally initial degradation products may be more toxic than the original parent compound. Fipronil, a commonly applied insecticide for agricultural and household uses, degrades in the environmental to the more toxic forms, fipronil sulfide and fipronil sulfone (Schlenck et al. 2001, Weston and Lydy 2014). Organophosphate pesticides also degrade to their more toxic "oxon" form, which is the product of phase I cytochrome P450 metabolism in vertebrate species. The final degradation end products in oxic conditions are generally non-toxic, commonly ~~including~~ carbon dioxide, water, and mineral salts.

Transformation/Speciation

Transformation and speciation affects the solubility, sorption, and volatility of contaminants (Werner 2008). Parameters that affect these processes include temperature, organic carbon concentrations, salinity, hardness, and pH, as discussed previously (see Site Attributes). Transformation and speciation may result in chemical forms that are more or less toxic than the parent compound. For example, speciation causes the formation of ~~the~~ bioavailable and toxic monomethylmercury from metallic mercury in anoxic conditions (Alpers et al., 2008). On the other hand, organic compounds chelate metals in solution rendering them less bioavailable and less toxic. Compounds containing toxic inorganic elements may be transformed from one chemical species to another, however these environmental reactions will not destroy the inorganic elements itself
(Hemond and Fechner 2014).

Loss

Besides loss due to the mechanisms described above - degradation, transformation and speciation - loss of a contaminant can occur due to volatilization, storage in sediments or export from the Delta. Volatilization of a contaminant is significant in all surface waters, and is primarily affected by a chemicals' water solubility (affinity to water molecules) and size (molecular weight). Dissolved ionic molecules (e.g. dissol~~ve~~d NH_4^+) are not volatile because of their strong attraction to polar water molecules, whereasile small hydrophobic molecules such as in gasoline, readily volatilize (Weiner 2012). Hydrophobic contaminants can also become buried in sediment over time rendering them no longer bioavailable. An effective environmental contaminant mitigation strategy employs burying contaminants with gravel or other material to reduce or eliminate the bioavailability of the toxic material, where removal is too costly or unwieldy. Contaminants can also be exported from a site via sediment or water column transfer, as detailed in the previous Contaminant Loading section herein.

Tier 4: Biological Responses

The biological effect of a toxicant (i.e. contaminant) is a function of the concentration of the active form at the target site of action, as well as its binding affinity and efficacy. The internal concentration of a toxicant at a biological target site is primarily driven by toxicokinetics, including chemical absorption, distribution, metabolism and elimination (ADME). The binding affinity and efficacy of a toxicant at a target site is controlled by the biological environment and the physicochemical characteristics of the toxicant and binding ligand(s), described by toxicodynamics. Escher et al. (2001) provide a useful conceptual model illustrating the relationship between total, external, and internal effect concentrations of a toxicant, and the distribution and effects at different target sites within an organism (Figure 1). While the toxic effects of contaminants are usually considered individually, they always occur in mixtures in the environment and their cumulative impact can cause deleterious effects at low, even trace individual chemical concentrations (Hasenbein et al. 2015).

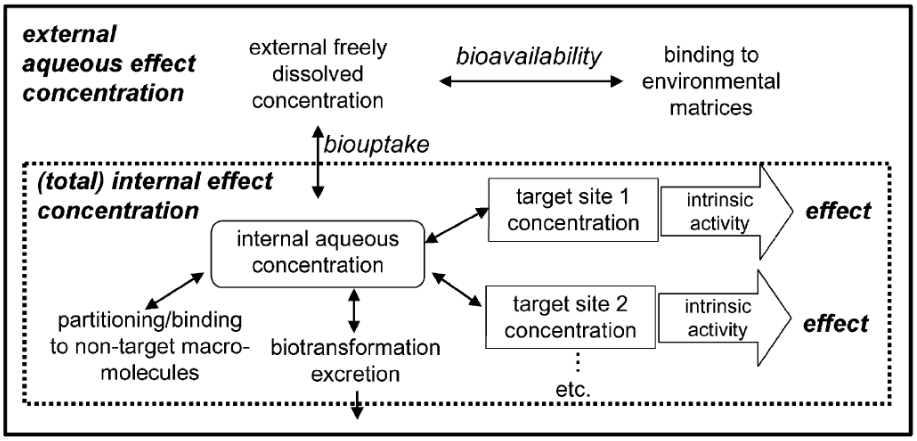

Figure 1. Relationship between ~~today,~~ external, and internal effect concentrations on one or more target sites (Escher et al. 2001).

[Toxicokinetics]

ADME

Exposure signifies the degree of contact of an organism with a contaminant. This takes into account the spatial and temporal distribution of a chemical in the environment (addressed in the previous sections), and its subsequent disposition in an organism dependent on processes of absorption, distribution, metabolism and elimination. Mechanisms of exposure to aquatic species include via water, sediment, and food, and exposures can occur simultaneously depending on an individual's life stage, life history, trophic level, and feeding strategy (Werner et al 2008). Contaminants that occur at high concentrations in the environment may not result in toxicity due to either poor tissue absorption or efficient metabolism or elimination. As such, the disposition of toxicants contributes to the concept that "the dose makes the poison" (Klaassen 2008). Some contaminants, such as organophosphate and carbamate insecticides, are initially transformed into toxic metabolites during the initial phase of metabolism (Phase I). Enzymes drive all metabolic transformations and many elimination processes, and are dependent on environmental temperature in ectothermic organisms like fish and invertebrates (Werner et al 2008).

Link to Toxicant DREIRIP Models

Bioaccumulation

Bioaccumulation occurs when the rate of toxicant elimination in an organism is less than the rate of absorption. The physiological distribution of a chemical within an organism is a major factor that affects the rate of elimination. Toxicants that readily distribute or redistribute to fatty tissues or bone become trapped (precluding elimination) and accumulate over the duration of exposure

(Klaassen 2008). Refer to the DREIRIP Chemical Stressors Conceptual Model (Werner 2008) for specific information on models describing the partitioning of chemicals into living tissues.

Though toxicants that are sequestered in the fatty tissues or bones of an organism are rendered innocuous, they can cause toxicity as soon as they are released back into the blood stream (or body cavity of an invertebrate), either by internal redistribution (e.g. via fat metabolism) or consumption by a higher trophic level species. Trophic transfer of bioaccumulative toxicants can result in biomagnification up the food chain. Bioaccumulation and biomagnification of contaminants often overlap, and it is difficult to distinguish which mechanism is involved when tissue concentrations are detected in field organisms. Because bioaccumulative contaminants are generally hydrophobic, they are often most concentrated in field sediments. As such, biomaginfication may be particularly important for fish species that prey on benthic species, which readily accumulate sediment bound hydrophobic contaminants via food and dermal exposure. In this way, benthic invertebrates in a contaminated tidal wetland may act like “pesticide pills” to higher trophic level species, such as salmon and smelt (Weston and Lydy, in prep.).

Tissue concentrations of chemicals that bioaccumulate, such as mercury, selenium, and PCBs, are good indicators of environmental exposure and biological effects, though a number of toxic chemicals, such as current use pesticides, are rapidly metabolized and tissue concentrations are not prognostic. Tissue concentration or body burden increases with the size and weight of an organism, likely a factor of age and/or trophic level. Older individuals have a longer period to accumulate toxicants, and higher trophic level species tend to accumulate higher levels of contaminants due to biomagnification up the food chain (caused by higher trophic levels consuming smaller individuals with high body burdens of contaminants). Bioaccumulation can also depend on the sex of an organism due to differences in lipid content and the transfer of toxicants to reproductive cells and offspring during spawning and birth (Werner 2008). Other factors that can affect bioaccumulation include the physiological condition of an organism, food source, food quality, and prey availability (Werner 2008).

[Toxicodynamics]

Lethal and Acute Effects

Lethal and acute effects from contaminants on a variety of fish and invertebrates have been observed in water samples collected from tributaries to the Delta (Werner et al. 2010). Most lethal results are from laboratory tests conducted on invertebrate or algae species cultured in the lab, however some field exposures have also resulted in acute affects (e.g., Brander et al. 2009, Weston et al. 2012). Sensitivity differences between lab and field organisms, even of the same species, indicate that laboratory results are not representative of ecological effects (Major et al. 2013, Weston et al. 2013), but laboratory toxicity test results do still provide useful information on the environmental concentrations of toxicants. There is little empirical evidence for lethal impacts of contaminants on fish species in the Delta over the past two decades, however impacts on prey species could lead to contaminant-mediated food limitation for sensitive fish species like Delta Smelt (MAST report).

Sublethal effects and Chronic Effects

Though pesticides and other anthropogenic contaminants do not commonly occur in the Delta at concentrations that are acutely toxic to fish, sublethal effects have been well documented (MAST report). Of the currently applied pesticides, herbicides and fungicides are most commonly detected in Delta water and sediment, and are also found in fish tissue (Orlando et al. 2013, Smalling et al. 2013). Herbicides and fungicides are known to primarily affect algae and phytoplankton species, though some also affect endocrine and reproductive functions in fish (Ankley et al. 2005, Xie et al. 2005). Though insecticides are less frequently detected in surface waters, effects can occur at very low concentrations, even below analytical detection limits (Hasenbein et al. 2015a). Fecundity, swimming performance and growth are affected in both fish and invertebrate species by a wide range of current use pesticides (Connon et al. 2009, Hasenbein et al. 2015, Scholz et al. 2000). Exposure to legacy contaminants also causes sublethal effects in invertebrates and fish.

Mixture Effects

In the field a~~A~~ll ~~environmental exposures to~~ toxicants occur as mixtures. Effects of toxicants in combination are most commonly additive or close to additive (Backhaus and Faust 2012), but synergistic and antagonistic effects also occur. Effects of contaminants that have the same or similar target sites of toxicity (e.g. neurotoxins like many insecticides and some metals) are most commonly additive or close to additive on both lethal and sublethal levels, while effects of toxicants with different target sites of activity can be less than additive (Altenburger et al. 2000, Backhaus et al. 2000). Some contaminants that are not toxic on their own, enhance the toxicity of other contaminants and are thus synergistic. This is true for a variety of insecticide and herbicide additives. In some cases environmental conditions ~~can~~ enhance the toxicity of contaminants, as in the case with UV light and PAHs, and temperature and pyrethroid pesticides.

