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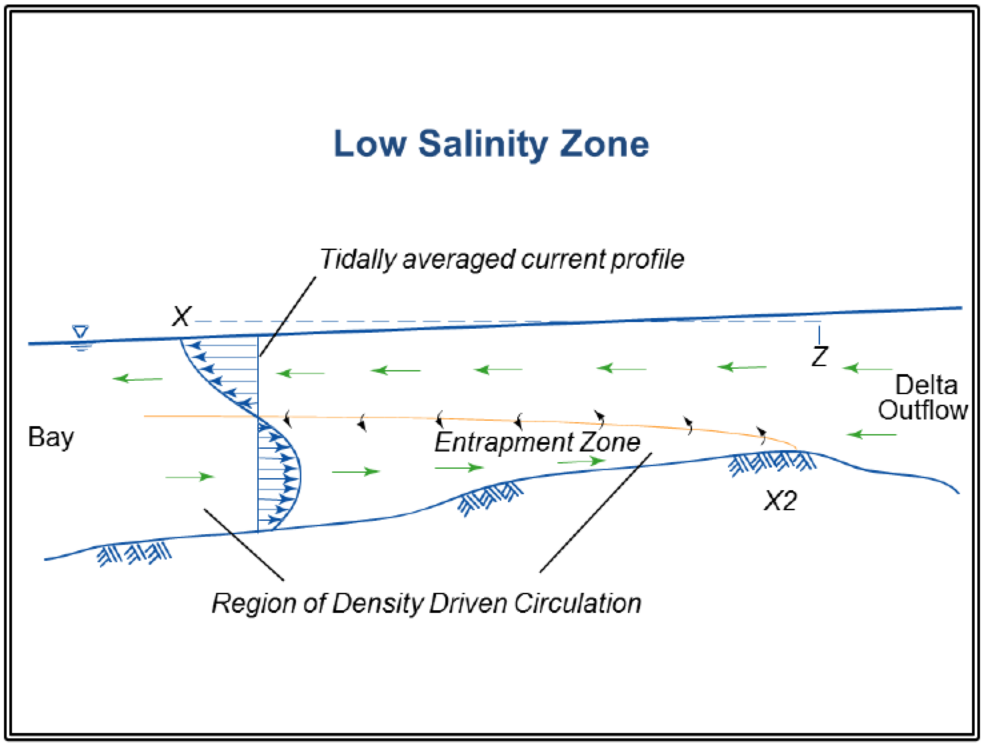

Workshop on Delta Outflows and Related Stressors
Panel Summary Report

Panel

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Delta Stewardship Council
Delta Science Program

Contents

1. Introduction

This report was prepared as part of the State Water Resources Control Board's ("Board") process of developing and implementing updates to the Bay-Delta Plan and flow objectives to protect beneficial uses in the Bay-Delta Watershed. The focus of this report is Delta outflows and related stressors. The report is based upon reading extensive background materials selected by the Delta Science Program as well as materials identified by individual Panel members to be relevant, a two-day public meeting that included a number of presentations and during which public comments were received by the Panel, review of some of the materials provided during and after the meeting, and the Panel's internal discussion and deliberations.

The Board conducted a review of the current 2006 Bay-Delta Plan in 2009 and determined that Delta outflows and other requirements for the protection of fish and wildlife beneficial uses should be considered for revision. "Delta Outflows and Related Stressors" was further identified by the Delta Science Program as one of four topics emerging from a series of Board workshops in 2012 for which additional workshops should be conducted to provide input on the best available scientific information.

Delta outflows and their management have been the subject of extensive scientific and management discussion for decades. A benchmark in this discourse is the report from a series of technical workshops facilitated by Dr. Jerry Schubel (Schubel et al. 1993). Schubel notes in the preface to that report that estuarine standards are required to protect the estuarine ecosystem from "further degradation" until "debate and disagreement over the relative importance of the benefits of low salinity habitat and therefore of flow, on the one hand, and of the liabilities of the physical diversion of a portion of that flow and the associated processes of entrainment of organisms, on the other," can be resolved with a degree of scientific certainty acceptable to the Board. To some extent, this Panel has been asked to revisit whether standards for Delta outflow are still required, and to identify the degree of scientific certainty regarding the importance of Delta outflow to the ecosystem relative to other stressors.

The current requirements for Delta outflows are contained in the Board's 2006 Bay-Delta Plan (SWRCB 2006) and Water Right Decision 1641 (SWRCB 2000). Depending on the water-year type and season, the flow requirements for fish and wildlife beneficial uses are based either on specific Delta outflow requirements or a water quality standard specifying the position of "X2," the horizontal distance in kilometers from the Golden Gate

Bridge up the principal estuarine axis to where the tidally averaged near-bottom salinity[1] is 2 in the Bay-Delta estuary (SWRCB 2010). The Delta outflow requirements are expressed in terms of a Net Delta Outflow Index (NDOI), which is a daily average flow at the confluence of the Sacramento and San Joaquin Rivers calculated as daily Delta river inflows, minus estimated net Delta consumptive use and minus Delta exports. The X2 requirement is based on interpolated values from electrical conductivity (EC) measurements (a surrogate for salinity measurements) made at monitoring stations along the axis of the estuary. The springtime (February through June) standard for X2 is indexed to monthly flows into reservoirs in the eight largest rivers draining into the Bay-Delta. This requires water to position X2 further downstream in wet months than in dry months either by increasing reservoir releases or decreasing exports from the Delta (USEPA 2012). By requiring that X2 be positioned seaward of one of three locations in Suisun Bay for various numbers of days each month, variability in flow is introduced depending on the hydrologic conditions derived from the previous month's "Eight River Index." The Board has so far not set standards for managing X2 in times of year other than springtime, relying instead on the specific NDOI requirements in those months. The minimum NDOI standards in summer through winter (July through January) range between 3,000 and 8,000 cfs, depending on water-year type. The X2 springtime standard does allow options in different months for compliance based on outflows in the range of 7,100 cfs to 29,200 cfs. Exact details on the current Delta outflow requirements can be found in SWRCB (2006, Tables 3 and 4) and in SWRCB (2000, p. 150).

In considering our charge (below), the Panel has been mindful of several of the conclusions drawn from the Schubel workshop report. In Conclusion #2, the report notes that standards should be based on an index that is straightforward to measure, is ecologically relevant, that reflects a number of estuarine properties and processes, and is meaningful to many. The X2 standard satisfies many of these qualities, and the monthly indexing of specific positions for the isohaline within the estuary to a measure of unimpaired flow[2] (the Eight River Index) was intended to meet one of the Schubel report's other conclusions (#5), that seasonal, annual and interannual variability is a key characteristic of estuarine systems.

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[1] Salinity in this report is expressed according to the Practical Salinity Scale, 1978 (PSS-78). Because salinity is a ratio, the value is dimensionless (no units), although it is sometimes reported as "practical salinity units" (psu). Before the development of the PSS-78, salinity was commonly reported in "parts per thousand." The unit of "ppt" was in use at the time when X2 was first considered for use as a salinity standard for the San Francisco Bay-Delta. Salinity values in ppt and psu are essentially equivalent, by design.

[2] Unimpaired flow is a hypothetical flow that would be delivered to the estuary without water storage, diversions, and exports, both upstream and in the Delta, but in the presence of the existing channels and levees.

Simple indices that can be readily understood are undoubtedly useful management tools, but they do not, as Schubel et al. (1993) also emphasize, imply cause and effect. However, in some instances statistical relationships based on X2 have been used as a foundation for flow-related management actions. The National Research Council (NRC 2010) reviewed RPA Action 4 in the U.S. Fish and Wildlife Service (USFWS) Biological Opinion for Delta Smelt (USFWS 2008, p. 369), and identified key questions and uncertainties surrounding the statistical relationships used to determine a suitable position for X2 in wet years to benefit Delta Smelt. This is an example of how generalized indices, despite their broad utility, may be used for purposes beyond those for which they were originally intended.

An additional context for the Panel's work was a 2010 report on Flow Criteria produced by the Board as required by Water Code section 85086(c) (2009 Delta Reform Act). That report (SWRCB 2010, p. 2) observes that "the best available science suggests that current flows are insufficient to protect public trust resources." That technical assessment focused only on flow and operational requirements that provide fishery protection under existing conditions. In addition, the report notes that, whenever possible, flow criteria should be expressed as a percentage of the unimpaired hydrograph. For Delta outflow criteria, the report primarily considered the following species[3]: Longfin Smelt, Delta Smelt, Starry Flounder, Bay Shrimp (*Crangon* sp.), and Zooplankton (mysid shrimp and *Eurytemora affinis*). Following are the summary Delta outflow criteria that are promulgated in the report (p. 98) based on analysis of species-specific flow criteria and other measures:

1. Net Delta Outflow: 75% of 14-day average unimpaired flow for January through June
2. Fall X2 for September through November
    - Wet years X2 less than 74 km (greater than approximately 12,400 cfs)
    - Above normal years X2 less than 81 km (greater than approximately 7,000 cfs)
3. 2006 Bay-Delta Plan Delta Outflow Objectives for July through December

The report ranks criterion 1 as a Category "A" criterion because it has more and better scientific information, with less uncertainty, to support numerical criteria than criteria 2 and 3, which are Category "B" criteria having less scientific information to support specific numeric criteria, but enough information to support the conceptual need for flows. Categories A and B criteria are described as both equally important for protection of the public-trust resource, but there is more uncertainty about the appropriate volume of

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[3] No specific Delta outflow criteria are provided in SWRCB (2010) for Chinook Salmon (various runs) because it was considered that any flow needs would generally be met by Delta inflow criteria for the Sacramento and San Joaquin Rivers, and by Delta outflow criteria determined for the estuarine-dependent species.

flow required to implement Category B criteria. Criterion 2 (fall X2) applies to Delta Smelt and is consistent with the fall X2 action in the 2008 USFWS Biological Opinion (RPA Action 4, as mentioned above). Regarding these criteria, Diane Riddle (SWRCB), during her presentation at the workshop, stated "these [criteria in the report] were developed without balancing other beneficial uses of water, and without considering the cold water pool for salmonids, and without considering economics and other factors." These criteria suggest flows that are needed under existing conditions in the Bay-Delta ecosystem if fishery protection is the sole purpose for which its waters are put to beneficial use (SWRCB 2010, "Note to Readers"). Diane Riddle also commented that "the Board knows it cannot meet 100% of any beneficial uses."

The Panel provides the current report in response to its charge, recognizing that the science on the issues we have considered is rapidly evolving. While the focus of the workshop was on published literature and finalized reports, additional information was available at the time of report submission that the Panel members were not able to consider. In addition, the Panel appreciates the submission of additional background materials by many interested parties after the meeting in February 2014, but has not had the opportunity to review all of this information in detail.

This report begins with a section that provides an overview of X2, its application and how it is calculated, including discussion of recent modeling approaches to assessing the position of the Low-Salinity Zone. X2 is emphasized because it has now been used for nearly 20 years in the springtime Delta outflow standard for fish and wildlife beneficial uses. The main body of this report is structured by the questions posed to the Panel in its charge (see box below). Question 1, regarding key studies and syntheses, is not addressed in narrative. Rather, the Panel has highlighted key papers and reports throughout the text so that the context for their utility is readily apparent. Where particular studies or reports are found to be especially unreliable or questionable in their conclusions, this is pointed out in the narrative responses to questions 2-5 or in the section on X2.

Charge to the Panel

The Panel is charged with reviewing and assessing the provided written materials and oral presentations in order to identify the best available science to inform the State Water Board's decisions on Bay-Delta Plan requirements related to Delta outflow and related factors (Delta outflow requirements). The Panel will evaluate and synthesize the best available scientific information and prepare a report that addresses the following questions:

1. What are the key studies and synthesis reports that the State Water Board should rely on in making their decisions on Delta outflow requirements? Please comment on the strength and relevance of the science presented and reviewed.

2. The existing Delta outflow objectives are based largely on documented relationships between a suite of estuarine organisms and the 2 ppt isohaline (X2).
   - Should these flow relationships still be used as the basis for protecting estuarine fish, estuarine fish habitat, and other important ecosystem attributes?
   - Are there other methods or indicators available to serve as the basis for protecting estuarine fish, estuarine fish habitat, and other important ecosystem attributes? If so, what are they and how could they be applied?

3. What scales (magnitude and duration) of outflow change are needed to produce measurable changes in native species population viability and/or ecosystem function over what time frame? Are there thresholds for achieving specific responses? How could adaptive management experiments be conducted on these scales to inform manipulation of Delta outflow to better protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

4. How are other factors that affect estuarine fish, estuarine fish habitat, and other ecosystem attributes likely to interact with Delta outflow requirements?
   - Are there tools or methods available that could help the State Water Board to better assess the interactions between flow and other factors that affect the estuarine fish, estuarine fish habitat, and other important ecosystem attributes?
   - Can we reasonably expect that addressing other stressors without addressing flow will lead to specific improvements in the status of estuarine fish, estuarine fish habitat, and other important ecosystem attributes?
   - Conversely, can we reasonably expect that addressing flow without addressing other stressors will lead to specific improvements in the status of estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

5. How should Delta outflow be measured and managed to better reflect the flows necessary to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?
   - To what extent does managing winter-spring outflow by X2 reflect the flows necessary to protect estuarine fish? Are there other approaches to managing winter-spring outflow that could improve our ability to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?
   - How should summer-fall outflow be measured and managed to better reflect the flows necessary to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes? Are there other approaches to managing summer-fall outflow that could improve our ability to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

2. Overview of X2 and Delta outflows[4]

X2 was first proposed by Schubel et al. (1993) and later described in the peer-reviewed literature by Jassby et al. (1995). The distances in kilometers from the Golden Gate are illustrated in Figure 1 for Suisun Bay and a portion of the western Delta. The value of X2 is defined as the position, on this distance scale, where the tidally averaged bottom salinity is 2. Salinities between 2 and about 30 are roughly linearly distributed between X2 and the mouth of the estuary (Monismith et al. 1996). X2 marks the Low-Salinity Zone (LSZ), which is defined as a region with salinities of 0.5 to 6 (Kimmerer 2002a), and often marks the vicinity of an important estuarine turbidity maximum (Arthur and Ball 1979). X2 reflects the general physical response of the estuary to changes in flow and provides a geographic frame of reference for estuarine conditions (Kimmerer 2002a). X2 has been shown to have significant statistical relationships with annual indicators of abundance for many estuarine organisms and with estuarine processes, including the supply of phytoplankton and phytoplankton-derived detritus from local production and river loading, benthic macroinvertebrates (molluscs), mysids and shrimp, fish survival, and the abundance of planktivorous, piscivorous, and bottom-foraging fish (Jassby et al. 1995). As such, X2 has been considered a useful index for managing the estuarine gradient to achieve desirable ecological outcomes (Schubel et al. 1993). X2 locations are also correlated nonlinearly with the amount of habitat area and volume within the LSZ (Michael MacWilliams' workshop presentation, Kimmerer et al. 2013).

During the Schubel workshops (Schubel et al. 1993), when X2 was first proposed as a habitat indicator for estuarine populations, X2 was viewed as a variable that could be measured with greater accuracy and precision than alternative habitat indicators such as net freshwater inflow into the estuary. At that time, USGS measurements of Delta outflow using hydroacoustic instruments were not available; these became available a few years later in 1996. It was understood by the Schubel group that X2 would actually be estimated, not truly measured, by interpolation between surface salinity monitoring stations that were located as much as 10 km apart in Suisun Bay. In the recent workshop that was held to provide a foundation for the development of the present report, Russ Brown and Michael MacWilliams discussed a number of persistent issues regarding the accurate estimation of X2 using either the surface salinity measurements or predictive equations based on Delta outflow.

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[4] This section has strong relevance to Question 5.