Tier 5: Ecological Responses

The toxic effects of contaminants are not only ~~can be~~ deleterious to organisms~~ fitness and reproductive success~~, but also ~~as well as~~ disrupt community structure and reduce species richness and abundance. Ecological effects of c~~C~~ontaminants may be ~~are thought to be~~ among the primary stressors causing the Delta Smelt decline (Johnson et al. 2010).

Adaptation

Environmental contaminants can lead to adaptation by resident species, causing them to be resistant to contaminant effects. This ~~has been shown to~~ occurs across a wide range of species and regions. Among wild *H. azteca* populations in the Delta, Weston et al. (2013) found specific genetic point mutations that reduced the sensitivity to pyrethroid pesticides. These genetic point mutations, and thus resistant populations, were especially found in areas highly contaminated with pyrethroid pesticides. Adaptation on a molecular and genetic level can enhance protective mechanisms involved in either the toxicokinetics or toxicodynamics of a contaminant. Toxicokinetic mechanisms include rates or processes of absorption, biotransformation, and excretion of chemicals, while toxicodynamic mechanisms involve modifying specific molecular target sites to reduce sensitivity (Kennedy and Tierney 2012). Such genetic changes, however, may reduce genetic and biological diversity, and thereby alter which m~~ay have consequences on the food~~ web (Weston et al. 2013). The presence of such resistance and genetic mutations in other delta species, such as the delta smelt, in response to contaminant exposures has not been investigated (MAST report).

Distribution

Contaminants can affect species distribution via stress that leads to species-specific mortality or relocation. Lethal effects of toxicants have been discussed previously (see Biological Responses). Environmental conditions, such as temperature, salinity and food availability drive the distribution of species, as can the presence of contaminants. Avoidance of copper and other metals by fish species, such as salmon, has been well documented in the literature, however this is not the case for all species or toxicant exposures (Ward et al. 2013). While relocation is a useful means to temporarily avoid exposure to a stressor, this strategy is not useful if higher quality habitat is not accessible.

Richness and Abundance

Native species diversity and abundance is under stress from a variety of anthropogenic effects on the Delta (Kimmerer 2002, McKinley 2011, Moyle 2002), and contaminants have a particularly strong affect on species richness in marine habitats (Johnston and Roberts 2009). Though adult fish are fairly resilient to contaminant stressors, possibly due to their high mobility, invertebrates and fish larvae are more sensitive targets and may be significantly impacted (Johnston and Roberts, 2009; McKinley and Johnston, 2010).

Species Composition and Trophic Structure

Contaminant presence can lead to shifts in species composition and trophic structure due to their species-specific toxicities. The loss of sensitive species at a given site or within a community can allow other more resilient species to encroach and settle. Some field studies have shown relatively high species abundance at contaminated sites due to an invasive pro~~pensity of highly~~ invasive ~~and resi~~lient aquatic species (McKinley 2011).

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Interagency Ecological Program

Tidal Wetlands Restoration Monitoring Project Work Team

Wetland evolution:

Processes contributing to trajectory of geomorphic change in restoration sites

Conceptual Model

DRAFT

Prepared by: Rosemary Hartman, Hildie Spautz, Chris Enright, Judy Drexler

Based on the Suisun Marsh conceptual model:

Siegel, S., C. Toms, Gillenwater, D., Enright, C. (2010). Tidal Marsh. Suisun Marsh Tidal Marsh and Aquatic Habitats Conceptual Model (draft), Suisun Marsh Habitat Management, Restoration and Preservation Plan

PREFACE

This Conceptual Model is part of a suite of conceptual models which is designed to guide monitoring of restoration sites throughout the San Francisco Estuary (SFE), but particularly within the Sacramento-San Joaquin Delta (Delta) and Suisun Marsh. The conceptual models have been developed based on the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) models, and are designed to aid in the identification and evaluation of monitoring metrics for tidal wetlands restoration projects. These models are designed to describe our understanding of how tidal restoration may benefit at-risk fish species.

Contents

Introduction –

The General Tidal Wetland model in this suite of models describes relationships between major wetland structures and processes benefitting salmon and smelt. This model focuses on the process of change to the physical structure of the wetland. It falls within tier 3: Physical Site Attributes in the General Model, and it describes the processes of sedimentation, organic matter accumulation, erosion, and vertical accretion that change the habitat landscape fish encounter.

Wetlands are dynamic systems. Depending on elevation, tidal regime, and a suite of other factors, wetlands may be composed of mosaics of vegetated tidal marsh plain, shallow intertidal pannes and ponds, tidal channels, tidal mudflats, subtidal channels, and upland transition zones. The distribution and function of these components may change through time. Sediment from riverine and bay sources combine with vegetation on-site to form peat and produce vertical accretion. This process is countered by erosion, subsidence, and decomposition of peat to reduce vertical elevation. Differentials in accretion and erosion form marsh channels and produce topographic heterogeneity across the wetland. Feedback between present surface elevation and geomorphic processes lead to channel development, pannes, mudflats, and broad vegetated plains. These structures provide a variety of habitat types for resident fish and wildlife.

Tidal wetland restoration often aims to restore natural processes of marsh evolution to areas where they have been interrupted by human-made levees that exclude or regulate tidal forces. Prior to restoration, these sites may have been drained for agriculture (primarily in the Delta) or have had managed flooding schedules to maximize habitat and food conditions for waterfowl (primarily in Suisun Marsh; see "Baseline conditions"). If restoration actions lead to appropriate residence times, sediment inputs, and surface elevations, the wetland will increase in elevation and form complex tidal channels over time due to inherent feedback mechanisms (Fagherazzi et al. 2012). In general, deeper areas favor sediment deposition because they are inundated longer, whereas shallower areas increase in elevation more slowly (See Wetland Elevation section below). Growth of emergent vegetation in the marsh is highest at intermediate inundation zones. Vegetation aids in vertical accretion both through production of organic matter and trapping suspended sediment (Kirwan and Megonigal 2013). These processes are critical not only to develop natural wetland processes within subsided restoration sites, but also to maintain intertidal zones in natural and mature restored wetlands in the face of sea level rise (Kirwan and Megonigal 2013, Schile et al. 2014) (See Sea Level Rise section below).

Formation of heterogeneous wetland complexes provides greater habitat heterogeneity with thermal and predation refugia for native fish species (Rozas and Zimmerman 2000, Moyle et al. 2010, Enright et al. 2013, Ferrari et al. 2014). Habitat heterogeneity also allows for mixed residence times across the wetland to increase primary productivity (Levin et al. 2001, Holland et al. 2004).

The process of vertical accretion is particularly relevant to tidal wetland restoration sites because most leveed islands in the region are significantly subsided from groundwater withdrawal and/or desiccation. To return many such areas to intertidal wetlands, they must gain elevation either through natural processes or through artificial import of sediment. Natural sediment deposition and peat accumulation is considered preferable due to its lower cost and lower concentration of contaminants than dredge spoils (Schoellhamer et al 2012).

The previous DRERIP conceptual model for tidal marshes (Kneib et al. 2008), described the processes and structure of a “mature” marsh and did not explicitly identify the processes involved in formation of tidal wetlands after restoration action. This Wetland Evolution model, in conjunction with the General Tidal Wetlands model, aims to fill that gap. It draws heavily on conceptual models of marsh structure and function in the draft Suisun Marsh models (Siegel et al. 2010), updated to reflect more current understanding of the processes involved, and to include characteristics of freshwater tidal wetlands of the Delta. It also draws on many of the processes described in the DRERIP sediment model (Schoellhamer 2012).

Model Structure

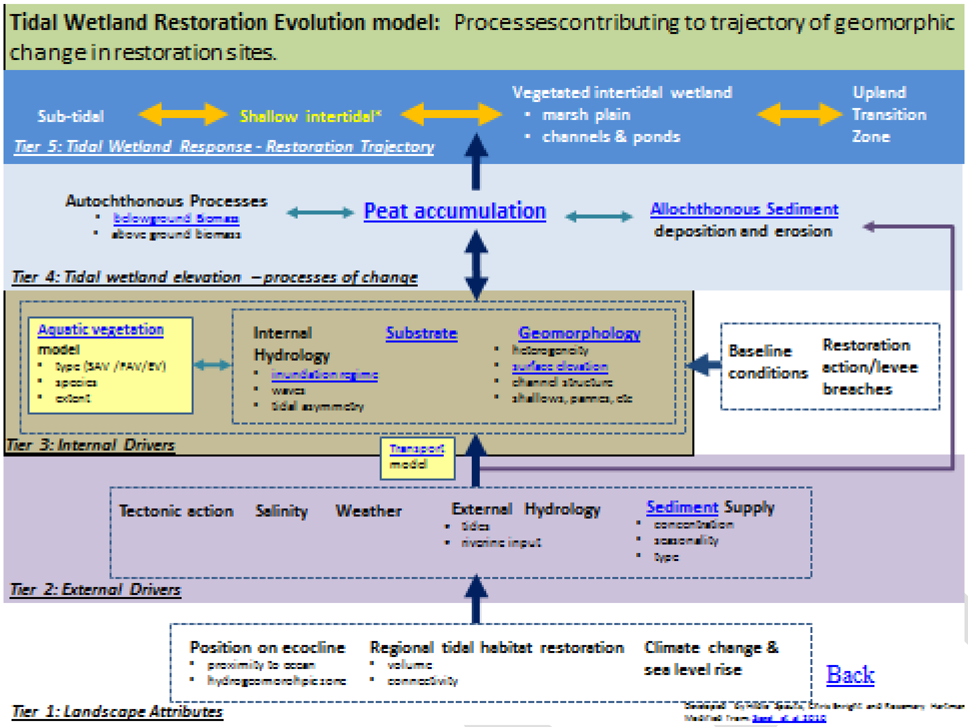

Figure 1. Model diagram

The model diagram follows the General Tidal Wetland Model and other PWT models in basic structure. Groups of attributes surrounded with dotted lines interact strongly. Arrows represent specific, describable linkages or relationships between attributes, groups of attributes, and tiers. Thick arrows moving between tiers that do not connect to specific factors represent simplified relationships between all the attributes in one tier with those in another. The strength of the relationships within and between tiers is variable, and these relationships are described in more detail in the text. Arrow and text color is used to aid in following arrows and does not convey meaning. Yellow boxes are links to sub-models which provide greater elaboration on the specifics of particularly important processes, structures, or species. This is a process-based model rather than a structural model like the General Tidal Wetland Model, so it is organized differently and describes “drivers” rather than “attributes” at tiers 2 and 3.

For the Wetland Evolution model (Figure 1), there are four tiers: 1) Landscape Attributes, factors acting on the scale of the entire San Francisco Bay and Central Valley ecoregions, 2) External drivers, which act at the local scale of operational landscape units (the area directly adjacent to the focal wetland), 3) Internal or “intermediate drivers”, including baseline conditions, that occur within the focal wetland and determine how the drivers act on the particular site, 4) Wetland elevation – Processes of Change are factors contributing to conversion between types of habitat, chiefly organic biomass processes and allochthonous sediment processes and 5) Restoration Site Trajectory – the process of habitat conversion, through elevation change, from subtidal to unvegetated intertidal, to vegetated intertidal, and to upland habitat, or the reverse trajectory.

The output of this model is the matrix of habitat types resulting from the process of restoration site evolution. At any given point in time, there will be a particular ratio of habitat types, but as the marsh evolves this will shift. It may shift towards vegetated intertidal/upland if vertical accretion processes outpace erosional/decompositional processes, or it may shift towards shallow intertidal/subtidal if erosional forces dominate.

A more detailed discussion of sub-models describing processes within this model can be found in the Suisun Marsh models (Siegal et al 2010).

Body of model

Tier 1: Landscape Attributes

Position on ecocline -

In this set of conceptual models, we have defined two “ecoclines.” First, a longitudinal ecological gradient from the Golden Gate to the headwaters of the rivers . Secondly, a lateral gradient from the center of the river channel to the furthest upland zone perpendicular to the first ecocline Thus, position on the ecocline controls the “Hydrogeomorphic zone” (sensu Drexler 2011), the salinity, sediment availability, and the amount of hydrodynamic energy (tidal range and riverine influence) in a given wetland, all of which will affect the rate and direction of wetland evolution.

Proximity to Ocean –

The relative strength of riverine versus tidal flow determines the location of the estuarine-riverine boundary for the entire estuary (Wright and Schoellhamer 2005). The position of the wetland along the longitudinal ecocline influences both hydrodynamics and water quality at the site. The most obvious effect of oceanic influence is the salinity gradient, which influences aquatic community species composition, vegetation species composition and growth rates, rate of organic matter accumulation, sediment flocculation, and other water quality parameters (see Salinity below). Position along the longitudinal ecocline will also control whether the internal hydrology of the site is dominated by tidal action or riverine flow; sites closer to the ocean will generally have a larger tidal range than those further inland. Colonization by marine organisms will also be higher down Estuary, since the presence of marine propagules increases with proximity to the ocean. The proximity to the ocean also determines the site’s position relative to the estuarine turbidity maximum, where suspended sediment concentrations are greatest. Site closer to the turbidity maximum will have higher turbidity. In some systems this is where rates of sediment deposition are also greatest (Butzeck et al. 2014), though connectivity and hydrogeomorphic zone (see below) may be more important than distance to the ocean in the SFE.

Hydrogeomorphic Zone (Distance to and identity of major river channels)

In the Delta, which river is closest to the restoration site will determine much of sediment loading on the site. The Sacramento River and Yolo Bypass are major sources of both water and sediment to the Delta, providing about 85% of the water and sediment supply (Shoelhammer et al 2012, See Riverine Flow below), so restoration projects closer to the Sacramento may increase in elevation faster than those close to the San Joaquin (Reed 2002, Wright and Schoellhamer 2005). However, the lateral ecocline may be more important than river identity. If a restoration site is located along the main channel instead of a slow-flowing tributary, there will be a higher sediment supply even along the San Joaquin (Drexler 2011). Speed of elevation increase may also depend on amount of sediment available and degree to which the site is subsided (See Sediment Supply and Baseline Conditions below) (Drexler et al 2009, Mount and Twiss 2005).

Regional tidal restoration

Volume open to tides

In the San Francisco Estuary, the tidal prism (change in volume from high tide to low tide), is restricted by geomorphology. The tidal prism entering the Estuary spreads out across the area of the Bay and the Delta. Tidal wetland restoration will increase the area of the Estuary, meaning the tidal prism will spread over a greater area. This will reduce the tidal range at any point in the system, and reduce the total tidal energy (Dongeren and Vriend 1994).

Preliminary modeling estimates as much as a two-foot reduction in tidal range over the long term with large-scale restoration (65,000 acres), though in most places it will be less than one foot (BDCP 2013). However, the design of restoration sites, location of restoration sites, amount of vertical accretion and total amount of area restored will affect the degree to which tidal range is changed at any given point (BDCP 2013). Tidal stage should be monitored within restoration sites to track potential changes in tidal range, but the extent to which this will be a problem remains unclear.

Connectivity – Link to transport model

The degree to which sediment can enter the wetland will depend on attributes of the surrounding landscape that may facilitate or block entry to the wetland (see Transport model for more info). Wetland restoration sites may act as sediment sinks, not only mediating some of the effects of sea level rise discussed above, but also potentially reducing the supply of sediment to restoration sites downstream (Williams 2001, McKee et al. 2006). Simultaneously opening up several adjacent, highly-connected tidal wetlands may reduce the amount of sediment available to individual restoring sites. Large, fast-moving channels adjacent to the wetland will have the capacity to transport more sediment than smaller, slow-moving sloughs. Large beds of aquatic vegetation (particularly *Egeria densa*) in surrounding channels will reduce water velocity and trap sediment (Wilcox et al. 1999; Jones et al. 2012), reducing the supply of sediment to the target wetlands (see Vegetation below).

Climate change and sea level rise

Sea-level rise is an on-going process, progressing at 1.38 ± 0.64 mm yr^{-1} along the California coast south of Cape Mendocino over the past 60-100 years (NRC 2012, Chapter 4). However, the rate of future sea level rise and the ability of tidal wetlands to maintain their relative elevations in the tidal frame over the next 100 years has been a topic of much debate (Kirwan and Mudd 2012). Historically, the feedback between relative surface elevation and vertical accretion (see Peat Accumulation section below) has prevented wetlands from being drowned during historic periods of sea level rise. Currently, in much of the Delta, diking and draining of wetlands has caused extensive subsidence, with much of the former wetland area many meters below sea level (See Baseline Conditions below). Given projections for rates of sea level rise and potential failure of levees, more of the Delta may be comprised of flooded islands in the future (Mount and Twiss 2005). Vertical accretion may not be able to restore elevations to intertidal levels, or they may even erode if water velocities are too fast (Marani et al. 2011).

Without incorporating the dynamic nature of vertical accretion in marshes into the models, an estimated 50% of remaining intertidal wetlands in the SFE would be below mean sea level by 2100 (Knowles 2010). Even with vertical accretion processes taken into account, there has been wide speculation over whether vegetated intertidal marshes in the SFE can keep pace with sea level rise, with numerous models that incorporate sediment deposition and organic matter accumulation in different ways (Orr et al 2003, Mariotti and Fagherazzi 2010, Stralberg et al 2011, Schile et al 2014,). Rate of sediment import into the system is one of the key uncertainties driving differences in model outcomes. Lower sediment loads may mean vegetated intertidal marshes will be drowned by sea level rise, whereas higher sediment loads may mean marshes will be sustainable at least until 2050 (Schile et al 2014). See Sediment Section below and DRERIP Sediment model (Schoellhamer et al 2012) for more information on sediment flux in tidal wetlands.

Rate of organic matter accumulation in response to sea level rise is also uncertain. Different communities of plants have different tolerances to salinity and waterlogging, so communities may change as water depth and salinity change with sea level (Parker et al. 2012). Temperature is also expected to rise, favoring plants with C4 phososynthesis; however CO_2 levels are also expected to rise, favoring plants with C3 photosynthesis (Zedler 2012). The combination of changes to salinity, temperature, and CO_2 may result in strikingly different vegetation communities with different rates of above and below-ground biomass production (Reed 1995). These critical differences in vegetation reactions to sea level rise explain much of the differences in model predictions of wetland responses (Schile et al 2014; see Vegetation section below). However, several marshes in the Estuary have been found to be at upper intertidal elevations despite the lack of imported sediments recently and over several years of monitoring (Culberson et al. 2004).

Tier 2: External Drivers – Energy inputs from the surrounding environment

Weather

Both short-term weather patterns and longer-term climatic cycles are major influences on wetland development, chiefly through their effects on wind, precipitation, and temperature. However, weather conditions can vary across the Estuary, so regional weather patterns will interact with position on the ecoclines to influence local conditions. California has a Mediterranean climate, with hot, dry summers and cool, wet winters. This high intra-annual variability in weather is accompanied by inter-annual variability caused by El Niño Southern Oscillation (ENSO), shifts in the jet stream, ocean upwellings, and events such as atmospheric rivers. See Riverine Flow section below for more on how climatic variability translates to hydrodynamic variability.

Wind

High winds, common in the Delta and Suisun Marsh, produce waves. The height and energy of these waves will depend on fetch (uninterrupted distance of open water), which is often low in vegetated marsh habitat, but high in adjacent open water bays. Wave energy re-suspends sediment from the bottom, increasing turbidity and affecting net sediment deposition. High wave action can also erode levees and can prevent plants from establishing. (see Waves below)

Rainfall -

Rainfall, snowmelt and reservoir operations control riverine inputs to the system. Local rainfall provides a small amount of local input of water, sediments and contaminants via runoff from the surrounding land (see contaminants model and transport model for more detail). See Riverine Flow section below for changes in the hydrograph over the course of the year in response to changes in rainfall.