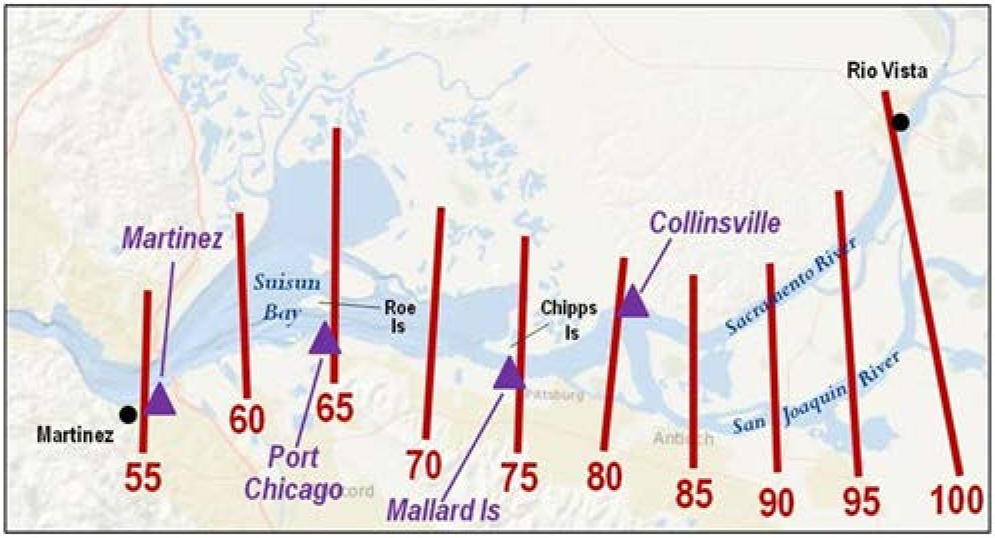

Figure 1. Suisun Bay and western portion of Delta with lines positioned at nominal distances (km) from the Golden Gate Bridge along the axis of the estuary (adapted from Jassby et al. 1995). Also shown are the locations (triangles) of four continuous monitoring stations for electrical conductivity used in interpolating daily values of X2.

Measuring and estimating X2

As long as X2 continues to be used as an indicator of the response of the estuary to outflow, the procedures used for measuring and estimating its value will remain important. X2 has been estimated using four methods:

- By interpolating between observed surface salinities at shoreline monitoring stations located along the axis of the estuary
- Using auto-regressive relationships based on the previous value of X2 and Delta outflow
- From calculations with hydrodynamic models (most recently 3D models)
- By interpolating between observed bottom salinities (taken from full vertical profiles of salinity) collected approximately 5 or 6 km apart during monthly USGS cruises down the central, deep-water channel of the estuary

The original time series of daily X2 values was estimated for the period 1967–1992 through interpolation of surface salinity using six shoreline monitoring stations and assuming a correction for surface-to-bottom salinity variation (stratification) of 0.24 (Schubel et al. 1993, Appendix A by Kimmerer and Monismith). During periods of data gaps, the following equation was used to estimate X2:

$$\mathrm{X2}(t) = 10.16 + 0.945 \cdot \mathrm{X2}(t-1) - 1.487 \log_{10}(Qout(t)) \qquad (1)$$

where X2(t) and X2$(t-1)$ are the positions of bottom salinity 2 at times t and $t-1$, respectively, and $Qout(t)$ is the net Delta outflow in cfs. Equation 1 is now used in estimating X2 by the Department of Water Resources (DWR) DAYFLOW[5] computer program (DWR 2002), and is the equation currently recommended for use by the IEP (Mueller-Solger 2012). Although the above equation (repeated as eq. 1 in Table 1) was attributed to A. Jassby by Kimmerer and Monismith (the authors of Appendix A), Monismith et al. (2002) cited a similar equation (eq. 3, Table 1),[6] but with different parameters, from Jassby et al. (1995). The actual equation in Jassby et al. (1995) (eq. 2, Table 1) is different from both equation 1 and the equation cited by Monismith et al. (2002). This apparent mix-up in attributing similar, but three clearly different, equations to Jassby seems to have created some confusion over the years. Based on discussions the Panel has had with S. Monismith and W. Kimmerer regarding the three different "Jassby" equations, we determined that equation 3 (Table 1) is incorrect. It resulted from an error in Monismith et al. (2002) converting equation 1 from cfs to cms flow units. Also, according to Kimmerer, the relatively slight differences in equation 2 (Table 1) from equation 1 was because of rounding of the parameters (in the metric form of equation 2) based on their respective confidence limits. Equation 1 has the parameter values carried out to more decimal places, which seems appropriate as these are the best estimates of the actual parameter values. For consistency, and to avoid any further confusion, equation 1 should be the "Jassby" equation that is used henceforward.

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[5] DAYFLOW is also the program used for estimating the Net Delta Outflow Index (NDOI).

[6] Except for equation 1, all of the X2 equations were presented in their original papers using units of flow in cubic meters per second (cms) rather than cfs. In Table 1, all the equations have been converted to units of flow in cfs so that they can be more directly compared. The reader is reminded that the equations in Table 1 use a mixed set of English (cfs) and metric units (km).

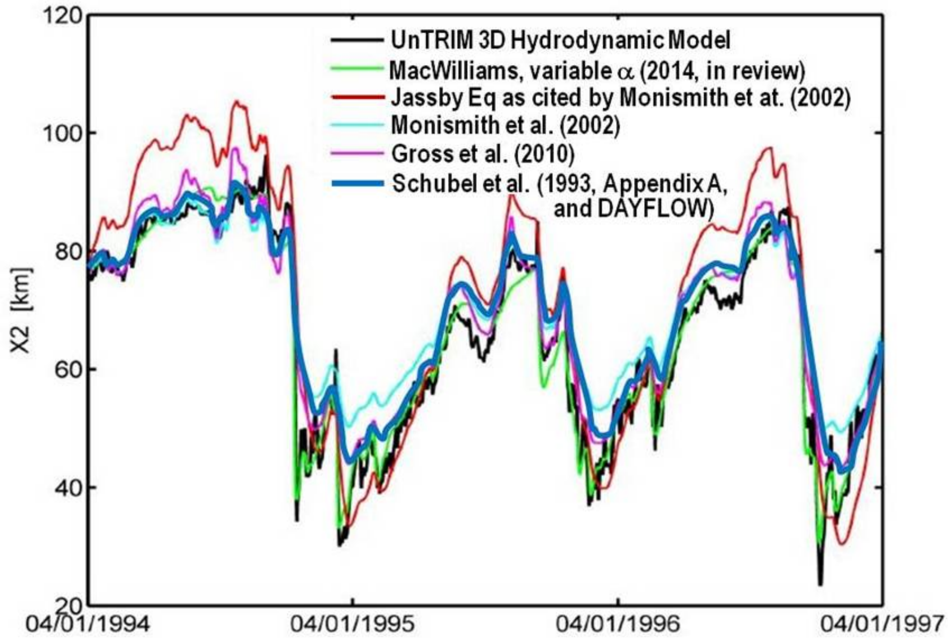

Figure 2. Predictions of X2 for the three-year period from April 1994 through April 1997 using various auto-regressive equations (see Table 1) and the UnTRIM 3D hydrodynamic model. (Revised Slide 16 from M. MacWilliams presentation.)

Citation	Autoregressive Equation (X_2 in km, Q in cfs)	RMS Error (km)[1]
1.) Schubel et al. (1993), Appendix A, (DAYFLOW)	$X_2(t) = 10.16 + 0.945 \cdot X_2(t\text{-}1)\ \text{-}\ 1.487 \cdot \log_{10}(Q_{cfs}(t))$	6.11
2.) Jassby et al. (1995) (not plotted in Figure 2)	$X_2(t) = 10.3 + 0.945 \cdot X_2(t\text{-}1)\ \text{-}\ 1.5 \cdot \log_{10}(Q_{cfs}(t))$	7.33
3.) Jassby eq. as cited by Monismith et al. (2002)	$X_2(t) = 13.76 + 0.945 \cdot X_2(t\text{-}1)\ \text{-}\ 2.3 \cdot \log_{10}(Q_{cfs}(t))$	9.22
4.) Monismith et al. (2002)	$X_2(t) = 0.919 \cdot X_2(t\text{-}1) + 22.43 \cdot Q_{cfs}(t)^{-0.141}$	7.47
5.) Gross et al. (2010)	$X_2(t) = 0.910 \cdot X_2(t\text{-}1) + 36.16 \cdot Q_{cfs}(t)^{-0.182}$	5.31
6.) MacWilliams et al. (in review) with flow-dependent α	$X_2(t) = \alpha \cdot X_2(t\text{-}1) + (1-\alpha) \cdot 644.9 \cdot Q_{cfs}(t)^{-0.230}$	Constant α 4.17 Variable α 3.10

[1] RMSE based on differences with X2 calculations using UnTRIM 3D hydrodynamic model for 4/94 to 4/97

Table 1. X2 auto-regressive equations and RMS errors. (Adapted from M. MacWilliams presentation with citations revised and all equations converted to units of flow in cfs).

Since 1992, the daily X2 estimates used for "X2-abundance[7]" relationships and for other interpretive analyses have all used the auto-regressive relationships. Equation 1 appears to have been used most often because of its inclusion in the DAYFLOW program, although other equations (eqs. 5 and 6, Table 1) have been proposed recently that may be promising alternatives, as noted in the presentation by M. MacWilliams. In Appendix A of the 1993 Schubel report, Kimmerer and Monismith also provide a regression equation for estimating monthly X2 values that is used in the DWR/USBR CALSIM II planning simulation model (for SWP and CVP operations) to determine compliance with the X2 requirement in the Board's 2006 Bay-Delta Plan. As noted in the presentation by Russ Brown, the CCWD G-Model (Denton 1993), which uses a somewhat more complex regression to relate Delta salinity to Delta outflow, has also been used to estimate X2 and is available as an option in CALSIM II.

To meet the springtime operational objectives for X2 and Delta outflow, the continuous monitoring stations are being used (data available online starting in 2007) for interpolation of X2 when it lies between 56 and 81 km (Fig. 1). These interpolated daily values of X2 are referred to as "CX2" and are available in the DWR California Data Exchange Center (CDEC) database

(see: http://cdec.water.ca.gov/cgi-progs/stationInfo?station_id=CX2).

The four stations used are those at Martinez (56 km), Port Chicago (64 km), Mallard Island (74 km), and Collinsville (81 km)(Fig. 1). The three Suisun Bay stations of Martinez, Port Chicago, and Mallard Island are spaced about 10 km apart. Each of the stations has upper and lower measuring probes, although the lower probes were added in later years and are at varying depths from the free surface, so the surface salinities are still being used in the operational computations for X2 with a vertical salinity difference of 0.64 (M. MacWilliams, workshop presentation) built into the computations as the implied stratification between the surface and bottom at the location X2.[8] The stratification of 0.64 assumed in the operational procedure is meaningfully greater than the value of 0.24 that was originally used by Jassby et al. (1995) in developing their daily time series for X2. Whereas the lower probes at the monitoring stations are mostly positioned near the estuarine bottom at the shoreline location of the stations, they are often well above the bottom elevation in the center of the deep-water channel (see Bergfeld and Schoellhamer 2003) where the salinity is needed for the estimate of X2. It should be noted, however, that

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[7] Jassby et al. (1995) and others related X2 to fish abundance, fish survival, and invertebrate abundance. In the present report, all organism responses to X2 are referred to as "X2-abundance" relationships.

[8] The calculations for CX2 are based on EC. They assume the bottom salinity of 2 (EC of 3.80 mmhos/cm) occurs where the surface EC is 2.64 mmhos/cm (salinity of 1.36).

because a primary source for the vertical turbulent mixing in estuaries is the flow over the rough bottom boundary, it is typical to observe less stratification in the lower half of the water column than the upper half, and so a measurement exactly at the bottom, although desirable, may not be essential.

Regarding CX2 and how it is used operationally in regulating X2, the Panel believes MacWilliams had a valid point in his presentation that the stratification assumption may introduce an error in the estimation of X2 by as much as 3 km. This error occurs mostly in the landward values of CX2 in the approximate range of 70 to 81 km. Within this range, the assumed stratification of 0.64 appears to be too high, as demonstrated by 3D numerical simulations; X2 is more likely located where the surface salinity is higher than the assumed value of 1.36 (EC of 2.64 mmhos/cm). This suggests CX2 may consistently over-predict X2 values greater than 70 km. For example, when CX2 indicates X2 is at 79 km, it may more likely be at 76 km. This could be leading to greater water costs to meet the standard than intended. The error is biased because the stratification assumed in the CX2 calculation (0.64) is significantly higher than the stratification (0.24) assumed in the X2 time-series data that are used in deriving the X2-abundance relationships. Recent continuous measurements of bottom salinity collected by S. Monismith and M. Stacey at locations along the axis of Suisun Bay as part of the FLaSH studies may shed more light on this error and should be useful in validating the stratification predictions from the 3D hydrodynamic model.

As mentioned by the speakers and also in notes by Mueller-Solger (2012), there are "significant discrepancies" between the CX2 estimates of X2 and those calculated from equation 1 that are available in the DAYFLOW database. The Panel was not entirely surprised to see that the magnitude of errors in X2 from the equations used to predict X2 were so large over the three-year period (April 1994–April 1997) compared by M. MacWilliams (Table 1 and Fig. 2). In general, these equations respond much too slowly on a daily basis to rapid changes in Delta outflow (when the salt field is adjusting) and are not very accurate downstream of 56 km (where stratification is very high) or upstream of 81 km (where stratification is low and when the relative precision of the NDOI estimate used in the equation is sometimes poor). The period considered by MacWilliams is a period of very high variability in flow, and includes an extreme high flow period (the New Year's flood of 1997) and a period of very low flows (summer and fall of drought year 1994). The comparison was made using values calculated from the 3D UnTRIM hydrodynamic model, which itself has an unknown amount of error, but the differences among the equations themselves are relatively large (Fig. 2). The poor results and the especially large RMS error of 9.22 km from equation 3 (Table 1) is explainable because we now know that the equation is incorrect. The RMS error presented for equation 1 is 6.11 km, which is much higher than the standard deviation (basically equivalent to RMS error) of 3.54 km reported for equation 1 by Kimmerer and Monismith (in Schubel et al. 1993, Appendix A, p. A-7). Kimmerer and Monismith compared equation 1 against interpolated X2 data from October

1967 through November 1991. The Panel suspects that the standard deviation reported by Kimmerer and Monismith may have been lower because a sizeable portion of the X2 estimates for high flows were missing from their data, and therefore potentially large errors in X2 predictions are not reflected in their error measure. The high variability in flows during the relatively short (3-year) period used by MacWilliams and the use of 3D model predictions for X2, rather than interpolated measured values, most likely also contributed to the larger error estimate for equation 1 by MacWilliams.

The Panel does not know if the Board has any plans to make use of the X2 auto-regressive equations on a regulatory basis, but because they have been used extensively for various types of analyses by others (most notably in deriving the X2-abundance relationships), we expect the Board has some interest in these. The Board should understand that the errors in these equations for X2 predictions can be high, especially during periods of significant variability in Delta outflow or when X2 lies seaward of 56 km or landward of 81 km.

The measurements of salinity profiles from monthly USGS cruise data have proved useful for estimating water-column stratification under a range of flows at locations where the bottom salinity is 2. Those data, however, do not directly allow estimation of tidally averaged values for X2 because the cruises occur only monthly and the profiles are collected during only one phase of the tide. The use of 3D hydrodynamic models is a promising new approach for estimating X2 directly (and has also been used in combination with the USGS cruise data to estimate X2 for the day of each cruise), but the skill of the 3D models for predicting X2 should be further established with measurements of bottom salinity before they are fully relied upon.

In general, there should be no expectation that the species responses to X2 indicated by the existing regressions, which involve correlations with multi-year collections of seasonal field sampling across multiple stations, would be manifest at the fine time scales that salinity distributions can now be estimated within the estuary.

Key Papers: Jassby et al. (1995), Kimmerer (2002, 2013), Kimmerer et al. (2009, 2013)

Overall, considering the uncertainties in all of the X2 estimating equations and measuring techniques (including CX2), if new X2 standards are proposed or existing standards are continued, we recommend that the Board consider implementing a new field program to provide data to support the estimation of X2. Salinity measuring probes could be deployed at both the surface and bottom of the water column on channel markers at regular intervals along the axis of the estuary. The cost for operating this type of data collection program has come down significantly in recent years because of self-cleaning salinity probes and the falling costs of instrumentation. The new measured data for bottom salinities in Suisun Bay, which the Panel was informed is already available from the FLaSH studies, should provide valuable information on what can be learned from this type of data.