Temperature -

Water temperature is determined chiefly by local air temperature, which in turn is affected by position on the ecocline. Temperatures and seasonal changes in temperature are generally higher in the South Delta than the North Delta or Suisun Marsh (Kimmerer 2004). Because both water and air temperature are higher during the summer when day length is longer, most plant production and decomposition occurs during the summer. This means processes important to vertical accretion change from being sediment-driven to being organic matter driven over the course of the year, with most inorganic deposition occurring with the peak in the hydrograph between March and August, and most organic matter accumulation occurring between June and August (Reed 2002). These seasonal peaks are likely to shift with changes in temperature and snowpack under climate change (Zedler 2012).

As the topographic heterogeneity of the site changes during the process of wetland evolution, temperature and other water quality parameters will change as well. Tidal wetlands may become heat sinks during warm months and provide thermal refugia for fish due to higher temperature variability than managed wetlands (Enright et al. 2013).

Hydrology/hydrodynamics

Tides –

Tides determine Inundation Regime and Tidal Asymmetry and are determined by Position on both Ecoclines. The SFE is mesotidal, with tidal range generally <2m. The tidal range in any given wetland will be determined by its position on the ecoclines, the geomorphology of the site and its connection with the surrounding channels. The tide range changes monthly (spring-neap cycle), seasonally (king tides), and spatially (greater tide range closer to the Golden Gate and closer to the main river channel, lower range in the Eastern Delta).

Further upstream, unidirectional riverine forces replace dispersive tidal forces as the dominant source of energy and water movement (Whipple et al. 2012).

Tidal action is responsible for movement of sediment, nutrients and other materials onto and off of the wetland. Flood tides move sediment-laden water from river channels into the wetland. Sediment precipitates out of the water during slack tides, and a certain amount of it is re-suspended and exported on ebb tides (Ganju et al 2004). In the SFE, sediment deposition and erosion are controlled more by the spring-neap (14 day) tidal cycle than the daily (24.8 hr) tidal cycle (Schoellhamer 2000). The periods of still water on the slack tides are relatively brief, so suspended sediment increases leading up to the spring tide and decreases leading up to the neap tide. Tidal action also drives the erosion of tidal channels within the site (Kirwan and Murray 2007) (See channelization, and tidal asymmetry below).

Riverine flow -

Rivers are the major source of both water and sediment transport in the estuary. Most of the sediment entering the Delta is from bank erosion or runoff from the surrounding watershed (Ganju et al 2004). River flow is unidirectional, and fairly constant on a daily basis, but varies greatly on an annual and inter-annual basis. The unimpaired hydrograph typical of California is dominated by higher flows in winter, a spring snowmelt pulse, followed by low flows throughout the dry summer and fall. These seasonal swings can be extreme, with flows through the Delta of 1700 +/- 300 m3/s in the winter versus 540 +/- 40 m3/s in the summer. Inter-annual swings may be just as severe, with flows as low as 230 m3/s in a low year and as high as 2700 m3/s in a high flow year (Jassby and Cloern 2000). Differences in flow are due to differences in precipitation as well as water diversions. The unpredictable nature of riverine flow and the sediment it provides means restoration planners may have difficulty predicting short-term rates of sediment deposition within a given wetland.

Riverine flow and sediment load depends greatly on river identity. The San Joaquin River is snowmelt-dominated, with highest flows in late spring to early summer, while the Sacramento is rainfall dominated, with higher flows in winter. The Sacramento River provides the majority of the water to the Delta (84%), with much less entering through the San Joaquin River (13%) or eastern tributaries (3%) (Jassby and Cloern 2000).

Dams, diversions, and other human modifications of the system have decreased winter and spring flows while increasing summer flows, but intra-annual variability in river flow and associated sediment transport can still be dominant drivers in the system (Brown and Bauer 2010, see Sediment section below). Decreases in riverine flow reduce total sediment loading, but may increase sediment deposition and decrease grain size since water will be flowing more slowly (Wright and Schoellhamer 2005).

Salinity

Site salinity is driven primarily from the interaction between Position on the Longitudinal Ecocline and amount of Riverine Flow (which varies from year to year). Sites closer to the ocean (ie, Suisun Marsh) will have higher salinity than those further inland (the North Delta), but salinity at any given time will also be affected by precipitation and freshwater flow. During the spring, when Delta outflows are higher, salinity across the estuary is lower. Salinity generally increases in the fall, as freshwater flows decrease. The position of the Low Salinity Zone is used as a broad index of the salinity regime of the estuary over short time frames (days, weeks or months); this index, X2, is the distance in km from the Golden Gate to the location where daily average is 2 ppt; thus X2 increases in dry years and dry seasons and tends to decrease at times of higher outflow. Changes in the distribution, and even abundance, of many organisms in the estuary show correlations with X2 (Jassby et al. 1995; Kimmerer 2002). X2 is often used as an index of the longitudinal ecocline.

The dominant driver of vertical accretion changes as a function of salinity; sediment is generally dominant in salt marshes and organic matter is dominant in tidal freshwater marshes (Orr et al 2003, but see Butzeck 2014). The dominant vegetation species change with the salinity gradient, with freshwater species generally accumulating biomass faster than salt water species. Furthermore, the species-independent production of wetland vegetation decreases with increasing salinity (Watson and Byrne 2009, Parker et al 2012). Salinity also decreases the rate of decomposition (Mendelssohn et al. 1999), so while accumulation of organic matter will be slower in salt water than fresh water, breakdown of peat will also be slower.

Sediment Supply –

For a more complete description of sediment dynamics in the SFE, see the DRERIP sediment model (Schoellhamer et al 2007) and its companion paper (Schoellhamer et al 2012).

Concentration -

Higher sediment concentrations lead to faster sediment deposition and vertical accretion. Long-term reductions in sediment concentration over the past 50 years due to the cessation of hydraulic mining, changing land use, armoring of river banks, and building of large, sediment-trapping dams may mean historic rates of sediment loading and marsh accretion may not apply to current wetland restoration sites (Wright and Schoellhamer 2004; Schoellhamer et al, 2013)).

The pre-mining sediment supply of the estuary is unknown, but is estimated to be approximately 1.5 Mt per year, increasing to around 10 Mt per year during the height of the hydraulic mining period (Barnard et al 2013). Current sediment load is estimated between 1 and 4 Mt per year, with only about 10% of that supply depositing on wetlands (Schoellhamer 2005). The amount of sediment trapped in wetlands is expected to increase with tidal wetland restoration, however there is also the potential for upstream wetlands to trap sediment and reduce sediment availability for downstream wetlands (Schoellhamer et al. 2012).

Seasonality and interannual variability

The majority of sediment enters the estuary during the winter and spring storms (Morgan-King and Schoellhamer 2013, see Riverine Flow section above). An estimated 87-99% of total sediment transport is brought in to the Delta during the rainy season, with single storm events often transporting large percentages of yearly or decadal sediment supply (Barnard et al 2013). Water regulation and “leveling out” of the hydrograph has reduced the magnitude of spring floods and could lead to reduction in net sediment flux over the course of the year (Brown and Bauer 2010). In contrast, future increases in the frequency of extreme events due to climate change may increase sediment flux – both sediment import and scour (Zedler 2012).

Type –

Suspended sediment in the SFE is predominantly fine particles (<63 microns in diameter) (Schoellhamer et al. 2012). Coarser materials (sand) drop out of the water column first, and at higher water velocities than fine silts. Therefore, bed material in the large, fast-moving river channels in the Delta is dominated by sand, whereas wetlands with emergent macrophytes that decrease water velocity are dominated by fine silt (Schoellhamer et al 2012).

Tectonic action -

Neotechtonic uplift and subsidence may cause the bedrock beneath the wetland to rise or fall, which may remediate or exacerbate the problems of sea level rise. Seismic action may cause relatively sudden changes in ground level, and may contribute to levee failures that alter regional hydrology and increase the area open to the tides (Mount and Twiss 2005). Rapid changes in elevation caused by earthquakes may be very damaging for wetland communities since they may suddenly become drowned or dessicated due to shifts in elevation (Zedler 2012). Neotectonic subsidence, though sporadic, averages out to approximately ~1 mm yr^{-1} or more, although GPS-measured vertical land motions have a broad range of values (-3.7–0.6 mm yr^{-1}) (NRC 2012, Chapter 4). While California is a naturally active seismic area, incidence of earthquakes may be increased by changes in water management, such as construction of major impoundments or groundwater withdrawals (Gupta 1992), and extraction of petroleum products (Yerkes and Castle 1976).

More gradual, anthropogenic land-surface subsidence may also contribute to decreases in elevation on the landscape level. Subsidence is a chronic problem resulting from groundwater withdrawal, natural gas withdrawal, and/or drainage and reclamation of marshlands for agriculture (Rojstaczer et al. 1991; Deverel and Rojstaczer 1996, Deverel and Leighton 2010). Currently, land-surface subsidence on reclaimed marshlands in the Delta ranges from approximately 0.7 to 3.9 cm yr^{-1} (Deverel and Leighton, 2010). Tidal wetland restoration is expected to remediate subsidence caused by compaction, desiccation, and erosion of dried peat in reclaimed islands, but will not remediate subsidence caused by groundwater withdrawals.

Tier 3: Internal drivers

Geomorphology

Channelization

Formation of marsh channels is one of the key drivers of habitat creation for fish on the marsh plain (see Restoration Trajectory section below), and there is a strong feedback between channel formation and further geomorphic change. Channels form as ebb tides scour sediment from between vegetation beds. Because of this, areas with greater tidal energy are likely to channelize faster than those with weak tides.

Most rapid vertical-accretion dominated tidal channel development occurs at mid-marsh elevations. Lower elevation marshes develop channels more slowly, but allow for greater fish access immediately, whereas high marsh allows for greater vegetation colonization but little tidal channel development (Kirwan and Murray 2007). Channel formation may be either erosional or depositional depending on marsh age and sediment supply (Williams et al. 2002, Hood 2006). As sea level rise continues, further, upward progression of the existing tidal channel network will occur into areas of higher, fringing elevations, if accommodation space for the upward progression of the tidal prism available, and is planned for in restoration designs.

Channels may drive the sediment deposition over the vegetated marsh plain and provide important connectivity between intertidal vegetation and subtidal open water. Sediment deposition tends to be greatest near the channels, decreasing inland over the marsh plain (Temmerman et al 2005a). Emergent vegetation may aid in formation of channels by slowing water. This causes water velocity to increase in unvegetated areas and cause channels to erode (Temmerman 2005b). However, if vegetation colonizes over the entire mud flat before channels can form, emergent vegetation may prevent channelization (Cain 2008). This is especially true of tules in the freshwater marshes of the Delta, and less so in *Spartina* salt marshes of San Pablo and San Francisco Bay, in part because tules colonize at a lower elevation than *Spartina* (See Vegetation section below). If channel morphology changes over the scale of a tidal excursion, as is common in complex, highly branched channel networks, tidal asymmetry will cause greater sediment deposition than erosion, contributing to further vertical accretion (Dronkers 1986).

Relative surface elevation –

Deeper areas favor sediment deposition, while shallower areas increase in elevation more slowly due to inherent feedback mechanisms. When an area is submerged during a flood tide, sediment will fall out of the water column as the water slows at slack tide. Lower elevations remain submerged for longer, increasing the length of time in which sediment can deposit. As sediment accumulates, elevation increases, decreasing time spent submerged and decreasing sedimentation rates (Schile et al 2014). In high marshes, most sediment and material is exchanged via intertidal channels, decreasing the deposition on the high marsh plain. In contrast, low marshes have much greater exchange of sediment through the marsh edge (Temmerman et al 2005). Therefore, the spatial, as well as the temporal, sedimentation pattern is most related to elevation differences and distance from channels.

Surface elevation interacts with vegetation to affect rate of vertical accretion. Growth of emergent vegetation in the marsh is highest at intermediate inundation zones, where it aids in vertical accretion both through organic matter accumulation and by trapping sediment (Kirwan and Megonigal 2013, see Vegetation section below).

Colonization and productivity of vegetation is highly dependent on depth. For example, in an experimental nontidal wetland area in the western Delta (Twitchell Island), a site with shallower (25cm) water was colonized by tules in 2 years, whereas the deeper site (55cm) had spatially heterogeneous depths due to patchy tule colonization and still had open water 9 years later (Miller et al. 2008).

Internal hydrology/hydrodynamics

Inundation regime

Inundation regime is determined by the interaction of surface elevation and tides. Therefore, inundation regime in any particular site is highly dependent on position on the ecoclines, and may change with increasing area of wetlands open to tides.

The inundation regime determines the rate and location of sediment (and other) exchanges between the marsh plain and adjacent subtidal channels. The longer the wetland is inundated, the greater the rate of sediment deposition and elevation increase (Crooks et al. 2002, Butzeck et al 2014). Permanent shallow flooding can allow marsh elevation to increase in managed wetlands to allow for subsidence reversal and eventual tidal restoration (Miller et al. 2008), though processes in these permanently impounded wetlands are significantly different than fully tidal systems.

In high marshes, exchange of materials changes form from spring to neap tide. During neap tides (shallow inundation cycles) water is supplied via the creek system, while during higher inundation cycles more water (and sediment) is supplied via the marsh edge (Temmerman et al 2005). This flow pattern is in accordance with the observed decrease in sedimentation rates with increasing distance from creeks and from the marsh edge (Butzeck et al. 2014). Historically, high sedimentation along channel backs formed natural levees, especially in the north delta (Whipple et al. 2012). This caused highly productive floodplains to form behind the levees, though these wetlands were only connected on spring tides or riverine floods.

Waves

High wave action may re-suspend sediments and limit establishment of vegetation, retarding the process of vertical accretion, sometimes severely (Williams and Orr 2002). Most vegetated intertidal wetlands have relatively short fetches, so wind wave resuspension is not as significant as on tidal shelves; however it may be locally important and result in significant transport of sediment off the wetland site (Dronkers 1986). If vegetated intertidal marshes are adjacent to wide, open-water bays, waves can cause erosion of vegetation (Marani et al. 2011). Waves can also erode levees over time, changing the internal hydrology of the system and increasing area open to tides. On the other hand, sites at the landward end of prevailing winds, as the North-east coast of Grizzly and Honker bays can accumulate sediments suspended by the wind.

Tidal asymmetry -

The maximum tidal velocities on the ebb versus the flood tide onto the wetland may be different if the wetland area is a similar depth as the tide (or shallower), or if the channel geometry changes over the same distance as the tidal excursion (Dronkers 1986, Friedrichs and Aubrey 1988). Tidal asymmetry has important implications in sediment transport because, if the tide enters at high velocity and leaves at low velocity, it will tend to bring a larger amount of sediment into the wetland than it transports off the wetland. This will increase wetland elevation through sediment deposition. When the area of intertidal flats becomes large relative to the total area of the estuary, the asymmetry may become ebb-dominated, encouraging scour of sediment from the flats and keeping the total area of intertidal flats in equilibrium (Dongeren and Vriend 1994). Tidal asymmetries are common in intertidal wetlands, and modeling of the SFE as a whole has shown tidal velocity asymmetries dominate sediment loading on the estuary-wide scale (Elias and Hanson 2013).

Aquatic vegetation (Link to aquatic veg model)

Emergent and aquatic marsh vegetation can increase marsh elevation, both directly and indirectly. Vegetation will contribute directly to the organic matter component of wetland peat through below-ground biomass accumulation (root and rhizome matter) and through leaf litter falling on the marsh plain (Cahoon et al. 2006). In high-energy systems where leaf matter may be flushed away, below-ground biomass production may contribute more than above-ground biomass to total vertical accretion (Culberson 2001). Vegetation will also influence channelization processes through bed stabilization and bank slumping (Kirwan and Murray 2007)

Vegetation also contributes indirectly to wetland elevation by trapping sediment on their stems and by reducing turbulent kinetic energy and water velocity. Reduction in water velocity causes sediments to settle out of the water. Changes in sediment capture and kinetic energy contribute more than organic matter inputs in some systems (Mudd et al. 2010), but organic inputs dominate other areas (Miller et al. 2008). The relative contribution of these two components also changes over the ecoclines. Organic matter accumulation tends to be highest in freshwater areas where overall biomass production is higher than salt water, and where emergent vegetation (particularly tules) grow at lower elevations than the dominant emergent plants in salt water (*Spartina foliosa*). Relative importance of sediment trapping will also increase as concentration of suspended sediments increase, meaning higher-energy sites closer to the high-sediment Sacramento River will have greater inorganic sediment deposition (Drexler 2011).

Due to the sediment-trapping action of plants, sediment deposition is much higher along channel edges than further up the latitudinal ecocline (Temmerman 2005). Submerged aquatic vegetation can trap sediment and block it from entering the wetland altogether. In the Delta, the invasive waterweed *Egeria densa* is slowing water in channels, precipitating sediment out of the water column and potentially blocking it from accreting in surrounding wetlands (Wilcox et al. 1999; Jones et al. 2012).

Baseline Conditions and Restoration Action

Restoration of tidal wetlands may involve varying degrees of hydrologic and geomorphic change. Some restoration projects restoretidal hydrology by simply breaching levees and “letting nature take its course”, expecting the ecosystem to develop along trajectories predicted by ecological theory (Zedler and Callaway 1999). Other projects perform extensive grading of site elevations, excavation of tidal channels, control of invasive plants, and planting of desired vegetation in an attempt to “jump start” the natural process of wetland evolution, or to prevent the site from falling into an alternate stable state (Suding et al. 2004). This conceptual model is geared toward determining important monitoring metrics to inform future restoration, but will not attempt to prescribe certain restoration designs, merely help evaluate the success of the design that has been planned.

When the only active restoration action is breaching a levee, the size and configuration of the breach(s) will greatly influence the internal hydrology, inundation regime, and tidal asymmetry (Monsen et al. 2002). More numerous and/or larger breaches decrease residence time and increase flux, whereas fewer, smaller breaches increase residence time and may increase productivity and sedimentation. Using water control structures or excavations to change the depth of water on the site can affect rates of vegetation colonization, sediment deposition, and channel formation (Cornu and Sadro 2002, Miller et al. 2008).

The effect of different restoration designs will be highly site-specific. In areas with high tidal energy, marsh channels may form naturally and rapidly (Rice et al. 2005), whereas in areas of lower tidal energy (such as the Eastern Delta), vegetation may colonize the entire marsh plain without forming channel networks (Cain 2008). Similarly, in some areas vegetation may colonize rapidly and naturally (Boumans et al. 2002), whereas in other areas desirable vegetation may not establish themselves due to poor soil conditions, high water velocities, or pressure from herbivores (Handa and Jefferies 2000, Brooks et al. 2015). Even when vegetation does establish naturally, planted sites may have different diversity and/or productivity when compared to unplanted sites (De Steven et al. 2006, Mitsch et al. 2012).

Often, despite pre-project modeling and extensive design, restoration sites do not develop along their designed trajectory (Zedler and Callaway 1999). However, choosing monitoring sites based on areas expected to have high rates of change will aid in finding the most responsive monitoring metrics and aid in informing future restoration designs.

Tier 4: Wetland elevation – processes of change.

The elevation of the marsh plain is subject to numerous physical and biological processes that alter marsh elevation relative to the tidal frame. Processes that increase elevation include gradual vertical accretion and episodic neotectonic uplift (Redfield 1972; Drexler 2011; Deverel et al. 2014). Processes that result in reductions in marsh plain elevation include scour, eustatic sea-level rise, neotectonic subsidence, and land-surface subsidence (Redfield 1972; Drexler 2011; NRC 2012; Deverel et al. 2014). In the Delta, all of these processes are at work.

Vertical accretion on the marsh surface consists of both inorganic sedimentation, which is largely an allocthonous process (i.e., originating from the greater watershed), and organic accumulation, which is mostly an *in-situ* (autochthonous) process (Turner et al. 2001; Neubauer 2008; Drexler 2011). The relative importance of organic versus inorganic contributions to vertical accretion changes with position on ecoclines, seasonality, and water depth. The elevation of the marsh plain at a particular time is a function of the *balance* of all the above processes. Marshes are constantly evolving and changing. Their long term sustainability depends on whether or not the processes building up the marsh are robust enough to keep pace with relative sea-level rise (eustatic sea-level rise plus all forms of subsidence) (Swanson et al. 2013).