However, even if improved measurement techniques are implemented for acquiring more accurate estimates of daily variations in X2, it should be understood that the X2-abundance relationships indicate nothing about a species response to changes in salinity at time scales finer than one month. In the X2-abundance relationships presented by Jassby et al. (1995) and later papers, mean monthly or seasonal X2 values were used. The monthly or seasonal temporal resolutions of the various abundance indices also are too coarse to provide information on species responses to flow or salinity variations of less than one month. In general, there should be no expectation that the species responses to X2 indicated by the existing regressions, which involve correlations with multi-year collections of seasonal field sampling across multiple stations, would be manifest at the fine time scales that salinity distributions can now be estimated within the estuary.

The Panel is aware of the suggestion in USEPA (2012) to "de-discretize the X2 trigger points" and make the X2 standard more responsive to "the continuous nature of the flow-abundance relationship" by introducing a finer temporal scale to the standard than one month and capturing the temporal variability of flow pulses. Statistical relationships of point data can often infer "continuous" relationships and it is clear that, in nature, physical-biological interactions occur at time steps of less than one month. Ruhl and Schoellhamer (2004), for example, provide some useful insights into the sediment-transport processes in Suisun Bay that occur during the first freshwater pulse of the season. However, we should be mindful of what we do and do not understand about the processes we are trying to manage, especially biotic responses to flow management, and we thus need to give careful consideration to the time and space scales of responses to outflow management. If a reasonable biological rationale for fine-scale management of X2 can be clearly expressed and agreed upon, then it may be implemented in an adaptive management experiment where field data regarding both the physical character of the system and the biological response are also collected to test the rationale. Until this has been accomplished, it is important to remember that the existing X2-abundance relationships do not provide the rationale for fine-scale management of X2.

X2 compared to net Delta outflow

As noted by Jassby et al. (1995), relationships between estuarine resources and net (tidally averaged) Delta outflow can be demonstrated in a manner similar to relationships with X2. Because of the inherently close association between X2 and Delta outflow, biological relationships with either variable are expected to be reasonably similar.

During periods of significant variability in flow, the correlation between X2 and Delta outflow weakens. Monismith et al. (2002) analyzed the covariability of the two variables and determined that the time period required for the salinity field to adjust to inflow variation was approximately two weeks. Kimmerer at al. (2013) determined that this adjustment time varies inversely with flow, and at a low Delta outflow of approximately 3,500 cfs, the time required for X2 to move halfway from its initial position to its steady-state value can be greater than 25 days. There are relationships that have been developed for estimating the approximate steady-state outflows necessary to maintain a given X2 (see: Schubel et al. 1993, Appendix A, Table 2; Monismith et al. 2002, eq. 10; Kimmerer et al. 2013, Table 2), but there is significant scatter in these relationships because the salinity field is influenced by factors other than flow (most notably tidal conditions).

At the time of the Schubel workshops, there was considerable debate over the issue of whether a standard should be based on flow or salinity (X2). Some participants favored flow, and others favored X2. It was argued that any salinity standard would just be a surrogate for a flow standard, so why not just regulate flow if that was the objective?[9] There also was the realization that the relationship between the two variables could change with any engineering modifications to the estuary, such as installing physical barriers in the Bay or Delta or altering Delta channels to improve flow patterns. Today, there would be more concern that the relationship could change as the result of Delta levee failures, restoration activities, or sea-level rise.

At the end of the Schubel workshops, the consensus was to endorse the X2 standard. In their peer-reviewed paper, Jassby et al. (1995) stated X2 was preferred as a predictor because of the higher uncertainty in the estimates of Delta outflow (NDOI from the DAYFLOW program), especially during periods of low flow. Jassby et al. (1995) wrote:

> "*Estimates of X2, with a well-chosen series of monitoring stations, although requiring interpolation between stations, can certainly be accomplished with less uncertainty* [than outflow]*. The more noise in the predictor variables, the weaker the apparent relationship between the response and predictors; we are thus more likely to discover subtle relationships when using measured X2 than when using outflow, particularly at low flows. This difference between the precision of X2 and Qout is most important at short time scales (days), as the fluctuations will compensate to some extent on monthly scales. On the other hand, these short scales may be of interest for some organisms, particularly those that can be affected by pulse flows at certain points in their life cycles.*"

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[9] In Florida estuaries, the distribution and/or abundance responses of various fishes and invertebrates have been related to average surface salinity (Peebles and Flannery 1992), isohaline position (Peebles 2002), and freshwater flow (Flannery et al. 2002). The statistical fits of flow- and salinity-based independent variables have been found to be similar; for freshwater management purposes, organism relationships with flow are preferred because flow is managed directly and because the difficulties of salinity estimation (which are analogous to those encountered during X2 estimation) can be avoided altogether.

The Panel wishes to point out that the existing X2 standard does allow several options for compliance including an equivalent NDOI, so both flow and salinity are actually incorporated in the standard. We do not know, however, if in achieving compliance one option typically takes precedence over another.

During his workshop presentation, MacWilliams raised the issue of inaccuracies in the NDOI estimates during low flows, expressing concerns similar to those alluded to by Jassby 20 years ago. He indicated the NDOI estimates during fall 2013 were more than double the USGS measured outflows and that, based on measured data for salinity intrusion and X2, the NDOI estimates appeared to be clearly incorrect. The average measured Delta outflow during fall 2013 was approximately 2,000 cfs, which failed to meet the Board's minimum outflow requirement of 3,000 to 3,500 cfs for fall months of a critically dry year. This issue may be a concern for the Board if NDOI estimates are found to consistently overestimate the measured outflows during the summer and fall months of future years. It is logical to ask why the measured outflows (rather than NDOI) aren't used for the specific outflow standards during the July-to-January period, and also why they aren't used as the alternative flow compliance option in the springtime X2 standard. Also, does the availability of the measured outflows now remove any concern that Jassby et al. (1995) had regarding uncertainty in using outflow as the predictor variable during low flows? For the USGS estimates to be used as an outflow standard, several problems will need to be addressed, including gaps (missing data, especially during gage servicing), availability, short-term variability (because of the spring-neap tidal cycle and meteorological influences), and negative values (during periods when the Delta is filling). Although a precise estimate of the accuracy of the measured outflows is not known, the measured values should be more accurate than the NDOI as long as the four monitoring stations used in the calculations are operating properly.

X2 and calculations of habitat area

Salinity is often used to define habitat suitability for coastal species. Habitat Suitability Index (HSI) analyses involve the specification of functions that assign values from 0 to 1 over the range of each important environmental variable (USFWS 1981, Draugelis-Dale 2008). These functions can be either continuous or piece-wise linear. The basis for the shape of these functions is usually determined by expert opinion and monitoring data. If there are multiple environmental variables, then the suitability values are arithmetically or geometrically averaged. This results in a single, final value for habitat suitability that also ranges from 0 to 1. These HSI metrics have many advantages, but also some key weaknesses (Ahmadi-Nedushan et al. 2006, Gore and Nestler 2006).

The main advantage to habitat suitability and related habitat-based analyses is that these approaches have a long history of use in wildlife management in general and especially in fish habitat management. They use readily available environmental data and avoid the controversy and debates associated with population dynamics models (USFWS

1981). However, they are periodically questioned. Major disadvantages are: an increase in suitable habitat does not necessarily result in an increase in fish or wildlife; the outcomes of the HSI analyses are quite subjective because the models are often based on opinions that are seldom peer-reviewed; the HSIs are seldom calibrated; and they are always based on single species and may not reflect actual habitat requirements or community dynamics (e.g., Brooks 1997, Roloff and Kernohan 1999, Van Horne and Wiens 1991). HSIs are nevertheless used in many situations, such as environmental impact assessments and habitat protection plans, because the advantages often outweigh the limitations; some management decisions must be made with whatever data, science, and informed opinions are readily available at the time.

Standard HSI analyses differ from, but are related to, the "Resource Selection Functions" (RSFs) in the habitat analyses reported by Kimmerer et al. (2009, 2013). Kimmerer et al. used field data for abundance (mean catch per trawl) and frequency of occurrence, which were related to salinity, depth, and Secchi depth using generalized additive models (GAMs); the GAMs constituted their 0 to 1 functions (i.e., their RSFs). RSFs were calculated for multiple species. Standard HSI analysis is usually one or two dimensional, meaning it is site- or area-specific. In contrast, Kimmerer's analyses were three dimensional and calculated volume of habitat. Habitat volume is most relevant to pelagic organisms.

MacWilliams (USEPA 2012, p 24-31) recently used the three-dimensional UnTRIM hydrodynamic and salinity model to generate maps and figures, producing estimates of two-dimensional areas and three-dimensional volumes of salinity-based habitat; this facilitated the visual presentation of spatial salinity patterns in the LSZ and identified the position of the LSZ relative to physiographic features of the estuary (such as tidal flats in Suisun Bay). This presentation also included demonstrations of how the locations and sizes of particular salinity zones changed through time under different outflows and water-year types. The utility of such model-derived indicators depends, in part, on how well the underlying model (e.g., hydrodynamics and salinity) simulates the system. Hydrodynamic models of the Delta are steadily improving, although whether they are sufficiently calibrated and validated to generate fine-scale dynamics related to variable outflows is yet to be determined. In addition, these models have not been extended to dynamics of nutrients and lower trophic levels, which would help refine the descriptions of salinity-zone areas and volumes. Adding such habitat-related factors to spatially and temporally dynamic maps of salinity area/volume would provide additional ecological context for the interpretation of X2 and outflow.

In new results presented at the workshop, MacWilliams extended his calculations with the UnTRIM hydrodynamic model and displayed daily time series of area, volume, and depth of the LSZ for historical simulations during the period 1991–2010. The Panel feels that this work is valuable, but that the conclusion "long-term trends show a decrease in fall LSZ area" should be examined more closely, and only after longer simulations have been investigated. This conclusion is important because it has ramifications for analyses related to fall habitat for Delta Smelt. The hydrodynamic model appears to be calculating anomalously low (seaward) values for X2 and high values for LSZ area during the drought year of 1992. The simulated X2 values are approximately 10 km lower than the DAYFLOW equation estimates, and do not appear to match measured salinity data in the western Delta. These results should be verified to determine if the model was out of calibration during 1992. If the simulations were to be extended backward in time through the drought years of 1987–1992, we believe they would reveal that the drought period had smaller areas and volumes of fall LSZ habitat than the later six-year period (2000-2005) of the Pelagic Organism Decline (POD), when catches of four pelagic fishes (Delta Smelt, Longfin Smelt, juvenile Striped Bass, and Threadfin Shad) simultaneously declined in Fall Midwater Trawl survey and other surveys. We expect that the drought-year areas and volumes would be much lower than those of the wet years that occurred in the mid- to late-1990s.

Expansion of indicators to include rates, processes, and early-life stages rather than just standing stocks will be useful. It is well known that nearly 95% of coastal organisms have an estuarine-dependent life cycle (Day et al. 1989), and it is common for only the early-life stages, and not the adult stages, to be responsive to estuarine habitat conditions (e.g., conditions in the LSZ).

Use of percentage of unimpaired flow as an outflow objective

One of the conclusions from Schubel et al. (1993) was that seasonal, annual and interannual variability in salinity and other properties is a key characteristic of estuarine systems. In addition, one of the key summary conclusions from the Board's Flow Criteria Report (SWRCB 2010) was the determination that the ability for flow variability to mimic variability in the "natural hydrograph" should be built into flow criteria. The report states that "criteria should reflect the frequency, duration, timing, and rate of change of flows, and not just volumes or magnitudes." Moyle and Bennett (2008) point out that the life history strategies of all native estuarine delta fishes have adapted to the natural variability of flows in the estuary. Moyle et al. (2010) discuss how both habitat variability and complexity are needed by these species.

The Flow Criteria Report has proposed the use of percentage of unimpaired flow (UF) as an objective for Delta outflow, as well as for upstream flow objectives on the Sacramento and San Joaquin Rivers. Additional supporting information is provided in Fleenor et al. (2010). The specific numeric criteria for Delta outflow calls for 75% of the 14-day average UF for January through June to replace the existing X2 standard that presently runs from February through June. The report points out that the UF criteria are not to "be interpreted as precise flow requirements for fish under current conditions, but rather to reflect the general magnitude of flows under the narrow circumstances analyzed." The Panel interprets "narrow circumstances analyzed" to mean considering fish and wildlife beneficial uses only.

Although the details are unspecified for exactly how UF would be used in formulating a standard (e.g., At what frequency can values be made available? Would flow-routing to the Delta be considered?), the Panel supposes it would be implemented as either a direct outflow standard for the NDOI or (possibly) be translated into an X2 standard in a similar way that the existing standard uses the Eight River Index.

UF is an imprecise estimate, as it is based on a number of assumptions, but it is widely used to represent the total potential water supply available to the estuary. It also is interpreted as an approximate indicator for the natural variability in the hydrograph, and is used as an index for D-1641 water-year type classification. UF is a hypothetical flow that would be delivered to the estuary without water storage, diversions, and exports, both upstream and in the Delta, but in the presence of the existing channels and levees. "Full natural flow," "natural flow," "natural runoff," and "unimpaired flow" are all phrases that have been used by the DWR in various publications to represent the runoff from a basin that would have occurred had man not altered the flow of water in the basin (DWR 2006). DWR now, however, makes an important distinction between "natural flow" and UF (Chung and Messele 2011). Natural flow is a theoretical flow derived with the watershed in a pre-development or virgin state, where "pre-development" refers to the mid-18th century before the first European settlers arrived and land use began to change. Estimates of natural Delta outflows have been constructed using models to calculate the amount of flow that would occur under the pre-development land use conditions, but assuming the contemporary climate. DWR notes at least four reasons that UF differs from "natural flows":

1. The ground water accretions from the very large area of the Central Valley floor probably were considerably higher under natural conditions.
2. The consumptive use of the riparian vegetation and the water surfaces in the swamps and channels of the Central Valley under a natural state may have been significant.
3. During periods of high flow under natural conditions, Central Valley rivers would overflow their banks and water could be stored in the valley for long periods of time and could interact with item 2 above.
4. There were differences in the outflow from the Tulare Lake Basin under natural conditions.

According to presentations made to the Panel and the additional materials provided, The Bay Institute estimates mean natural Delta outflow as 23 million acre-feet (MAF) per year, or about 85% of the estimate for mean annual unimpaired Delta outflow. The State Water Contractors' (SWC) estimates of natural Delta outflow are in the range of 15–16 MAF/yr, which is under 60% of the mean annual unimpaired Delta outflow. Speaking for the SWC during the workshop, Chuck Hanson concluded that the SWC analyses indicate current annual Delta outflow is already about equal in magnitude to "natural" Delta outflow. In reality, there is very large uncertainty in estimating natural flows. It is not possible for our Panel to comment on whether either of these is a correct number. If the Board would like further clarification on best estimates for natural flows, an independent review of the work done on this issue should be conducted. The debate about natural flows may continue as long as a percentage as high as 75% of UF is considered for use as a possible flow objective.

In a prior presentation to the Board on UF that our Panel reviewed, DWR (Chung and Messele 2011) stated that the use of UF as an operational flow criterion "will require further improvement" and "careful design, time, and expert effort." Implementing a UF criterion in real-time operations would require timely acquisition of additional field data to estimate UF; these calculations are currently made retroactively at multi-year intervals after data become available.

The Board should recognize that there are advantages and disadvantages to a flow objective based on percentage of UF. An objective based on UF does not take into account antecedent conditions or reservoir storage levels, existing biological conditions, or alternative priorities for allocating water. In some years, a UF standard may not meet the minimum flow needs of one or more species. For example, a UF standard may not meet minimum outflow needs during a critically dry January, thus failing to address concerns that Longfin Smelt eggs that began incubating in December are vulnerable to salinity intrusion (as discussed by Randy Baxter during the workshop). A small increment of flow above the required percentage of UF may, during times of dry hydrology, result in direct benefit to one or more estuarine species or to the ecosystem. In general, the Board's analyses have so far only considered the percentage of years during which flows of certain above-average magnitudes are exceeded (frequency of exceedance), but (to the Panel's knowledge) their analyses have not examined the percentage of years during which certain minimum or low flows are not reached (frequency of non-exceedance). When considering UF standards, the Board should also consider that situations will occur where there are trade-offs between species. For example, if upstream reservoir levels are low in April or May during a period of late season rains and above-average flows, should runoff be captured in the reservoirs for maintaining a cold-water pool for salmonids, should it be used to increase fall outflow for the benefit of Delta Smelt, or should it be released to the estuary to meet a standard based on UF? When trade-offs of this kind develop, it may be possible to make a choice based on an assessment of overall conditions.

If the Board decides to increase the allocation of environmental water with new Delta outflow standards, doing so with at least a portion of new water dedicated to use in adaptive management may be appropriate. We mention this only for the Board's consideration, and not as a recommendation, as this is beyond our charge. There may be an opportunity to consider using water for directed purposes in either winter (outflow to benefit Longfin Smelt during January of critically dry years), spring (increased outflow to benefit multiple species), or fall (outflow to benefit Delta Smelt in wet or above-average years). Allocations of environmental water could be looked at on an annual (or even longer basis), and water that is saved in one season may be reallocated to another with, of course, an understanding that reservoir storage needs must be met.

3. Question 1

Question 1. What are the key studies and synthesis reports that the State Water Board should rely on in making their decisions on Delta outflow requirements? Please comment on the strength and relevance of the science presented and reviewed.

This question is not addressed in narrative form. Rather, the Panel has highlighted key papers and reports throughout the text so that the context for their utility is readily apparent. Where particular studies or reports are found to be especially unreliable or questionable in their conclusions, this is pointed out in the narrative responses to questions 2-5.

4. Question 2

The existing Delta outflow objectives are based largely on documented relationships between a suite of estuarine organisms and the 2 ppt isohaline (X2).

Should these flow relationships still be used as the basis for protecting estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

Are there other methods or indicators available to serve as the basis for protecting estuarine fish, estuarine fish habitat, and other important ecosystem attributes? If so, what are they and how could they be applied?

For additional discussion of topics related to the third part of this question, the reader is referred to the Panel's answer to Question 4.

X2 as an indicator

The long history of relating X2 to certain species' abundances has been confirmed by several re-analyses (Kimmerer 2002a, b, Kimmerer et al. 2009, 2013). In essence, X2 is the "salinity zone" approach, which is the standard approach used nearly universally to set estuarine flow standards in the U.S. and throughout the world (Montagna et al. 2013). X2 has many good features as an indicator of conditions that relate outflow to species abundance, and is appealing as a single, simple metric for studying and managing the effects of freshwater inflow on the Bay-Delta estuary, but X2 by itself does not capture all of the biologically relevant elements of flow dynamics that affect the estuary. Such extensive capabilities were never the intent of the index. Jassby et al. (1995) recognized that other factors that influence species abundance, but are not correlated with X2, should be considered, and cautioned against "blind adherence" to X2 as a management tool. For example, factors such as the relative contributions to Delta outflows by the Sacramento and San Joaquin Rivers, the distributions of flows in other interior Delta channels, inflows to the Delta from small tributaries and sloughs, the redistribution of flows by operation of Delta gates [Delta Cross Channel (DCC) and Montezuma Slough] and barriers, all may have important effects on abundances and spatial distributions of certain estuarine species that cannot be managed solely by adjusting the position of X2. X2 also does not capture all of the important flow dynamics affected by the proportion of Delta inflow diverted for within-Delta consumption and pumping or any recruitment effects related to organism entrainment at the water pumping facilities. Although X2 is clearly useful and is arguably the primary indicator for those conditions in the LSZ habitat that should be considered when setting outflow objectives for the Bay-Delta, other indicators need to be considered as well.

We recommend that in setting Delta outflow objectives, the State Board should use a suite of indicators, including X2, to ensure ecosystem (beyond individual species) health and to better understand and anticipate how outflow changes will affect not only target species but also other aspects of the ecosystem.

Key papers: Cloern and Jassby (2012), Kimmerer (2004)

We suggest the development of Delta outflow objectives should use a suite of indicators, with X2 remaining as an indicator and accompanied by other, supplemental indicators. Supplemental indicators should be used to ensure ecosystem health (beyond the single-species approach) and to better anticipate and reflect how changes in outflow will affect not only individual species but also other aspects of the ecosystem.

There are several reasons for expanding the indicators beyond X2. First, X2 is based on community structure, not function (i.e., knowing the composition of a community does not necessarily tell you how the community functions), and it is not sufficiently related to *all* species to stand alone as a single indicator that captures the ecological constraints of all species of interest. Second, relationships between X2 and abundance indices are variable in strength and thus have variable predictive confidences (Kimmerer et al. 2009, Table 3). For example, the R^2 for significant regressions of species abundance indices on X2 may range from 3% to 43%. Third, the X2-abundance relationships for some species have exhibited shifts over time, such that these species now show little dependence on X2 or outflow, or now have a changed relationships (e.g., Splittail as shown in K. Hieb presentations). These shifts emphasize the concern that the controlling variable might be a property that co-varies with X2 and not the salinity distribution per se.[10] Many of the statistically significant biological relationships with X2 are non-linear (Kimmerer et al. 2009, Feyrer et al. 2011), and X2 is also non-linearly related to outflow (Monismith et al. 2002). The different degrees of predictive strength and the various non-linearities in the relationships reflect species-specific differences in responsiveness to changes in outflow. Thus, outflow management based on the use of X2-abundance relationships will lead to clearer and quicker responses to changes in X2 or outflow in some species compared to species with highly uncertain X2-abundance relationships. The X2-abundance relationships are not uniform across all species.

Another limitation of X2 that can be addressed by using additional supplemental indicators relates to the relative simplicity of X2. X2 is measureable and estimable compared to many biologically-based indicators, and is a single number, all of which are important advantages. However, this simplicity also entails some limitations in terms of the underlying reasons why species' responses are correlated to X2 (i.e., due to the lack of mechanistic, process-based understanding of the functioning of the system). X2 is an indicator of an unresolved mixture of biological and physical conditions that are often referred to as "habitat quantity and quality," yet description of habitat involves multiple factors with importance that varies over space and time and by species, and whose effects can involve complicated interactions among all of the elements of the environment that sustain a species or a community (Day et al. 1989).

For example, the management-based definition of habitat may involve such easily measured factors as temperature, salinity, and turbidity (e.g., Feyrer et al. 2011) without explicitly knowing whether higher quality habitat was due to faster growth or lower mortality. The X2-abundance regressions use higher densities or more frequent presence, not processes like growth and mortality. The habitat description process then requires further defining the relationship between X2 and these processes to complete the management linkage. This overall discussion was followed in the FWS Biological Opinion for Delta Smelt, and led to debates concerning the statistical methods used and the conceptual interpretation of the inter-relationships involved (NRC 2010). This illustrates how a statistical relationship between habitat and a highly aggregated indicator like X2, without knowledge of the causes for the correlations, can lead to debate and uncertainty about the expected biological responses to changes in X2. This complexity was anticipated by Kimmerer and Monismith (Appendix A to Schubel et al. 1993), who noted "X2 is an *index* of habitat conditions, and can be used as a *predictor* in statistical models, but we do not assert that it is the direct *cause* of any of the responses observed."

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[10] The same estuarine species may aggregate in distinctively different salinities within different estuaries (Peebles et al. 2007).

The simplicity and individual, species-centric aspects of X2 also result in the potential failure of X2 to reflect important ecosystem-level responses that were statistically described under one set of ecosystem conditions, but then applied to ecosystem conditions that changed through time. The application of X2-abundance relationships to a variety of species that have different life histories provides some assurance that the system, as a whole, is responding to outflow management. However, fundamental shifts in the ecosystem, such as shifts in the food-web from pelagic to benthic organisms that affect energy transfer (Nichols 1985), might not be easily captured even by multiple X2-abundance relationships. An example of this is the shift in the relationship between X2 and Longfin Smelt before and after invasion of the estuary by *Potamocorbula*. Following the invasion, there was still a relationship between X2 and Longfin Smelt indices, but the magnitude of the response had shifted (Kimmerer 2009, Fig. 3). There are also likely to be future changes in the ecosystem that will influence ecosystem response to outflow management. For example, the influence of climate change on water temperature (Cloern et al. 2011), the effects of sea-level rise on tidal dynamics and inundation patterns in shallow-water areas (e.g., NRC 2012), and changing riverine sediment supply altering turbidity patterns (e.g., Wright and Schoellhamer 2004) are all examples of potentially important future changes in the system that could influence species abundance and that are not captured in the existing X2-abundance relationships. Further discussion of regime shifts in this system is provided in the answer to Question 4.

Independent analysis of multiple species (i.e., analysis in isolation, one at a time) can miss the signals of fundamental system-level change. The community is comprised of a set of interacting species, and multivariate techniques could be applied to determine how the community as a whole is changing spatially (i.e., with X2) or temporally (i.e., with floods and droughts or changes in turbidity). In addition, establishing robust X2-abundance relationships requires many years of data. Shifts in how energy is routed through the ecosystem can result in relationships estimated with data from one regime being used to predict responses in a changed ecosystem. An example of this is the shift in the relationship between X2 and the native community of bivalves before and after invasion by the Asian clam *Potamocorbula amurensis* (Nichols et al. 1990).

We recommend several steps be taken to further clarify the interpretation of X2 relationships. First, the X2-abundance relationships should be further standardized in terms of the data types and statistical methods used so they will be consistent among species; they should also include estimates of uncertainty derived using the same (standardized) statistical methods. This step should also include a standard and universally applied set of rules for identifying outliers and selecting the years that are included in an analysis. Second, X2-abundance relationships should also be shown using linear scales (i.e., these can be in addition to logarithmic and other transformed scales). The more appropriate transformations and best practices used for statistical analyses must still be used; linear plots are an addition to these analyses. This is important for more clearly showing the magnitude of the expected species response as X2 shifts. Third, the relationships should all use X2 (or else all use outflow) as the explanatory variable.

Additional indicators should be considered to supplement the X2-abundance relationships. As discussed above, formal adoption of a suite of additional indicators would result in outflow objectives that would ensure more effective use of water for environmental purposes and will be essential to consider if the Board is to balance multiple objectives for water use. Additional factors that the Board should consider as they develop additional indicators include: changes in X2 between seasons and water-year types, comparisons of flows to unimpaired flows, habitat suitability, spatial and temporal dynamics of the area and volume of habitat, location and size of the LSZ, water age (residence time), benthos community structure and function, patterns of gross energy flows in the system, and flowpath-related metrics such as the split between Sacramento and San Joaquin flows. It will also be important for species-specific indices to include vital rates (e.g., growth, mortality, reproduction or, by proxy, condition) in addition to indices for standing stock abundance.

Additional factors that the Board should consider as they develop additional indicators include: changes in X2 between seasons and water-year types, comparisons of flows to unimpaired flows, habitat suitability, spatial and temporal dynamics of the area and volume of habitat, location and size of the LSZ, water age, benthos community structure and function, patterns of gross energy flows in the system, and flowpath-related metrics such as the split between Sacramento and San Joaquin flows. It will also be important for species-specific indices to include vital rates in addition to indices for standing stock abundance.

Some of these additional indicators are already being explored by the Board (e.g., the recent workshop on Interior Delta Flows). Such an approach is consistent with the original recommendations from Schubel et al. (1993) who noted (recommendation #7—emphasis is in the original):

> "*At this time,* the most appropriate basis for setting salinity standards for the portion of the estuary on which this report concentrates is the position of the nearbottom *2‰* isohaline *alone,* unless it can be shown either that another variable is the controlling variable or that incorporation of additional variables improves the predictive capability. Further research should be conducted to improve prediction of the responses of important estuarine resources to variations in the position of the near-bottom *2‰* isohaline. That research should incorporate other variables where they can be shown to contribute significantly."

Two decades have passed since the Schubel report was published; using X2 as the sole indicator (at least during spring) has not resulted in the intended protective effect (e.g., Thomson et al. 2010). X2 is not perfect, and the development of additional indicators could ensure that management of Delta outflows will allow explicit consideration of a wider range of attributes than just salinity. However, X2 remains as an index that has some ecological significance—it is an index that integrates a number of important estuarine properties and processes, and thus remains meaningful and readily understood by stakeholders. Despite its shortcomings, we believe the use of X2 as a management tool should be continued, at least in the near term, but there should also be a concerted effort to explore and document the utility of viable alternatives. This is not to say that the specifics of the application of X2 to ecosystem management should not be reviewed and revised as needed, or that its current demonstrable imperfections should not be addressed. Scientific understanding of aspects of the physical and ecological complexities of the Bay-Delta is rapidly evolving. Translating this detailed scientific understanding into management tools that accommodate natural variability in the system (depending on how standards are set), and that do not evolve into over-managing the complex, incompletely understood estuarine system dynamics, is not feasible in the immediate future. Developing an improved approach to managing Delta outflow will require a concerted effort to consider ecosystem responses that are beyond the analysis of (multiple) individual species, allowing process-based anticipation of changes caused by system-wide and local drivers, and encouraging scientific consensus regarding the role of important (and unimportant) factors and processes. In the meantime, effort should be devoted to further understanding and communicating what X2 does and does not mean in an ecosystem context, and to develop agreement on its interpretation to ensure effective management.

Developing an improved approach to managing Delta outflow will require a concerted effort to consider ecosystem responses that are beyond the analysis of (multiple) individual species, allowing process-based anticipation of changes caused by system-wide and local drivers, and encouraging scientific consensus regarding the role of important (and unimportant) factors and processes

5. Question 3.

What scales (magnitude and duration) of outflow change are needed to produce measurable changes in native species population viability and/or ecosystem function over what time frame?

Are there thresholds for achieving specific responses?

How could adaptive management experiments be conducted on these scales to inform manipulation of Delta outflow to better protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

System response to outflow change

Examination of X2-abundance relationships provides insight on the magnitude of changes in X2 and Delta outflow predicted to achieve desired objectives for the protection of beneficial uses. In order to illustrate the issue of scale using actual data, we reproduced relationships for Longfin and Delta Smelt (Figs. 3 and 6) based on Kimmerer et al. (2009) and more recent work (IEP 2013), but present their results on a linear scale, rather than using log-transformed data. To provide some perspective, X2 values between 60 and 75 km result in a Low Salinity Zone in Suisun Bay, which translate to approximately 43 and 12 kcfs, respectively.

The Longfin Smelt abundance index has one of the strongest relationships with the average winter and spring X2 of the variables examined to date [upper panel in Fig. 3, see Kimmerer et al. (2009)]. Decreasing X2 from 75 to 60 km is predicted to result in a more than 5-fold increase in the abundance index. California Department of Fish and Wildlife proposed a winter-spring outflow ranging from 12.4 to 28 kcfs, equivalent to an X2 range of 75 to 65 km, respectively (SWRCB 2010). This is very similar to the current winter-spring range under D-1641 of ~7-29 kcfs. It seems unlikely that this modest increase in the minimum flow would result in a detectable change in the Longfin Smelt abundance index, given the very small difference between predictions for 7 kcfs (~80 km X2) versus 12 kcfs (~75 km X2) in the post-1987 relationship. SWCRB (2010) reported that outflows equivalent to 75% of winter/spring unimpaired flows would result in X2 values westward of 75 km at least 90% of the time. Average outflows of 51 kcfs (X2 of 58 km) could be achieved in 30% of years

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It seems unlikely that the predicted increase in the abundance index under any proposed regime would result in a substantive improvement in abundance of Delta Smelt in the short-term due to stock size limitations.

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under the 75% of unimpaired flow strategy. These larger flows produce X2 values that fall on the steeper part of the Longfin Smelt X2-abundance relationship, leading to potentially large and observable increases in the abundance index (Fig. 3). Under conditions where parent stock size is not limiting, the X2-abundance relationships describe highly variable population responses that are continuous and do not contain distinct thresholds or change points. However, as demonstrated here for Longfin Smelt, benefit-cost relationships vary along these population response curves. When stock size is limiting, multiple, successive years of favorable conditions are required to rebuild stocks, and this requirement is likely to be more important than achieving outflow threshold values during any single year. Evidence for the stock-rebuilding effect was presented by Randy Baxter (CDFG) using graphics derived from Thomson et al. (2010).