Allochthonous sediment deposition

Inorganic accumulation is the deposition of suspended sediment from the water column onto the marsh plain. In the Delta, the contribution of inorganic sediment has been shown to vary as a function of the sediment supply from the watershed and river discharge rate, which is largely controlled by climate (Wright and Schoellhamer 2005). Scour (re-suspension and loss of accreted material) can occur continuously during tidal flooding or as a result of intense episodic flooding during storm activity. Inorganic sedimentation is most rapid on young marshes, which are flooded more often and for longer periods, but sedimentation rate declines rapidly and nonlinearly as marsh surface gains elevation relative to the intertidal frame (Pethick 1980, 1981).

Autochthonous processes

Organic accumulation consists mostly of plant litter from tules and other emergent macrophytes that builds up on the marsh surface during and after the growing season. Vegetative biomass is usually a function of rooting zone salinity and water depth (Culberson 2001, Kirwan and Murray 2007). While both emergent and submerged aquatic vegetation may contribute to sediment trapping and organic biomass accumulation, emergent vegetation often has much higher biomass production than submerged vegetation (Miller and Fuji 2010). There is also high inter-annual variability in plant biomass production related to climatic and riverine flow variability (Miller and Fuji 2010).

Tier 5: Restoration trajectory – ecological functions of different habitat types.

Restoration sites ideally contain a complex of subtidal, intertidal, and upland habitat including vegetated and nonvegetated zones (Figure 2). These will form through wetland evolution, and each stage in wetland evolution has particular ecosystem functions associated with it (Zedler and Callaway 2000, Williams and Orr 2002, Cloern et al. 2011, Mitsch et al. 2012).

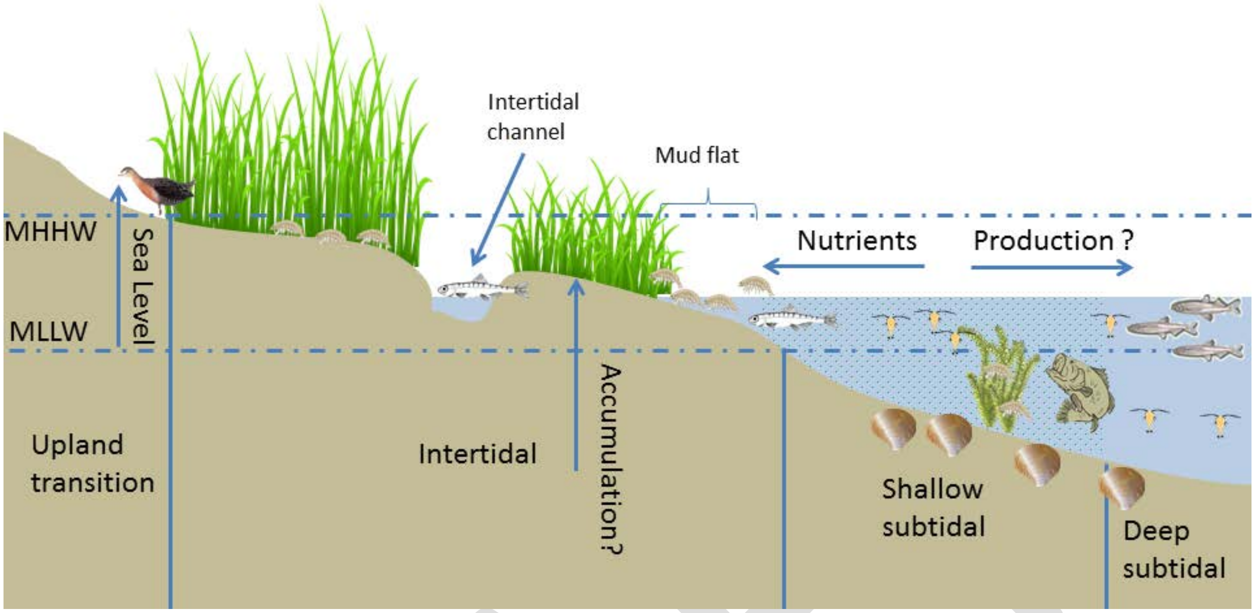

Figure 2. Diagram of major wetland habitat types and functions they are expected to provide.

Sub-tidal

Subtidal areas can be divided into channels, shallow open-water, and deep open water. Open water is the chief habitat for Delta Smelt and Longfin Smelt, as well as a variety of other native fish species (Baxter et al. 2010). Having subtidal open-water habitat adjacent to the highly productive intertidal habitat with a channel network theoretically increases productivity of subtidal habitat due to an influx of detritus and production from the marsh plain. Subtidal areas also provide space for fish foraging in the inter-tidal zone to retreat during low tide. In one study, as much as one-third of the fish population of subtidal channels moved into inter-tidal channels as high tide rose (Desmond et al. 2000). Large expanses of shallow subtidal areas will increase wind fetch and may contribute to sediment resuspension and increased turbidity, providing higher habitat quality and quantity for Delta Smelt (Sommer and Meija 2013).

Deep subtidal habitat may provide connectivity between tidal restoration sites and may provide an influx of nutrients to the intertidal habitat through movement of water and nekton (Allen et al. 2013). Shallow subtidal is expected to have high productivity due to light availability (Lucas et al. 2009). Modeling results suggest that high connectivity between shallow, autotrophic environments with excess productivity may drive secondary production in deep, heterotrophic environments, while nutrient recycling in deep environments contributes to greater productivity in shallow environments (Cloern 2007).

Presence of both native and invasive submerged aquatic vegetation (SAV) may alter the habitat function of subtidal open water (See vegetation model). SAV provides habitat for non-native predatory fish (Grimaldo et al. 2004, Ferrari et al. 2014) as well as invertebrate prey (Howe et al. 2014). It also slows the water and increases sedimentation rates, potentially decreasing sediment available for vertical accretion on the marsh plain (Mudd et al. 2010).

Currently, much of the available land within the Delta and Suisun marsh is subsided, so immediately breaching levees may result in high proportions of subtidal habitat. Historically, there was more vegetated intertidal and less open water habitat than current conditions (Whipple et al. 2012). Therefore, habitat value of subtidal habitat is expected to be greater when connected to intertidal habitat than when isolated from intertidal areas.

Shallow-intertidal open water

Intertidal open water can be categorized as either intertidal channels or intertidal mud flats. Intertidal channels may be particularly important habitat for native fishes. In Suisun marsh, native fishes are more common in smaller sloughs than larger sloughs, whereas non-natives do not show this relationship (Moyle et al. 1986, Meng and Moyle 1994). Given the fact that the SFE is “the most highly invaded estuary in the world” (Cohen and Carlton 1998), small sloughs in intertidal marsh habitat may be critical refuges for native species. In China Camp marsh, adjacent to the San Francisco Bay, a heterogeneous array of large and small channels within the marsh plain maximized fish community richness and food availability for different species and life stages (Visintainer et al. 2006). Engineered marshes often have simplified channel structures, which may reduce nekton access to highly productive vegetated intertidal areas (Simenstad and Cordell 2000), so monitoring formation of natural marsh channels may be important in quantifying benefits to native species.

Intertidal mudflat is also productive habitat, though often neglected in calculations of wetland productivity. Its food web is based largely on benthic algae and infaunal communities (Short et al. 2000). Productivity of benthic algae may be an order of magnitude less than phytoplankton or macrophytes (Cohen et al. 2014), but benthic and epibenthic invertebrates may be a large part of many fish diets (see Food Web Model). Mudflats are not commonly found in the Delta partly due to dredging and rip-rapping of most channels, but they are a regular feature of wetlands in San Francisco Bay and Suisun Bay.

Vegetated intertidal wetland –

Vegetated intertidal areas provide primary productivity through macrophyte detritus, as well as providing physical structure. Emergent macrophytes provide surfaces for epiphytic algal production and associated epiphytic invertebrates. They also provide shade, thermal refugia (see Temperature section above), and structural refuges from predation (Ferrari et al. 2014). Decaying vegetation provides a significant amount of food web support in many estuarine systems (Maier and Simenstad 2009), though its contribution to the SFE is thought to be less than phytoplankton (Sobczak et al. 2005). The contribution of this detritus may be increased with the planned increase in tidal marsh area (see Food Web model for more on detrital and epiphytic inputs to the food chain).

Large invertebrate communities in vegetated intertidal marsh may replace plankton as the main source of fish food in vegetated areas, including Suisun Marsh (Moyle et al. 2014). Fish collected in intertidal channels eat mostly benthic and epibenthic invertebrates, even fish often classified as “planktivores” (Howe et al. 2014). Invertebrate communities return to reference conditions within 6 to 10 years of restoration action, though community composition is heavily tied to vegetation species (Howe et al. 2014). (See Food web model for more info)

Most fish and nekton forage primarily along the marsh edges, with one study finding greatest abundance of most species within three meters of the edge (Peterson and Turner 1994). Fish foraging in intertidal habitat may consume up to 10 times more food in intertidal habitat than in subtidal habitat (Potthoff and Allen 2003). Edge habitat also contributes most to export of productivity from the marsh (Cicchetti and Diaz 2000, Simenstad and Cordell 2000), so restoration sites that include vegetated intertidal wetland subdivided with multiple branching channels will export more productivity than vegetated intertidal areas without channels.

Although not a focus of the current model, vegetated intertidal areas also support a suite of terrestrial vertebrates, invertebrates, and plants. Many of these species interact with the focal aquatic system through the food web (See Food web model for more info).

Upland Transition Zone.

Uplands adjacent to wetlands are important both for the unique community they support, but also as a transition zone allowing space for sea level rise. Vegetated transition zones are critical to tidal marsh resident vertebrate and invertebrate species, many of which use the upper elevations as refuge during high tides. Upland transition zones also provide allochthonous carbon to the marsh. Shading from banks provides thermal refugia. Many wetlands are surrounded by steep levee banks or are on islands, so there is little room for the marsh to move upslope (Orr et al. 2003). One study modeling marsh change with sea level rise found that at high rates of sea level rise many areas may restrict marsh to very small bands of what is now upland habitat if rates of sedimentation are not very high as well (Schile et al. 2014). More gently sloping levees with larger transition zones may increase both wetland and future transition zone habitat if sea level rise is rapid.