Relationships between winter-spring X2 and the tow net survey (TNS) abundance index for Delta Smelt were very different before and after 1982. The relationship actually had a positive slope based on data collected prior to 1982, and a slope near zero for data collected in 1982 or later (lower panel of Fig. 3). There is no evidence from this relationship that the current standard of 7–29 kcfs, or proposed flow criteria of 12-29 kcfs or 75% of unimpaired flow, would result in an increase in the TNS abundance index for Delta Smelt. More recent analyses suggest a negative relationship between the TNS index and X2 once parental stock size effects are accounted for (upper panel of Fig. 4). As current stock sizes are likely very low, the predicted increase in the TNS index with decreasing X2 is expected to be relatively small.

More recent analyses also reveal a potentially negative relationship between average X2 over the fall and the abundance of larval Delta Smelt, as indexed by the 20 mm tow net survey (lower panel of Fig. 4). Minimum flows during fall range from approximately 3- 5 kcfs under the 1995/2006 Bay-Delta Plan (X2 at 4 kcfs = 88 km). Minimum fall flows are 7 kcfs (X2=81 km) under one of the USFWS Reasonable and Prudent Alternatives, and between 7 kcfs (above normal years, X2 <81 km) and 12.4 kcfs (wet years, X2 <74 km) based on the most recent flow proposal (SWRCB 2010). Using only X2, a relatively small increase in the larval abundance index would be expected based on the difference between the current 88 km fall X2 standard and the proposed above-normal year standard (81 km). The fall X2-abundance relationship suggests a relatively large increase in the larval abundance index under the wet-year standard of 74 km. However, there is considerable uncertainty in this prediction because an X2 value of 74 km is well below the range of data used to fit the relationship for the more recent period (2003-2013), and there is substantial uncertainty in that

> *We saw little evidence that the relatively modest changes in fall Delta outflows that are being proposed are going to result in substantive increases in abundance of key pelagic fish species based on their X2-abundance relationships.*

relationship (lower panel of Fig. 4). As with the use of all indices of abundance, the link between changes in the index and changes in the population-level abundances are not claimed to be exact. We emphasize the importance of communicating uncertainty in functional relationships when using them to evaluate the efficacy of various flows.

In the Panel's judgment, based on X2-abundance relationships the evidence that the relatively modest changes in fall Delta outflows that are being proposed are going to result in substantial increases in abundance of key pelagic fish species is highly uncertain. Substantive increases in Longfin Smelt abundance index may be realized under the proposed 75% winter-spring unimpaired flow standard. Even in that case, population changes may be very difficult to detect given the variance of the regression, potentially high observation error in the sampling programs, and the infrequent implementation of high flows, even under the unimpaired flow strategy.

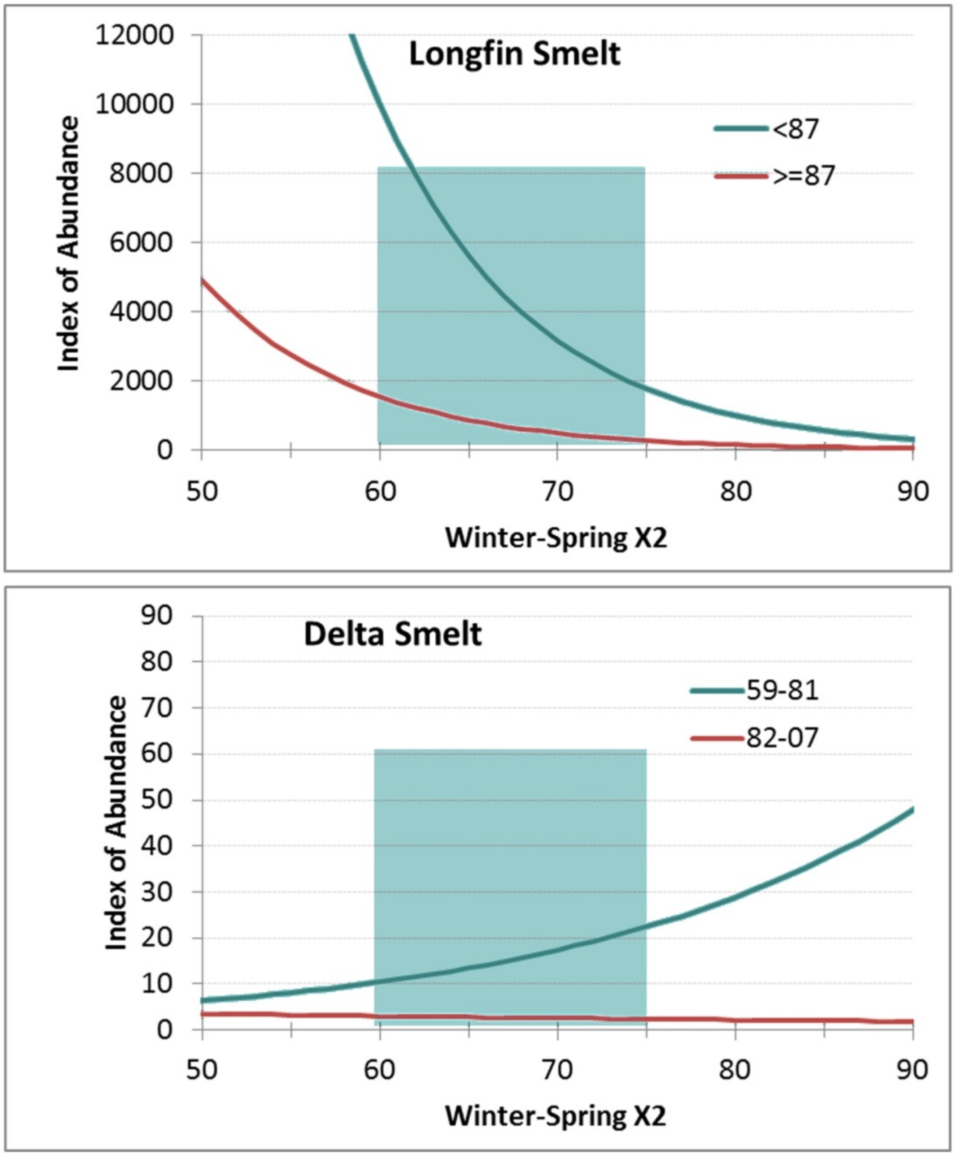

Figure 3. Relationships between Longfin (upper panel) and Delta Smelt (lower panel) abundance indices (mid water trawl and tow net series respectively) and average X2 over the winter-spring period during two different periods of time (before 1987 and after 1986 for Longfin Smelt; 1959-1981 and 1982-2007 for Delta Smelt). These relationships are based on parameters from Table 2 of Kimmerer et al. (2009) transformed from $\log_{10}$ to linear space. The blue boxes represent the X2 range required to achieve low salinity conditions in Suisun Bay.

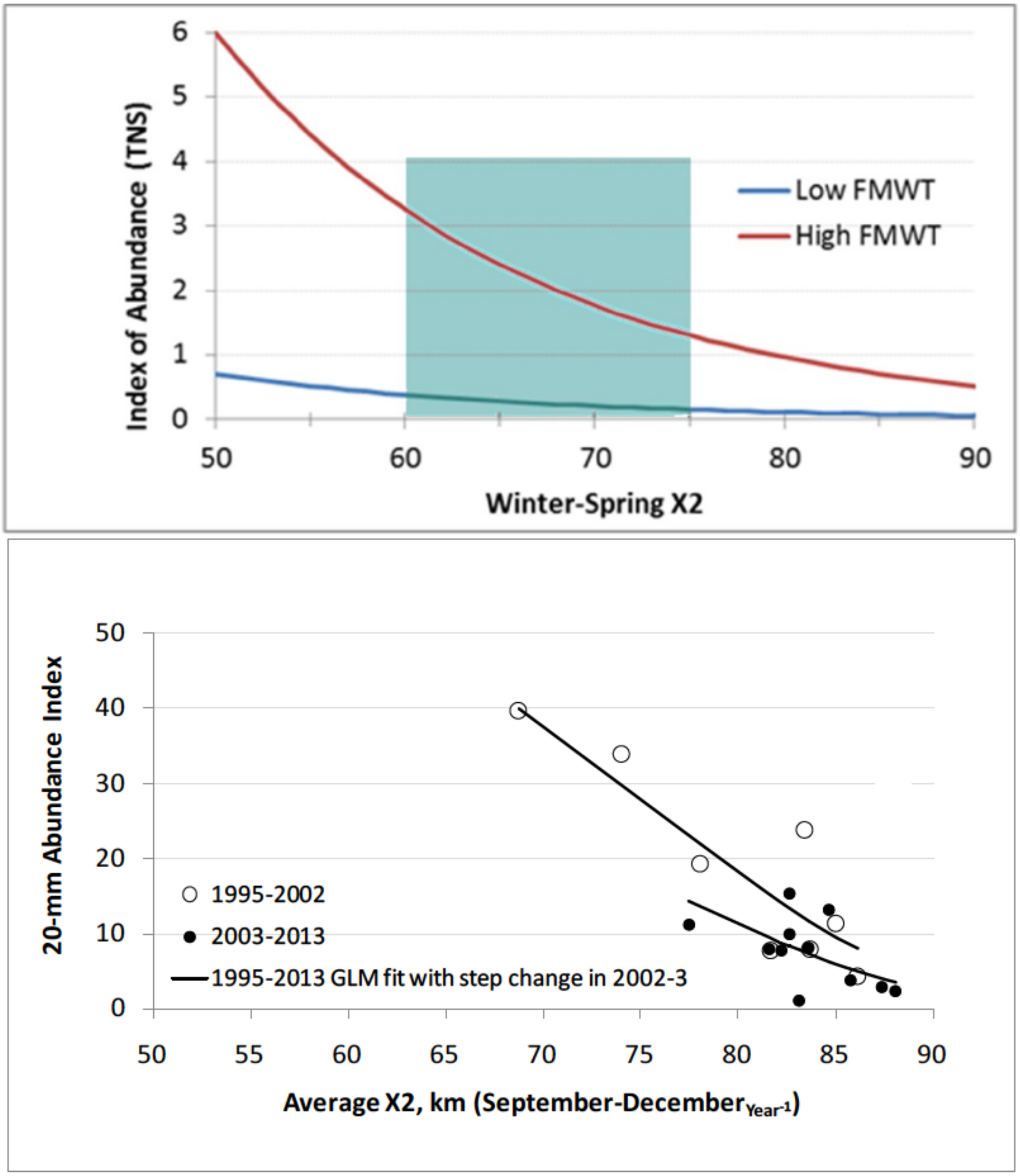

Figure 4. Relationships[11] between Delta Smelt abundance indices and average X2 over the winter-spring (upper panel, tow net series - TNS) and fall (lower panel, 20-mm series) periods (from IEP 2013). The TNS model includes an effect of parental stock size as indexed by the fall mid-water trawl (FMWT) survey. Low and high parental stock values for the plot were based on the approximate averages of indices before (high FMWT abundance) and after 1987 (low FMWT abundance). The blue boxes represent the X2 range required to achieve low salinity conditions in Suisun Bay.

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[11] Adding confidence limits to the preceding figures is not possible, as Kimmerer et al. (2009) only provide standard errors for slope and step-function parameters, but not for the intercept. Because the slope and step-function terms may be correlated, the upper confidence limit cannot be used for the X2-abundance relationship.

Models and uncertainty

A number of scientific publications present models of the relationships between the abundance of pelagic fish species (e.g., Longfin Smelt, Delta Smelt) and physical and biological characteristics in the Bay-Delta. The datasets used for the various modeling efforts are impressive, but also have limitations. Annual fish abundance indices are derived from trawl and tow-net surveys conducted at approximately 100 sites from San Pablo Bay to the eastern Delta over the last 40 years, where sampling was conducted monthly for more than half of the year. Extensive time series of physical and biological covariates (e.g., prey availability) are also available. Few coastal systems have such consistent, lengthy, and spatially extensive time series at multiple ecosystem levels (phytoplankton, zooplankton, fish).

While extensive, there remain important limitations in the dataset. For example, the fish survey indices (I) are a proxy for actual abundances (N). The proportion of a fish population captured by the survey (q, or catchability) cannot be estimated precisely, which at a minimum leads to imprecision in the relationship between the index and actual abundance ($I{\sim}qN$). Changes in q over time could lead to erroneous conclusions about trends in population size. For example, as argued by Presenter Robert Latour, increasing water clarity could lead to greater avoidance of nets (decreasing q), an underestimate of the size of the population, and thus an overestimation of the extent of population decline. Changes in X2 or Delta outflow could affect the spatial distribution of fish populations, which could change q, resulting in potentially biased assessments of the effects of flow on abundance. Finally, models predicting fish abundance indices are based on data from a survey design where a number of potentially important variables change over time in an uncontrolled way. As mentioned in the discussion of adaptive management, this can make it difficult to separate the effects of different variables, and it also leads to considerable uncertainty about the cause-and-effect relationships driving observed statistical relationships (e.g., X2-abundance). There are some studies of the Delta that attempt to provide more information on mechanisms by focusing on specific questions using specific techniques (e.g., acoustic tracking of smolts to study predation mortality in the southern Delta; otolith microchemistry of Longfin Smelt).

Model-based publications can be organized according to the complexity of the analysis, ranging from relatively simple models that describe the response of abundance at a single life stage to one or a few abiotic variables (e.g., X2), to models of intermediate complexity that account for the effects of multiple abiotic and biotic covariates and density dependence (effect of parental stock size), to complex life cycle models that consider the effects of parental stock-size at multiple life stages as well as the effects of abiotic and biotic covariates that can impact survival before and after density-dependent processes. Here, we provide a brief summary of important findings and limitations for each of these model types. We also comment on a fundamental relationship used in development of recent Delta outflow criteria (SWRCB 2010).

Simple statistical models

Jassby et al. (1995) provides a analysis and discussion on the utility of X2 as an index of an estuarine community's response to freshwater inflow, and examines relationships between the abundance of organisms at multiple trophic levels (phytoplankton, zooplankton, shrimp, pelagic fish) and X2. Kimmerer et al. (2002) extended the analysis to provide greater support for (generally negative) relationships between abundance and X2, and quantified the extent of step changes in the X2-abundance relationships in the late 1980s. As in Jassby et al. (1995), Delta Smelt was one of the few species analyzed that did not show a negative relationship between abundance and X2. These papers have thoughtful and balanced discussions on the potential mechanisms by which flow could affect the abundance of pelagic species through different food-web pathways. More recent papers (Kimmerer et al. 2009, 2013) extended the analysis to additional data (more years and life stages) and tested whether the effect of X2 on abundance was consistent with the effect of X2 on modeled habitat changes. They found large discrepancies between the slopes of the abundance-X2 and habitat-X2 relationships for many species (including Longfin Smelt, which showed a strong negative abundance-X2 relationship), suggesting that variation in the volume or area of physical habitat (as defined by salinity) is unlikely to be the direct mechanism behind X2-abundance relationships.

Variation in the volume or area of physical habitat (as defined by salinity) is unlikely to be the direct mechanism behind abundance-X2 relationships

Key papers: Kimmerer et al. (2009, 2013)

More complex multivariate statistical models

Mac Nally et al. (2010) examined the effects of a wide range of flow and non-flow covariates, including parental stock size, on abundance trends for pelagic fishes in the Bay-Delta. They found that X2 and water clarity were the most important variables affecting the abundance of multiple declining taxa, and also found relatively strong interactions between fish abundance and their prey, and between prey availability and X2. In a companion paper, Thomson et al. (2010) provided additional insight on the timing of abrupt changes in abundance trends for pelagic species, and identified 2002 as the year when four important pelagic species began their most recent decline. They found water clarity, X2, and the volume of freshwater exports were the most important factors explaining abundance trends, and that none of the covariates that were examined explained the post-2000 decline.

Full life-cycle models

There has been increasing development and use of life-cycle modeling to try to address the population responses to changes in flow-related variables. Examples for Delta Smelt include Maunder and Deriso (2011), Rose et al. (2013a, b), and an ongoing effort led by Dr. Ken Newman. There are also several efforts related to salmon modeling (Rose et al. 2011). To date, these models have not been fully vetted and evaluated sufficiently to be used for direct management applications. The potential for using life-cycle modeling remains, although such modeling rarely, if ever, resolves issues as complicated as those faced in the Bay-Delta regarding listed fish species.

Which level of model complexity provides the greatest insights?

Applying models of increasing complexity to Bay-Delta data has certainly led to greater insights into factors controlling abundance of pelagic fishes. Application of synthetic life-cycle modeling is appealing, as it integrates data for multiple life stages rather than providing separate assessments for each stage. However, at some point, model complexity surpasses the amount of information available, and predictions and inferences in such cases can become too unreliable for management decision-making. For example, there is often insufficient information in the data to distinguish the effects of different covariates, which then leads to uncertainty in specifying relationships between growth, mortality, and reproduction and the covariates in the model. Jassby et al. (1995) include an excellent discussion about the trade-offs among models of varying complexity in the context of the Bay-Delta. As shown in the simulation work by Walters (1986) that they cite, more complex models will almost always explain more variation than simpler models, but may have poorer performance when it comes to making reliable predictions for policy decisions owing to greater uncertainty and a higher probability of encountering spurious correlations due to over-fitting.

We encourage continued, thoughtful use of multi-stage life-cycle models. Confounding parameters and over-fitting issues can be addressed by simplifying the model structure and by using more restrictive prior assumptions about some parameter estimates.

In spite of the risks, we encourage continued, but thoughtful, use of multistage life-cycle modeling in the analysis of Bay-Delta data (as in the current effort by Dr. Ken Newman et al.). Parameter confounding and over-fitting issues can be addressed by examining alternative model structures (e.g., modeling two rather than three life stages), and by using more restrictive prior assumptions about the feasible range for some parameter estimates. There may be little empirical support for some of these more restrictive assumptions, but at least they will be explicit and their effects can be evaluated through a sensitivity analysis. At a minimum, such analyses provide a deeper understanding of the limitations of the data and have the potential to provide more complete and robust estimates of uncertainty. Many of the uncertain, but restrictive, assumptions that would need to be stated explicitly in a properly documented full life-cycle model are often implicit, but never evaluated, in simpler analyses. A good example here would be the negative relationship between the trend in the 20 mm tow-net series for Delta Smelt and fall X2 (IEP MAST 2013, as presented by Mueller-Solger at the workshop on day 2). If that relationship alone is used to support increased flows, then decision makers are implicitly assuming that increasing the abundance of larval Delta Smelt will lead to a similar increase in the population of adults. This may not be the case if flow has substantial effects on growth and survival in later life stages or if the effects of environmental factors unrelated to X2 are important in determining the ultimate survival to the adult stage. Life-cycle modeling offers a framework for making explicit the calculations from changes in larvae to population-level responses.

Longfin smelt population growth

The State Water Resources Control Board flow criteria report (SWRCB 2010) is an informative synthetic effort that provides the rationale for the most recent set of flow criteria intended to benefit the ecosystem and fish populations in the Sacramento and San Joaquin Rivers and the Delta. In regard to the Delta, much of the information in SWRCB (2010) comes from papers reviewed by the Panel, but the report also includes new analyses, some of which have an important influence on recommended flow criteria. Here, we focus on the relationship between Longfin Smelt population growth and Delta outflow during winter and spring (Fig. 11 of SWRCB 2010) developed by The Bay Institute and National Resource Defence Council (TBI/NRDC). The ratio of fall mid-water trawl (FMWT) indices across adjacent years was used to classify each year as having negative (y=0) or positive (y=1) population growth. These binary values were treated as data and predicted based on logistic regressions using Delta outflow from January through March and March through May. The analysis concluded that approximately 9.1 and 6.3 million acre-feet (MAF) from January through March and March through May would be required to achieve positive population growth in 50% of years, respectively. These volumes are equivalent to average flows of 51 and 35 kcfs and are used to support the January-through-June 75% of unimpaired flow criterion.

The TBI/NRDC Longfin Smelt analysis has some very useful and logical elements. The model predicts the direction of population growth, which is arguably the best metric to use when populations are at low abundance and at significant risk of extirpation or extinction. The model also provides a direct link between flow and the probability of population growth. On the negative side, we feel the strength of the relationship has been oversold because there is no consideration of uncertainty in model predictions. This deficiency is not unique to the TBI/NRDC analysis within the flow criteria report. Here, we repeat the TBI/NRDC analysis in a Bayesian framework, as an example, to highlight the importance of communicating uncertainty to policy makers.

Examination of the data points in the TBI/NRDC analysis shows considerable overlap in flows for years when populations decline (y=0) and grow (y=1), and only four of 20 years with positive population growth had flows larger than those of years with population declines (Fig. 5). Not surprisingly then, the uncertainty envelope for this relationship is relatively wide, and is also asymmetric (dashed lines in Fig. 5). There is greater certainty that very low flows (<5 MAF) limit the probability of positive population growth relative to the certainty in positive population growth at higher flows. Uncertainty in the flow-population growth probability relationship results in considerable imprecision in the recommended outflow criteria required to achieve population growth in 50% of the years (blue lines in Fig. 5). The median outflow required to attain this probable population growth frequency was ~6.9 MAF[12] with a 95% credible interval of 4.3-11.8 MAF. That is, outflow requirements to achieve population growth in 50% of years could be 40% lower or 70% higher than the reported median. Or, put another way, the flow criterion of 6.9 MAF results in a highly uncertain probabilities of positive population growth during a given year; this probability ranges from 20% (2.5 percentile) to 85% (97.5 percentile). These wide ranges illustrate a much different and more uncertain outcome than impressions based solely on the expected value, and the expected value is all that is provided in the flow criteria report (SWCRB 2010).

It is critical that quantitative analyses communicate uncertainty in recommended flow criteria to decision makers

Furthermore, the TBI/NRDC analysis also does not include effects of observation error. Each "data" point in Figure 5 is based on the ratio of abundance indices in adjacent years, which are assumed to be proportional to the actual abundances. However, due to sampling error and potential biases, the annual abundance indices do not track the actual abundance perfectly. Taking the ratio of two uncertain numbers potentially leads to large uncertainty in the determination of negative or positive population growth for each year. That is, there is an unknown but potentially large probability that each data point in Figure 5 is actually on the wrong end of the y-axis. We expect the probability of incorrect assignment to be relatively high for adjacent years with similar population estimates, which are not uncommon (see Fig. 5 of IEP 2013). Accounting for this uncertainty would lead to a wider prediction envelope than presented in Figure 5. However, conducting this

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[12] This result is slightly larger than the TBI/NRDC estimate of 6.3 MAF, likely due to errors introduced when digitizing points off the original plot, and potential differences in the likelihood used for estimation.

analysis is problematic because the precision in the relationship between the index and the actual abundance is unknown and likely variable between years and flow conditions. Exploratory analyses under different assumed precisions could be used to determine the potential increase in the uncertainty. It is critical that quantitative analyses communicate uncertainty in recommended flow criteria to decision makers.

We used the TBI/NRDC analysis to illustrate the role of statistical estimation and the importance of including uncertainty in predictions. This issue, however, applies to many of the other analyses reported in the literature, in parts of presentations to the Panel, and in synthesis reports such as the SWRCB (2010) report.

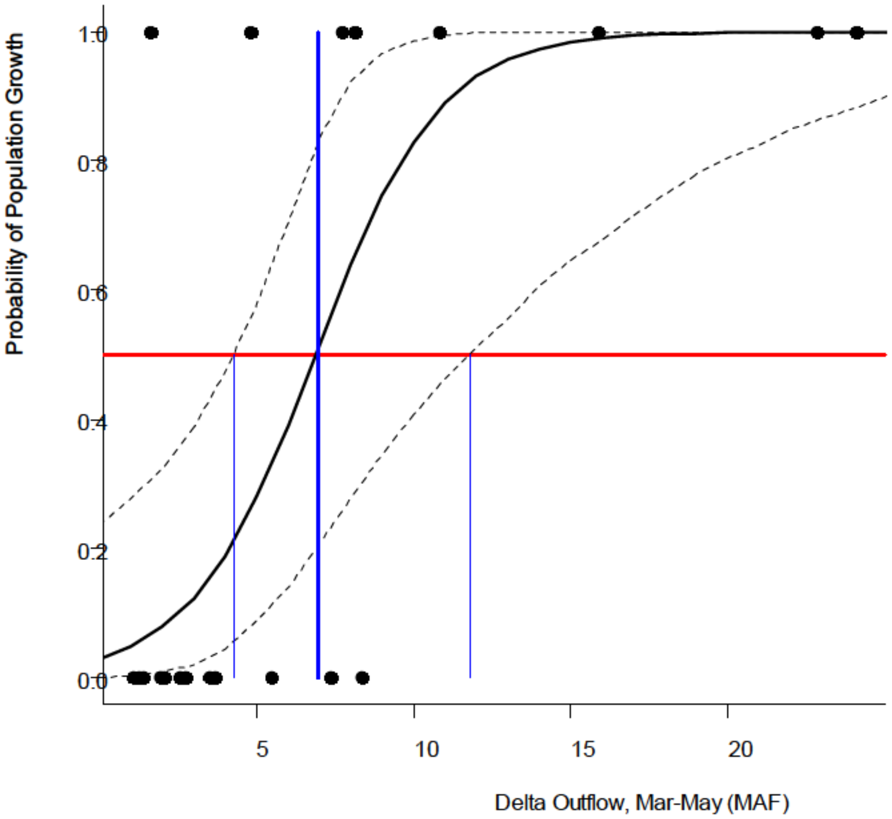

Figure 5. Logistic relationship between March through May Delta outflow and generation-over-generation change in abundance of Longfin Smelt (0 = negative or no population growth, 1= positive population growth). Points are values digitized from Fig. 11 of SWRCB (2010). The thick black line shows the expected logistic relationship based on a Bayesian model, and dashed lines show the 95% credible interval. X-values below the blue vertical lines show the 2.5% (4.3 MAF), 50% (thick line, 6.9 MAF) and 97.5% (11.8 MAF) outflows required to have population growth in 50% of years.

Adaptive management

There are three well-established steps common to all Adaptive Management (AM) programs (Walters 1986): (1) define objectives, the indicators used to represent them, and management actions, (2) develop conceptual and predictive models to evaluate how indicators change with management actions, and (3) implement actions to determine if predicted outcomes have been achieved, and then refine models and actions (and potentially objectives) based on this new information. Attempts to successfully implement AM in the Bay-Delta have been limited. AM was a central tenet of the CALFED Strategic Plan, and has been adopted as a key strategy by subsequent efforts. The Delta Reform Act requires the inclusion of science-based AM in the Delta Plan, and AM is defined in the California Water Code (section 85052) as "*a framework and flexible decision-making process for ongoing knowledge acquisition, monitoring, and evaluation leading to continuous improvements in management planning and implementation of a project to achieve specified objectives.*" The Delta Science Plan notes that "Past attempts to adaptively manage Delta water operations and ecosystem restoration have rarely covered the full AM cycle (i.e., Plan, Do, Evaluate and Respond). There has also been much disagreement about suitable AM actions and the science needed to evaluate their effectiveness." The Science Plan also lays out a nine-step AM process. While detailed discussion of proposed AM approaches are laid out in the Science Plan and in Chapter 3.6 of the Draft Bay-Delta Conservation Plan, challenges remain. The three general steps outlined above occur in some form or another in all AM discussions and thus provide a useful framework for discussing AM in the context of Bay-Delta flow objectives.

Defining objectives and actions

Federal and State Endangered Species Acts (ESA and CESA) provide strong direction on the need to improve the status of particular species in the Bay-Delta. Whether this direction translates into population-level objectives to be met by altering Delta outflow needs to be an explicit decision. Reed et al. (2010) identified the importance of setting specific objectives for any action distinct from overarching programmatic objectives that are more likely to be achieved through a suite of coordinated actions. The State Board's need to set flow criteria must therefore be set in the context of other actions being taken to achieve societal goals, the relative contribution of flow criteria to meeting those goals, likely success of each of the actions, including flow criteria, working collectively and independently, and trade-offs among numerous goals.

Assuming that an objective of setting and meeting outflow criteria is to produce a change in the population of a species, e.g., Longfin Smelt, then consideration needs to be given to uncertainty about whether measured indicators, such as the fall mid-water trawl or other smelt abundance indices, reliably track actual population responses to management actions. In addition, it may be very difficult to observe a population-level effect given limited replication of desired high flow/low X2 events, relatively high sampling error in the abundance indices, and natural inter-annual variation in recruitment and survival rates. For fall outflow, the review Panel for Fall Low Salinity Habitat (FLaSH) studies have recommended that AM activities, e.g., enhanced monitoring, need to occur even in years when the fall outflow action is not taken in order to provide context for response variables

(http://deltacouncil.ca.gov/sites/default/files/documents/files/FallOutflowReviewPanelSummaryReport_Final_9_11.pdf).

A range of possible flow options for the Bay-Delta have been identified by fisheries management agencies and NGOs to achieve pelagic fish and ecosystem objectives, and some of these options would come at very large costs to water users. These costs are also rarely quantified during outflow discussions. It is highly uncertain whether the collaborative adaptive management approach proposed by the Delta Science Program can resolve the extreme trade-offs that exist in the Bay-Delta AM setting. Implementation of new flow criteria is going to be very challenging. Given this situation, quantifiable, achievable objectives for outflow criteria need to be determined. The recent focus on specific, measurable, achievable, relevant, and time-bound or “SMART” objectives, as called for to the maximum extent possible by BDCP, is relevant here. In addition, a systems context for considering outflow criteria should also evaluate non-flow alternatives, such as predator control; to date, such consideration of other options has been relatively limited.

Predicting the response of indicators to actions

Models predicting responses of ecological indicators to management actions can be classified into three categories: (1) highly idealized conceptual models where even the direction of response is difficult to predict, (2) conceptual models (often species-specific) that attempt to qualitatively predict the direction of response, but where the magnitude of the response is unknown, and (3) quantitative models that provide somewhat reliable and often controversial estimates of both the direction and magnitude of response. There are a number of conceptual arguments and quantitative statistical models that support the notion that increased outflow (or lower values of X2) is better for fish. However, the ability of those models to reliably predict responses to particular flows in particular times of the year is likely low given that the response variables are indices that integrate over space and that are not focussed on vital rates (growth, mortality, reproduction). In the Panel’s view, many of the ecological analyses to date have used models that fall into category 2, and in the cases where numerical (category 3) models have been used, they have generally resulted in controversy and debate.

High uncertainty in models predicting biological responses in the Bay-Delta occurs because of potential biases and imprecision of measured indices and, to some extent, due to limitations that are inherent in monitoring data. Problems with the category 3 models are also largely derived from the inability to determine the functional relationships that underlie the models. Biological models of the Bay-Delta system are based on data from synoptic surveys rather than explicit experiments that address specific questions, but this is also the typical case at other modeled locations outside the Bay-Delta system. Although limitations of the monitoring design can result in weak inferences about the effects of a given variable, owing to changes in uncontrolled and potentially confounding factors, quantitative models developed for locations outside the Bay-Delta systems are also dependent on monitoring data, and the monitoring data are rarely of the quality and duration of the data that are available for the Bay-Delta. In many cases, these models have been shown to be successful management tools.

Quantitative models predicting the response of key indicators like Delta or Longfin Smelt abundance may produce relatively unreliable predictions over the generally limited range of flow actions that are being considered. Calls for greater effort in modeling activities are warranted; however, the utility of these efforts will be constrained by the available data and the lack of control of key factors that change over the period of data collection. Mechanistic modeling exercises (e.g., Rose et al. 2013a) may help improve understanding of cause-effect mechanisms and help guide future research and monitoring; however, they are rarely sufficient to exclude the need for large-scale experimentation to separate confounding factors, and are not currently suitable for use as management tools.