Conclusion and Model Application

The process of fish habitat formation post-restoration action will proceed slowly at most sites, and may be difficult to predict. This model highlights the major landscape and state variables that affect wetland evolution and the major response variables that are expected to change on the site as evolution progresses. Mapping the important physical features of the landscape and site setting will likely be done during the planning process before the restoration occurs. Combining this map with the restoration design will let the monitoring team bin the site into habitat types and target areas where they expect to see greatest change in habitat structure. Continuing to monitor these changes (surface elevation, channelization, hydrology and vegetation architecture) over time will allow the monitoring team to quantify the ecological functions of their restoration site. The large amount of restoration currently being planned means there may be opportunity to learn from restoration designs and plan future sites to develop more rapidly into high-quality habitat that is sustainable in the long term.

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Interagency Ecological Program

Tidal Wetlands Restoration Monitoring Project Work Team

Regional Transport Conceptual Model for Tidal Wetland Restoration

DRAFT

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Preface

This Conceptual Model is part of a suite of conceptual models designed to guide monitoring of restoration sites throughout the San Francisco Estuary (SFE), but particularly within the Sacramento-San Joaquin Delta (Delta) and Suisun Marsh. The conceptual models have been developed based on the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) models to aid in the identification and evaluation of monitoring metrics for tidal wetlands restoration projects. These models describe our understanding of how tidal restoration may benefit at-risk fish species.

Contents

Introduction:

Much restoration currently proposed for the San Francisco Estuary (SFE) assumes that restoration sites will export productivity in the form of phytoplankton, zooplankton, and/or detritus to the rest of the SFE (United States Fish and Wildlife Service 2008, National Marine Fisheries Service 2009). In order to test this assumption, we must determine the magnitude of productivity transported over varying distances. This is a complicated process with all aspects of SFE hydrodynamics, geomorphology, and biology playing a role. We have simplified these factors in this conceptual model to clarify transport and mixing processes and the likely fates of materials produced in tidal wetland restoration. Accurate models must incorporate ALL processes and functions, but simple models such as this one allow easy comparison of sites and designs that are likely to produce measurable effects.

To illustrate how this model can be used to describe several types of transport, we will use three examples of materials being transported in the estuary. One example with passive particles, one with active particles, and one with contaminants. :

1) The impetus for developing this model was to estimate the magnitude of transport of productivity from tidal wetlands, so the first example is the transport of phytoplankton from the Prospect Island Restoration site in the Cache Slough Complex to the surrounding channel. This example illustrates transport of passive constituents over a relatively short distance.

2) This same model can be used to think about transport of active particles, such as transport of salmon from Knights Landing to the Dutch Slough restoration site. This example also illustrates the ability to model constituents arriving at a tidal marsh from a great distance away.

3) The Delta is influenced by tides, which often transport materials in non-intuitive ways, so the last example illustrates a case where transport occurs against riverine flows. Transport of contaminants from the Sacramento Waste Water Treatment Plant (SWWTP) to the Lower Yolo Ranch restoration site also illustrates that this model may be useful for predicting potential stressor loads on restoration sites.

Though the material being transported, the source site, the connection, and the target site are different for each of these examples, the processes to be monitored in each case will be similar. These processes are determined by the nature of the material, the structure of the source and target sites, and the nature of the connecting channels. To a great degree, the actual quantity or constituent to be monitored will depend upon the management imperative at hand, such as transport of plankton to enhance pelagic food webs, or reduction of specific contaminants.

Model Structure:

The transport model follows the General Tidal Wetland Model and other PWT models in basic structure. Attributes surrounded with dotted lines interact strongly. Arrows represent specific, describable linkages or relationships between attributes. Thickness of arrows generally describes level of hypothesized importance; however this has not been quantitatively described. Thick arrows moving between tiers that do not connect to specific factors represent relationships with all attributes in the following tier. Arrow and text color only aid in following arrows and has no other implication. Yellow boxes are links to sub-models which provide specifics of particularly important processes, structures, or species.

The transport model is organized differently from the other models in this suite. Instead of narrowing the scale from landscape to local, to mechanistic, the model follows the flow of materials from a source site to a target site. For the Transport model (Figure 1), there are four tiers:

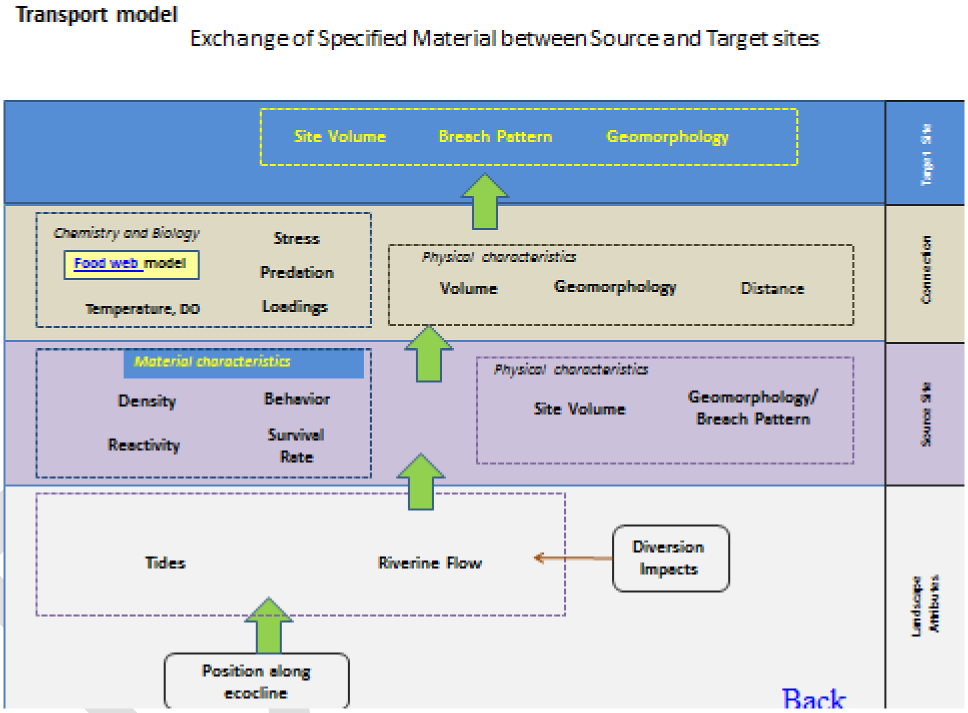
1) Landscape Attributes, factors acting on the entire San Francisco Bay and Central Valley ecoregions,

2) Source Site, which may be the wetland of interest or may be a region providing material to the wetland of interest,

3) Connection, which is the physical nature and minimum length of the path material must follow to reach the target site and

4) Target Site, which may be the wetland of interest or region expected to benefit from production exported from the wetland of interest.

The output of this model is the concentration or loading of material to the target site. This material may be nutrients, productivity, sediment, organisms, or contaminants, depending on the question of interest.



Landscape Attributes

Exchange of material between a restored wetland site and the larger estuary depends greatly upon landscape attributes, including geographic and tidal position within the estuary (called “position along ecocline” here). Particle tracking models of transport within the SFE found release location to be one of the main determinants of particle fate (Kimmerer and Nobriga 2008). Position determines the hydrodynamic forces acting on a particle. For example, tidal flows are generally stronger and more dispersive (mixing) further downstream nearer the San Francisco Bay, and more riverine and advective (transporting) further upstream (Kimmerer 2002). The distance of the source site from the river channel, and the identity of the river may determine how much riverine energy is available to transport the material. For example, since

85% of fresh water into the Delta comes from the Sacramento River (Jassby and Cloern 2000), there will be greater volume and energy available to transport materials downstream from a wetland on the Sacramento versus the San Joaquin, but concentrations of the material will be more dilute. In contrast, lower flows from the San Joaquin River mean that tidal energy will dominate further upstream and dilution of material will be less.

Both local and regional water diversions may also affect transport of materials. The pumping plants in the South Delta cause some river channels to reverse flow even under ebb tides, reversing the net transport of materials in the river and removing a proportion of those materials from the system. Smaller diversions throughout the Delta may also divert materials and reduce the connection between sites (Kimmerer and Nobriga 2008).

Location of the source and target site relative to the ocean will determine salinity. In the case of transport over relatively long distances, organisms or materials may transit between freshwater and saltwater environments. There are functional roles filled by organisms in upstream and fresh areas that are filled by different organisms downstream and they may exhibit different biological and chemical behaviors that affect their transport (Palaima 2012). Even non-living materials may differ in buoyancy or chemical composition in waters of different salinity, temperature, pH, or dissolved oxygen levels (Hwang et al. 2012), which can affect transport. More information on how organisms and chemicals are affected by salinity can be found in the food web and contaminants models respectively.

To return to the examples from the introduction:

1) In considering the transport of phytoplankton from Prospect Island to the surrounding channels (Miner Slough), the landscape influences our consideration in a couple of ways. First, location in the freshwater Delta will determine species composition and productivity of the phytoplankton, which of course will affect the composition and concentration of phyptoplankton exported from the marsh. Also, hydrodynamics are the primary determinant of transport of passive particles. In this example a combination of high flows from the Sacramento River and tidal influence will be the primary drivers in determining transport of phytoplankton from Prospect to the surrounding channels.

2) For salmon transported to Dutch Slough from Knights Landing, the position of Dutch Slough on the San Joaquin means salmon will mostly be transported via the less-used and lower-survival migration routes through Georgiana Slough and the Delta Cross Channel rather than the Sacramento River (Perry et al. 2009). Furthermore, because Dutch Slough is very far downstream (90 miles) from Knights landing, fewer salmon are expected to survive the journey than would reach closer restoration sites, such as Lower Yolo Ranch.