Implementing a plan

AM plans can be classified into passive designs, where climatic variability and other factors determine the magnitude, timing, and frequency of change in a particular action (e.g. X2 in winter-spring), and active designs, where actions are systematically varied over time and/or space. To date, most of the AM in the Bay-Delta has been passive. From a learning perspective, active designs are more informative and efficient, but are harder to implement in large, complex systems like the Bay-Delta, especially when there are severe trade-offs associated with the cost of some actions and where listed species are involved. The 1995 Bay-Delta Plan appears to have been implemented as a management action rather than as an experiment. Recommendations for Reasonable and Prudent Alternatives (RPAs) associated with Biological Opinions for Delta Smelt, Chinook Salmon, and Steelhead Trout include a mix of prescriptive actions (e.g., limiting pre-spawning Delta Smelt entrainment) and AM evaluation (e.g., fall X2 effects). The flow criteria report (SWRCB 2010) distinguishes between short- and long-term AM. Short-term AM uses real-time information to guide specific real-time actions. These actions would potentially increase the likelihood of attaining a particular objective (e.g., reducing entrainment), but would not help resolve whether such actions succeed in the ultimate objective of improving population status. The flow criteria report does recognize that some flow actions should be purposefully manipulated, but no details of experimental plans are provided. Perhaps those plans would be developed during the "balancing" phase that occurs prior to implementation. To facilitate that process, a range of implementation strategies needs to be provided that varies with respect to water costs, potential benefits to pelagic and anadromous fishes, and scientific rigor.

Although there is potential for some active AM experiments in the Bay-Delta, conducting informative experiments to reduce uncertainty about the effects of outflows on system components will be very challenging. As an example, the goals for the number of medium- and high-flow years over the 12-year VAMP study were not met due to the hydrology being different than expected. As a result of not achieving enough high-flow data points, there was uncertainty about whether more flow increased survival of salmon smolts in the Delta. The fall X2 recommendation from the flow criteria report (SWRCB 2010) is another interesting example. Fall X2 is considered a Category B action, which means the benefits of this action are fairly uncertain. The recommendation is for X2 to be less than 74 km in wet years and less than 81 km in above-normal years. Presumably, the benefits of this action can only be assessed by comparing indices of Delta or Longfin Smelt abundance in wet and above-normal water years when this new X2 rule is implemented, relative to these same water years under the original X2 rule (1995/2006 Bay-Delta plan). There is no discussion in the report of whether such a design is being considered. Given that the frequency of wet and above-normal years in a decade may not be very high, and that adequate replication is required for each year type, it may take multiple decades for this experiment to play out and yield informative results. The flow criteria report also does not mention whether there will be a return to the 1995/2006 Bay-Delta fall flow regime during wet and above-normal water years as part of the experiment. If this does not occur, then the comparison will be based on control-year data collected in different decade(s) than experimental years, which increases the possibility of confounding the analysis due to long-term ecosystem changes. More explicit implementation plans are required to provide decision makers with the information they need to evaluate the likelihood of success, including the time scale of expected responses to experimental manipulations of flow. Explicit AM plans and realistic experimental designs should be a fundamental part of setting outflow objectives.

> *Decision makers are hesitant to adopt costly policies in the absence of relatively convincing model predictions that indicate they will achieve the desired objectives. However, it is very difficult to improve model predictions without implementing these policies in the first place. Thus, the rate of learning about the efficacy of alternate flow policies in the Delta will likely be very slow.*

The challenge of AM

The challenge of implementing successful AM programs is highly variable among systems (Fig. 6). In the Panel's view, the situation in the Bay-Delta is very difficult because: (1) models predicting the response of resources to management actions are relatively uncertain, (2) there are very significant conflicts between the value of consumptive water use and recovery of endangered fish populations, and likely between species (policies that benefit species like Longfin Smelt may have negative effects on Chinook Salmon and Steelhead Trout), and (3) large hydrologic variability and high consumptive water needs make implementation of informative field experiments very challenging. These problems are not unique to the Bay-Delta. Common responses to these challenges in other systems include: (1) continued study under status quo management, (2) implementation of relatively constrained and thus minimally informative experiments (limited replication, relatively small policy changes), and (3) exploration of policy options where value conflicts are reduced (e.g., predator control). Adaptive management in the Bay-Delta, as in other challenging cases, is in a Catch-22 situation. Decision makers are hesitant to adopt costly policies in the absence of relatively convincing model predictions that indicate they will achieve the desired objectives. However, it is very difficult to improve model predictions without implementing these policies in the first place. Thus, the rate of learning about the efficacy of alternate flow policies in the Delta will likely be very slow. Conducting more mechanistic studies and more synthesis efforts will help, but our expectations about the benefits of such efforts over the short term are quite modest. Given this situation, more effort on non-flow options to achieve ecosystem goals has significant merit.

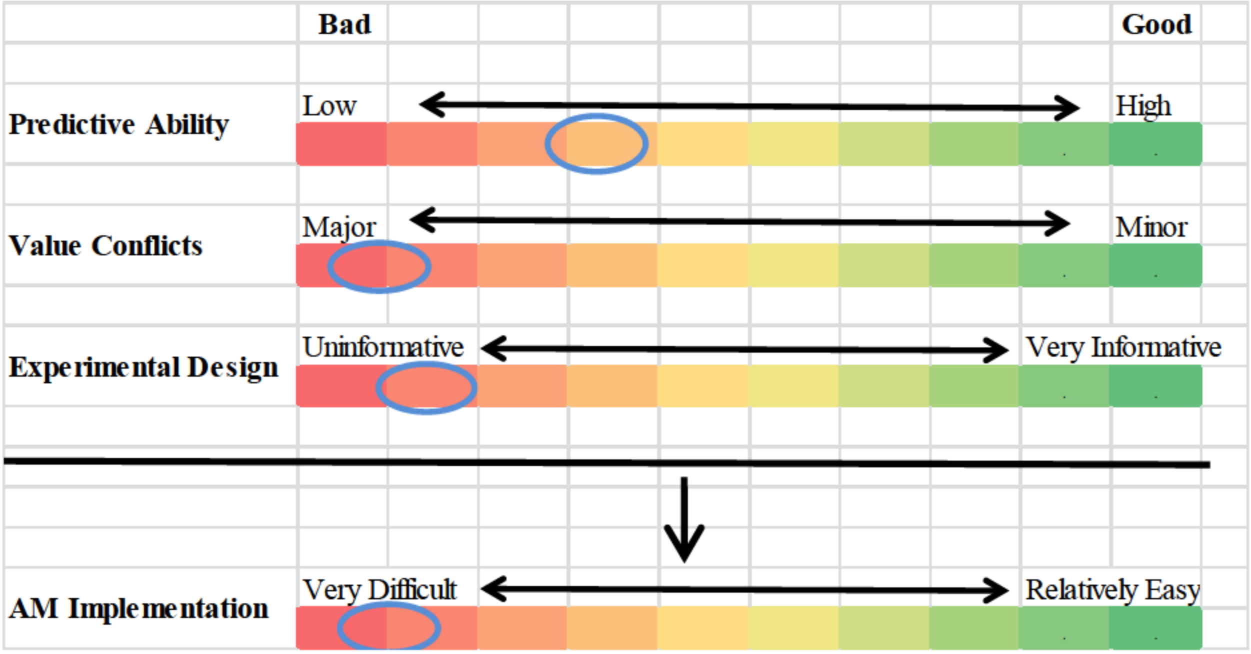

Figure 6. Implementation of Adaptive Management (AM) can span a range from very difficult to relatively easy, and depends on the ability of models to predict the response of objectives to management actions, the extent of value conflicts (e.g., water use vs. fish recovery), and the rigor of potential experimental designs (extent of temporal and spatial replication, control of confounding variables). The blue ovals represent the Panel's interpretation of the situation for the Bay-Delta AM program.

6. Question 4.

How are other factors that affect estuarine fish, estuarine fish habitat, and other ecosystem attributes likely to interact with Delta outflow requirements?

Are there tools or methods available that could help the State Water Board to better assess the interactions between flow and other factors that affect the estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

Can we reasonably expect that addressing other stressors without addressing flow will lead to specific improvements in the status of estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

Conversely, can we reasonably expect that addressing flow without addressing other stressors will lead to specific improvements in the status of estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

Interactions between outflow and estuarine processes

Freshwater outflows into estuaries support a myriad of processes that are linked to the distribution and abundance of estuarine organisms. The following paragraphs present overviews of physical, chemical, and biological processes that are closely associated with freshwater outflows into estuaries.

In the simplest terms, freshwater outflows affect water quality, water circulation, and the distribution of dissolved and particulate materials within the estuary. Water quality variables that are affected by outflow include temperature, salinity, nutrients, dissolved oxygen, organic matter, pollutants and turbidity. The interplay between turbidity (the concentration of light-attenuating materials), depth and stratification determines the locations where growth of aquatic primary producers is possible (i.e., growth of phytoplankton, benthic or otherwise attached microalgae, macroalgae, and submerged aquatic vegetation). The locations where such growth is possible may experience other physiological limitations—for example, those brought about by nutrient availability. Biomass accumulation is affected by factors like water residence time (in the case of phytoplankton) and variable levels of grazing pressure on the primary producers.

Light-attenuating materials consist of colored dissolved organic matter (CDOM) and suspended particles that are either organic (phytoplankton, plant detritus, peat) or inorganic (mineral-based sediments). In the estuaries of the world, the relative contributions of turbidity (particles) and CDOM (dissolved matter) to total water clarity (e.g., as indicated by Secchi depth) are highly variable over space and time, as are the relative contributions of living and nonliving particles to turbidity. However, in many estuaries including San Francisco Bay, light attenuation due to turbidity limits growth of primary producers (phytoplankton, benthic or attached microalgae, and submerged aquatic vegetation) (Cloern 1987). The particles (>0.45 µm) that contribute to turbidity may be transported in the water column by currents or as bedload at the bottom, and may settle or accumulate in depositional areas or at density discontinuities, only to be episodically re-suspended whenever outflows, wind and tides change the depositional characteristics of the area (Jassby et al. 1995; Turner and Millward 2002).

Chlorophyll *a* (Chl) is another commonly measured water-quality variable that is strongly affected by outflows. Chl represents phytoplankton biomass and is both a source of food for estuarine food webs and a contributor to light attenuation. Chl is an indicator of the standing-stock phytoplankton biomass, rather than of phytoplankton productivity. Chl generally represents <6% of the total carbon in a phytoplankton cell, with the actual amount depending on the cell's temperature, light, and nutrient histories (Cloern et al. 1995).

> *In the simplest terms, freshwater outflows affect water quality, water circulation, and the distribution of dissolved and particulate materials within the estuary. Mobile organisms actively orient to these environmental cues.*

Key Papers: Cloern et al. (1995), Cloern and Jassby (2012), Kimmerer et al. (2004), Lucas et al. (1999, 2009)

All of the factors above interact with outflows and, against this background, estuarine organisms exhibit behavioral responses and are subject to various ecological pressures that ultimately determine their distribution. The distributions of planktonic estuarine organisms (phytoplankton and zooplankton) may be affected directly by outflow and its effect on estuarine circulation and the dynamic location of productivity hotspots, whereas the distributions of other non-mobile (sessile) benthic organisms (e.g., bivalves, barnacles) reflect the consequences of the interactions between larval settlement and the numerous factors that contribute to their subsequent survival and growth. Mobile organisms that actively swim (nekton such as fish) or crawl across the bottom (epibenthos such as crabs) and those that actively regulate their location by rising into the water column (or sinking) to catch the preferred tidal current direction [flood vs. ebb; e.g., many mysids, amphipods, copepods (Kimmerer et al. 2014) and other important prey for juvenile fish] actively orient to environmental cues that are affected by outflows. On a species-level basis, such orientations may consist of responses to salinity, temperature, light, turbidity, olfaction (the smell of the water), prey or predator abundance, turbulence, current direction, and other factors (McEdward 1995).

Most of the above processes are intertwined because they are based, in one way or another, on water quality, estuarine circulation, and the distribution of materials in the estuary. Together, these processes form estuarine habitat, which in turn drives reactions and responses by biological resources. Outflow is thus the common denominator among the multitude of intertwined processes. In recognizing this, the Panel is unified in agreeing that the distribution, condition, or abundance of some estuarine organisms are statistically related to outflow and X2 because these two indicators reflect underlying physical and ecological processes that more directly affect the estuarine organisms. In statistical terminology, a number of important ecological factors "co-vary" with outflow and X2 and are more proximal influences on organism distribution, condition, and abundance. For example, some biotic indices may correlate with X2 because their distributions are driven by properties (for example salinity) that co-vary with X2, or because seasonal trends in X2 happen to coincide with inherent reproductive seasonality.

It is critically important for resource managers to realize that such statistical associations inherently assume unchanging, steady-state background conditions. In reality, the conditions under which regressions are developed are not guaranteed to persist through time, even if the most proximal processes remain relevant. Important processes may break down once thresholds have been crossed (i.e., excessively low growth, survival, or reproductive rates; changes to the physical configuration of the estuary and its watershed; changes in the light environment that allow or disallow primary production at depth), thereby altering the underlying basis for the original statistical relationship with outflows or X2. Moreover, many of the pelagic species that have declined are relatively short-lived, with only one or a few age-classes dominating their spawning stocks. This life history characteristic provides these species with little capacity to bridge long periods of poor environmental conditions.

The Panel unanimously agrees that the distribution, condition, and abundance of some estuarine organisms are statistically associated with outflow and X2 because these two indicators are tied to underlying physical and ecological processes that more directly affect the estuarine organisms.

Key Papers: Jassby et al. (1995), Kimmerer (2002a), Kimmerer et al. (2009, 2013)

Other things are important: Ecological regime shift

The sudden increase in *Potamocorbula* clam biomass that started in the mid-1980s, and the decline in Chl and pelagic organisms (POD) that followed it, are conspicuous ecological events. While such punctuating events are dramatic, it should be kept in mind that the Bay-Delta ecosystem had been changing continuously at all trophic levels before such conspicuous events occurred (Nichols et al. 1986, Winder and Jassby 2011, Cloern and Jassby 2012).

Decreasing turbidity of Bay and Delta waters is one gradual, long-term change that has been clearly identified. This trend, which is related to a decline in the supply of sediment to the Bay from the watershed, was discussed by presenter Hanson during the first day of presentations. Note that high sediment loadings of the past resulted primarily from hydraulic gold mining, itself a punctuating event, and that prior to this event, suspended sediment loads in San Francisco Bay may have been much lower than at present. Later in the workshop, presenter Latour concluded that changes in the abundance of various estuarine fish species were most strongly correlated with turbidity. When turbidity exceeds a certain threshold (10 NTU), it is believed to provide survival advantages to some estuarine fishes (Cyrus and Blaber 1987). As discussed above, turbidity is also a principal determinant of the light environment, and it thus affects the primary producers that support zooplankton and other organisms higher in the food web. Density stratification counteracts turbidity and enhances phytoplankton production by allowing phytoplankton cells to remain in relatively well-lit surface waters. In contrast, when the water column is deep and vertically mixed rather than stratified (for example in Delta channels in their current configuration), phytoplankton cells circulate between the well-lit surface waters (the top 1 m or less) and deeper waters where light does not penetrate. The ratio of the time spent in well-lit versus dark water directly affects growth rate, with negative growth rates (net respiration) dominating in deep, dark water columns like Carquinez Strait. Presenter Senn proposed that at certain locations, such as the south channel of Suisun Bay, outflow-induced stratification increases primary production. Stratification may also help isolate phytoplankton from benthic grazers such as *Potamocorbula*.

> *Ecosystem change in the San Francisco Bay estuary has been continuous on a decadal scale. However, this slow continual change has been punctuated by events such as the sudden increase in Potamocorbula clam biomass and the decline in chlorophyll and pelagic organisms that followed. While such punctuating events are dramatic, it should be kept in mind that continuous ecosystem change had been taking place at all trophic levels before such conspicuous events occurred.*

Key Papers: Lucas and Thompson (2012), Nichols et al. (1986, 1990), Nichols and Thompson (1985), Parchaso and Thompson (2002)

Phytoplankton production in the Sacramento-San Joaquin estuary is very low for a temperate estuary, and is generally believed to be light-limited rather than nutrient-limited (reviewed in Cloern and Jassby 2012). However, there has been concern that the increase in ammonium concentration and nitrogen-phosphorus ratio (N:P) that occurred between 1975 and 2010 is responsible for concomitant changes in the phytoplankton production and community structure. This change in ammonium concentration and N:P was discussed by presenter Hanson during the workshop, and was attributed to increasing wastewater inputs to the watershed (see also Parker et al. 2012a, c). Co-Presenter Fullerton credited a concurrent increase in small-celled primary producers to the N:P trend, citing Glibert et al. (2011). A growing body of research (Parker et al. 2012b, Dugdale et al. 2007, 2012, 2013) suggests total phytoplankton production in the San Francisco Estuary is inhibited (in cases where light is not already limiting) by increasing ammonium inputs and their effect of suppressing nitrate uptake. This hypothesis is controversial and an in-depth consideration of this possibility, along with other factors that affect estuarine phytoplankton growth, follows.