3) The Sacramento Wastewater Treatment Plant discharges into the Sacramento River, which confluxes with the Deepwater Ship Channel around Rio Vista. The Lower Yolo restoration site is located about 20 km upstream the Deepwater Ship Channel branch of the confluence, on a minor

slough. In transporting of contaminants from the SWWTP to Lower Yolo, position of the marsh upstream from the SWWTP, and on a side channel, means that tidal advection will be the major transport mechanism rather than riverine flow. Contaminants transported with the mixing effects from the tidal advection will be diluted compared to if the target site was directly on the Sacramento River. Because of the dispersive nature of tidal transport, contaminants from the main channel can reach restoration sites on sheltered side-channels.

Source Site

The characteristics of the source site will affect how much material is available for transport. The source site may be outside the estuary, for example salmon smolts, sediments, and contaminants move downriver from Vernalis or Knights Landing. Such inputs may be estimated from other research programs but are not generally monitored at an individual restoration site. Source sites within the estuary may be other tidal wetland sites or they may be agricultural discharges, waste-water treatment plants, or urban runoff points.

When the source site is a wetland, the concentration of materials within the source site and the volume of the source site are the most important factors in determining amount available for transport. Measuring or estimating these concentrations is often a primary goal of wetland monitoring. If we assume that a tidally-averaged flux across the wetland site boundary can be calculated (Lehman et al. 2010) (flux across levee breaches in the case of restoration sites), then loading of material is a function of the rate of production within the site, the nature of the site boundary, and the total site volume. Concentrations of some materials (such as phytoplankton) may be roughly estimated with a per-acre/per-depth/per-salinity range approach, as is described for primary production estimates in the food web model. Secondary productivity must be corrected for consumptive processes on the site, which is also described in the food web and fish models. Some sites may have negative export rates, or zero net export rates, because of consumers or hydrodynamic processes at those sites (Dean et al. 2005, Lehman et al. 2010, Herbold et al. 2014). This model applies the term “source site” to those sites with positive net export.

Movement of material from a source site will be affected by the connection of that source site with its surroundings. For restoration projects on previously diked islands, the size, location, and number of levee breaches will be the primary determinants of connectivity. Levee height, flow control mechanisms and seasonal hydrodynamics may also be important in certain situations. For example, the Lower Yolo Ranch site will be much more connected to the surrounding waters and influenced by advective, riverine flows when Yolo Bypass is flooded than when it is not. For some sites in Suisun Marsh, low levee heights may allow for increased connectivity during high tidal elevations, such as at the solstices. Thus we can expect high inter- and intra-annual variability in contributions from various source sites.

Going back to our examples:

1) The design of Prospect Island includes mixed residence times, large breaches, and a large expanse of shallow open water, features that are expected to result in high phytoplankton production (DWR and CDFW 2014). Phytoplankton will be transported off some parts of the site tidally, while other areas will retain water and phytoplankton longer and drain only on higher tides or in storms. Phytoplankton bloom seasonally, so transport will also vary seasonally, with more phytoplankton being transported offsite in the summer than the winter. Behavior is not likely to have much effect on transport, as phytoplankton cannot move against currents, even though some move or even exhibit diurnal vertical migrations.

2) The number of salmon coming from Knights Landing will depend on amount of upstream spawning habitat, reservoir management, and flow of the Sacramento River. Number of salmon leaving Knights Landing can be calculated from the rotary screw trap maintained by CDFW, but transport will depend greatly on behavior. Salmon have the ability to control their position vertically and horizontally, so may migrate against water currents (del Rosario et al. 2013). Salmon are also highly seasonal in their migration, so their position in the estuary will vary greatly with season.

3) Volume of water and concentration of contaminants from the SWWTP will depend on urban water use (Parker et al. 2010). Sacramento is a relatively large urban area when compared to other cities within the Central Valley, so contaminant load may be relatively high compared to other discharges. How the wastewater is treated and discharged will determine the concentration of contaminants. This will change over time as new technologies are incorporated into the treatment process.

Connection

Transport from source site to target site will depend upon the channel geometry, hydrodynamics, and the rates of mortality or chemical transformation in the connection. Factors to consider include the physical characteristics of the connection, such as volume of water in the connection and the geomorphology of the channels, breaches, or sloughs through which the material is transported. For example, deeper channels may reduce photic degradation of chemicals, resulting in greater flux of undegraded chemicals between sites. Deep channels may also enhance predator efficiency on outmigrating smolts, resulting in a reduced flux of smolts between sites. Channels constructed with flows perpendicular to the tidal dominant flow will mix and disperse differently than channels that are constructed to have flows more in-line with existing tidal flows. Resolving the nature of flows before restoration requires detailed knowledge of the geometry and hydrologic configuration of existing channels and to-be-constructed channels at the restoration site. Velocity and patterns of currents in the channels will affect transport times and dispersal of materials of all kinds (Gleichauf et al. 2014). Relative volumes are important to consider when assessing concentration of materials being transported. A highly productive wetland may have little effect on the concentration of exported materials in the nearby river if the river is of much greater volume.

Inherent characteristics of the material being transported, such as density, reactivity, behavior, and mortality can affect how and to where material is transported (Schoellhamer et al. 2012, Hwang et al. 2012, Murphy and Hamilton 2013, Gould and Kimmerer 2010). Some materials are reactive and change character and form as time passes and the environments around them change. Particulate organic matter may break down into smaller pieces or precipitate out of the water column. Poor water quality or high predation may reduce number of organisms transported between sites. For example, periods of low DO in the Stockton Deep Water Ship channel may be a barrier for adult salmon migrating upstream (Lee and Jones-Lee 2003). Predators often congregate at channel mouths and may consume organisms as soon as they leave the restoration site, resulting in no offsite transport. This phenomenon of predator hot spots outside of breaches should be considered in project design if the offsite transport of organisms is an important objective.

Many important processes occur at small spatial and temporal scales (centimeters to meters and seconds to hours) that are expensive and difficult to monitor. Many small scale processes can be estimated as rates of change over larger distances; chemical changes or predation events which may be instantaneous individually, can be represented as an overall rate per km or month. However physical features, aperiodic hydrodynamic events, or weather may be disproportionately important in affecting transport and are often hard to study.

Examples:

1) Transport distance (connection) for phytoplankton from Prospect Island to the surrounding channels is very short, just the length of the levee breach channel. Mortality losses will be low during transport because there is relatively little time for phytoplankton to be consumed or die naturally in the space of the breach.

2) The long stretch of the Sacramento River connecting Knights Landing to Dutch Slough means there is much opportunity for salmon numbers to change in transit between the two sites. Higher river flows may transport salmon more quickly, or they may be transported slower and arrive at a larger size if they spend time foraging in the Yolo Bypass before continuing seaward (Sommer et al. 2005). There are several different routes possible, with different rates of predation mortality, feeding success, growth rates, and ultimately survival (Perry et al. 2009).

3) Degree of mixing and hydrology of the connection, temperature, DO, and pH, UV radiation, and uptake of contaminants by organisms in the channels between the SWWTP and Yolo Ranch may change the concentration of contaminants before they reach the site (Dugdale et al. 2013, Hwang et al. 2012). The Sacramento River and Cache Slough Complex have relatively high volumes in comparison to the SWWTP outfall, so contaminants in the outfall will be greatly diluted in transit. However, because many contaminants may be biologically relevant at the parts-per-billion level, it is still possible that a meaningful, but difficult to measure, amount could be transported to Yolo Ranch.

Target site

The attributes of the target site are similar to those of the Source site. Productivity, concentration, and consumption in the target site do not affect transport, although the food web and fish on the target site may be greatly affected. If the target site is a tidal wetland restoration site, then considering the connectivity of the target site to the connection channel is crucial. Poor connectivity (small levee breaches) or reduced tidal prism may mean that little material in the connection channel makes it into the target site (Brand et al. 2012).

For materials transported to an open water target such as a river channel or subembayment, consideration of volume will be important in determining concentrations. This is particularly true if the target site is far away from the source site. For transport from a wetland to an open water location, the greatest change in concentration will be within one tidal excursion of the wetland site. To truly characterize these effects, monitoring must occur at high frequencies (once per hour), over long periods of time to capture tidal, monthly, and seasonal variability in transport. Effects at these time scales are not easily tracked, measured, or accounted for, so studying them will require a new sampling regime.

Examples:

1) The close proximity of surrounding channels (less than one tidal excursion) from Prospect Island means transport of phytoplankton to the target site of surrounding channels is direct and efficient. Even if this does not lead to net export of production from the island, increases in phytoplankton in the surrounding area on a tidal cycle may increase the availability of production to consumers in the channel (Lehman et al. 2010).

2) Because Dutch Slough restoration site is relatively small and inaccessible compared to the Sacramento River, transport of salmon onto the site will be restricted by the size of the site, and accessibility of the site from levee breaches., So, the total number of salmon entering the marsh restoration site will be likely lower than the number that pass Chipp's Island on the Sacramento River. However, transport of active particles such as fish can be affected by behavior, and good water quality or high food availability may attract salmon to the Dutch Slough restoration site.

3) Water volume at Lower Yolo is relatively small compared to that in the surrounding channels that will transport contaminants from the SWWTP. Therefore, the chief determinant of loadings entering the site from the channel will be the location, currents, and size of the breaches. Once on the site, the contaminants may affect the local organisms or be modified by local chemistry and biology.

Quantitative Models

There are a number of highly detailed and calibrated quantitative models of the hydrodynamics of the estuary, some of which accommodate modeling of particles with different behaviors. For some materials, a simple 2 dimensional model will be adequate, for others 3 dimensions will be needed for even the roughest of approximations. Inorganic particles such as sediments and contaminants can be modeled by coupling particle tracking with material properties, and existing models are robust for calculating rates of movement and rates of change or loss (such as Elias and Hansen 2013, Morgan-King and Schoellhamer 2013, Gleichauf et al. 2014). Models predicting the transport of biological organisms such as plankton and fish must account for growth, mortality, and behavior (such as Kimmerer 2008, Perry et al. 2009, Newman and Brandes 2010, Kimmerer et al. 2014). This qualitative conceptual model identifies what aspects of transport to consider in quantitative modeling of the effects of tidal wetland restoration.

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