Phytoplankton growth in the estuary

The role of ammonium

The suggestion that ammonium inhibition should be considered when setting outflow objectives is based on a model linking ammonium inhibition of nitrate uptake to Delta outflow (Dugdale et al. 2013). A simple numerical model was parameterized from observations made in mesocosm experiments described in Parker et al. (2012c) and used to predict higher phytoplankton productivity and chlorophyll concentrations in the LSZ under flow conditions (600-800 m^{-3} sec^{-1}; Dugdale et al. 2013) that balanced dilution of ammonium supplied from the Sacramento Regional Wastewater Treatment Plant (SRWTP) with wash-out of the phytoplankton crop. The numerical model is a simulation of a conceptual model described in Wilkerson et al. (2006) and Dugdale et al. (2007), and hinges on the idea that ammonium inhibits phytoplankton growth. Careful examination of the evidence presented to date reveals alternative explanations for the observations supporting this hypothesis and the Panel

> *There is a large body of work indicating that ammonium concentrations greater than some threshold inhibit the uptake of nitrate by phytoplankton. Because of these nutrient utilization dynamics, high ammonium concentrations and growth on ammonium will always correlate with low phytoplankton biomass, while growth on nitrate will always correlate with high biomass accumulation, i.e., blooms. If phytoplankton growth is truncated for reasons other than nitrogen limitation (e.g., light, grazing) prior to reaching "bloom" conditions, then no nitrate will be consumed and some ammonium will remain, which has been interpreted (we believe incorrectly) as evidence that ammonium had inhibited bloom formation.*

Key Paper: Cloern and Jassby (2012)

recommends further tests of the underlying conceptual model before incorporating its predictions into management actions.

The “ammonium toxicity” paradigm, as applied to phytoplankton dynamics in northern San Francisco Bay, derives from observations primarily of the inhibition of nitrate uptake by phytoplankton in the presence of elevated ammonium concentrations. There is a large body of work, including work done in San Francisco Bay, indicating that ammonium concentrations greater than some threshold value (values in the range of 1-4 μM are commonly cited) inhibit the uptake of nitrate, especially by diatoms (cited in Wilkerson et al. 2006, Parker et al. 2012c and Dugdale et al. 2013). Once the ammonium concentration is drawn down below the threshold by phytoplankton growth, nitrate uptake begins and phytoplankton growth continues unabated until nitrate (or another limiting nutrient) is depleted. This is a physiological response that has been reported previously (Conway 1977, Dorch 1990), and it results in the sequential use of these two nitrogen sources by phytoplankton. Ammonium inhibition of nitrate uptake is thus not in question. However, because events of high phytoplankton biomass in the LSZ are less frequent now than previously (Cloern and Jassby 2012), more or less coincident with higher ammonium concentrations as a consequence of SRWTP discharges (Parker et al. 2012a), ammonium inhibition of nitrate uptake has been implied to be ammonium inhibition of phytoplankton productivity, and has been interpreted as the cause of lower phytoplankton biomass in the LSZ.

> *Because of these nutrient utilization dynamics, high ammonium concentrations and growth on ammonium will always correlate with low phytoplankton biomass accumulation, while growth on nitrate will always correlate with high biomass accumulation.*

There is an alternative explanation for these observations that considers the importance of other factors in truncating algal blooms, and the role of advection in creating “bloom-like” conditions the LSZ. The discussion presented below is based on presentations to the Panel, presentations at the CABA[13] seminar that followed our workshop, and our reading of the relevant literature, and is offered to ensure that all interpretations are considered.

Because ammonium is typically present in the LSZ at concentrations of 1-10 μM (Parker et al. 2012a), phytoplankton growth will initially be based on ammonium utilization, as shown by Parker et al. (2012c). This is illustrated in Figure 7, where a starting concentration of 6.3 μM ammonium is assumed. Once the ammonium is consumed

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[13] Delta Science Program/UC Davis Center for Aquatic Biology & Aquaculture (CABA) seminar: Lower Foodweb Dynamics in California’s Bay-Delta Ecosystem, February 18, 2014

(day 6 in Fig. 7) or reduced to below the threshold for inhibition of nitrate uptake, growth continues on nitrate. Because of these nutrient utilization dynamics, high ammonium concentrations and growth on ammonium will always correlate with low phytoplankton biomass accumulation, while growth on nitrate will always correlate with high biomass accumulation. Thus, any “bloom” will have the appearance of “requiring” nitrate because all of the ammonium will be consumed while increasing phytoplankton biomass to the beginning of the “bloom” stage. Subsequent phytoplankton growth will then depend on the only remaining source of fixed N, which in this case is nitrate, and growth on nitrate will appear to have “caused” the bloom. Nitrate consumption is, in fact, simply a consequence of the bloom. Furthermore, if phytoplankton growth is truncated for reasons other than nitrogen limitation (e.g., light, grazing) prior to reaching “bloom” conditions, then no nitrate will be consumed and there may be some ammonium remaining, which could be interpreted (we believe incorrectly) as evidence that ammonium had inhibited bloom formation.

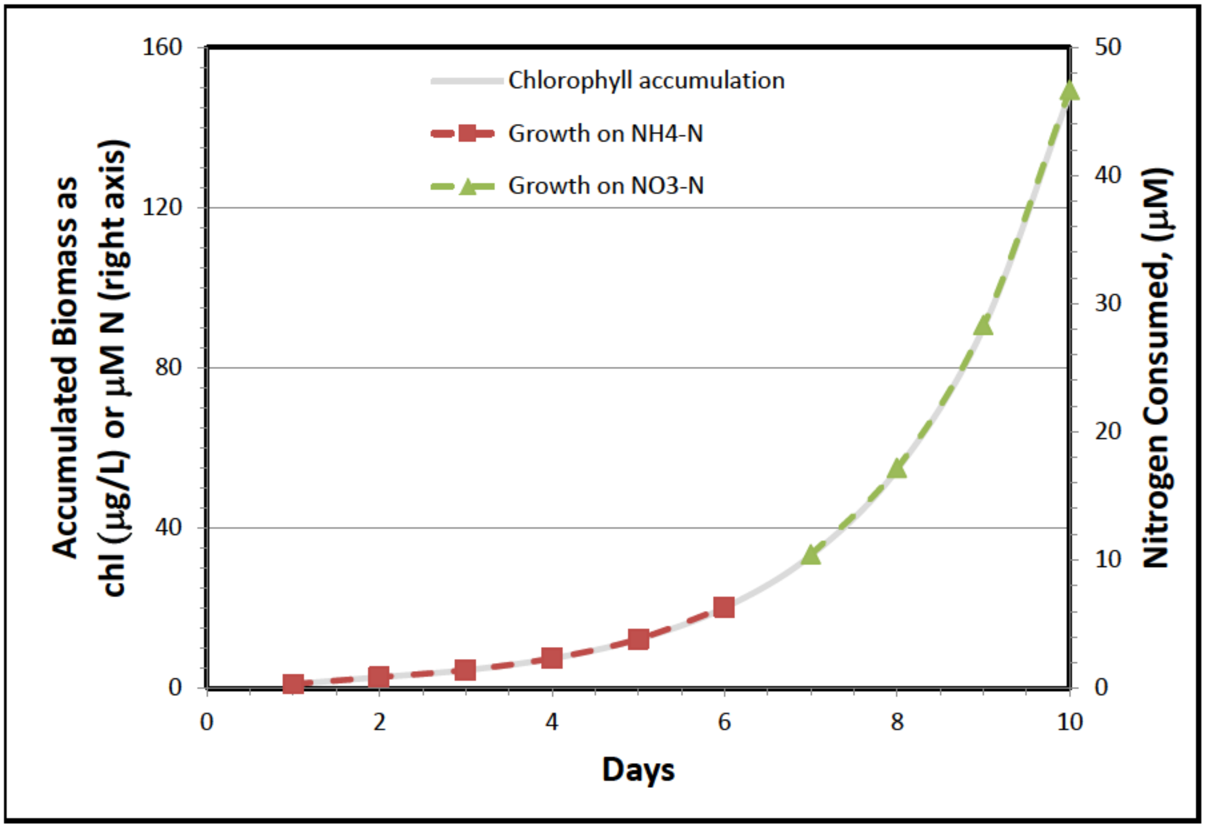

Figure 7. A simple model of phytoplankton growth dependent on successive utilization of ammonium and then nitrate. Net population growth rate is taken to be exponential at 0.5 d^{-1} from a starting biomass of 1 µg Chl/L and growth rate is never considered to be nitrogen limited. Accumulated biomass is converted to N equivalents assuming a C:chl ratio of 25 and a C:N ratio of 6.625. Phytoplankton growth is initially dependent on ammonium (assumed to be 6.3 µM) and then switches to nitrate once [NH4]=0 on day 6.

As mentioned above, ammonium inhibition of nitrate uptake has been interpreted as ammonium inhibition of phytoplankton growth. A critical question that has not been adequately addressed is whether or not phytoplankton grow "better" (faster, more efficiently) on nitrate than on ammonium. Would elevated ammonium concentrations (comparable to the concentrations of ammonium plus nitrate currently found in the Bay) support a bloom comparable in magnitude to that supported by an equivalent amount of nitrate, assuming bloom formation was not truncated by other factors? Related to this question is the possibility that phytoplankton community composition might change in response to growth on ammonium versus nitrate (all other things being equal), which might have implications for trophic transfers.

The literature on growth efficiencies presented to the Panel references higher C:N incorporation rates by phytoplankton growing on nitrate than ammonium (Parker et al. 2012c). The evidence presented to support this difference in growth efficiencies is one unreplicated experiment conducted in Delaware Estuary (Parker 2004) that concluded that the excess C was incorporated into dissolved organic matter, rather than into particulate biomass. A related set of San Francisco Bay experiments (Parker et al. 2012c) found that carbon and nitrogen incorporation rates were lower in mesocosms containing ammonium-rich water from Suisun Bay (Parker et al. 2012c); however, the ratios of C:N uptake reported were similar to those for mesocosms filled with water from other regions of the bay, and the statistical significances of the differences were not tested. Lower uptake of both C and N in Suisun Bay samples may have resulted from other causes including salinity stress or unknown toxic compounds. Field data presented in Wilkerson et al. (2006) indicate lower ammonium uptake by cells >5 um in diameter; however, nitrate uptake was independent of cell size. These observations, which were used to infer preferential growth of large cells on nitrate, are consistent with the sequential utilization of ammonia and nitrate discussed above: low, non-bloom N uptake is based on ammonium and is biased towards smaller cells for reasons that may not be related to nitrogen speciation (e.g., grazing, light limitation). This does not imply that nitrate causes blooms of large cells and the data presented suggest no difference in nitrate uptake between large and small cells under bloom versus non-bloom conditions (though this was not tested for statistical significance). The geochemical model described in Dugdale et al. (2013) incorporates an "acceleration factor" into the standard Michaelis-Menton formulation for nitrate uptake that increases nitrate uptake rates as a function of nitrate concentration, implying faster growth on higher nitrate. This factor was derived through a sensitivity analysis to fit model output to mesocosm data. Before policy decisions are made that assume ammonium inhibition is occurring, the Panel recommends that more information be obtained on whether the growth rate of phytoplankton is lower on ammonium or nitrate at the concentrations typically encountered in San Francisco Bay. These experiments should also examine selection for phytoplankton community composition by these two different N sources.

Some of the material presented to the Panel suggested high ammonium concentrations might be toxic to phytoplankton. Relatively poor photosynthetic performance of phytoplankton in mesocosms using Suisun Bay water was noted by Parker et al. (2012c) and attributed to ammonium toxicity; however, this could have resulted from sampling phytoplankton that had recently been advected into the estuary from fresher water, resulting in salinity-related stress. This seems a more likely explanation since a recent review (Collos and Harrison 2014) concludes that ammonium is only toxic to phytoplankton at concentrations much higher than those found in Suisun Bay, or even in the Sacramento River immediately downstream of the SRWTP. The apparent phytoplankton "blooms" observed in the LSZ in the studies cited above may well be the result of Eulerian sampling of an advecting chlorophyll field, influenced by changes in the flow regime through the Delta. Previous work has shown the Suisun Bay channels to be a net sink for phytoplankton due to light limitation and benthic grazing (Cloern et al. 1983, Nichols 1985, Nichols et al. 1990), with biomass imported from upstream playing a major role in determining chlorophyll concentrations within the LSZ [Jassby et al. 1993, Jassby and Powell 1994, Canuel and Cloern 1996, Jassby et al. 1996, Kimmerer presentation (CABA series) and Lucas presentation to the Panel and in the CABA seminar]. Thus, the occasional "blooms" seen in the LSZ under higher flow conditions may well be the result of advection of phytoplankton from the Delta into the LSZ, and not from higher growth rates in the LSZ, regardless of the cause, including the release of putative ammonium toxicity.

In support of this last point, the Panel recommends that Bay and Delta hydrodynamic models should be reviewed to determine if they can be modified to determine how advection of phytoplankton into the LSZ from the freshwater Delta (and from seaward) is affected by Delta outflow. If feasible, these models should be coupled to a biological model to determine how circulation and advection affect grazing losses of phytoplankton to benthic filter feeders in the Delta and LSZ.

Other factors potentially affecting species dominance

> *There has been a long-term change in the composition of the phytoplankton community, with a general trend toward smaller-celled phytoplankton. Factors that appear to affect the dominance of different types of phytoplankton include periods of reduced outflow (long water residence times) with decreased turbidity and higher water temperatures. These changes in the structure of the phytoplankton community are also consistent with increased benthic grazing in the LSZ and the Delta. Decreased turbulence affords an advantage to buoyant or positively phototactic cells by increasing the average amount of light they receive and by decreasing exposure to benthic grazers.*

Key Paper: Lucas and Thompson (2012)

An increase in the frequency of blooms of noxious cyanobacteria has been attributed to the combination of periods of reduced outflow (long water residence times, "water age") with decreased turbidity and higher water temperatures (Lehman et al. 2013). These conditions may also result in decreased turbulence, affording an advantage to buoyant cells. Presenter Senn provided figures that clearly illustrated long-term changes in the estuarine phytoplankton community, with a general trend toward smaller-celled phytoplankton. In aquatic ecosystems, phytoplankton cell size and type determine the types of animals that consume phytoplankton. Small-celled primary producers are captured less efficiently by typical crustacean zooplankton, resulting in lower trophic transfer efficiency and increased recycling of organic matter by bacteria because extra steps are needed to link this production to higher trophic levels, particularly in pelagic food webs (Azam et al. 1983). A shift in phytoplankton community dominance to smaller cells may also be a product of increased benthic grazing, as larger cells with higher sinking rates (especially diatoms) will be preferentially encountered and removed by filter feeding benthos. Conversely, buoyant cells like Microcystis or positively phototactic cells like dinoflagellates are less likely to be eaten by benthic grazers. Changes in the relative abundance of lower-trophic-level consumers (e.g. *Eurytemora* to *Limnoithona*) in the LSZ reflect the trends in phytoplankton cell size, supporting the idea that the efficiency of the pelagic food web has decreased over time. Specifically, the zooplankton community has switched from dominance by consumers of large-celled phytoplankton (e.g., *Eurytemora affinis*), with a concomitant decrease in mysids and other important fish prey, to dominance by smaller consumers that feed on ciliates and rotifers (e.g., *Limnoithona*).

Changes in the relative abundance of lower-trophic-level consumers (e.g. Eurytemora to Limnoithona) in the LSZ reflect the trends in phytoplankton cell size, supporting the idea that the efficiency of the pelagic food web has decreased over time.

Other tools and approaches

This section addresses the potential to apply new investigative tools to understanding the complex interactions between environmental factors and ecosystem conditions in the Bay-Delta. It stems from a realization that inferences based on correlation analyses, which so far have be the main tool applied to understanding the relationships between resources and processes in the Bay-Delta ecosystem, are limited because they do not inherently rely on knowing cause and effect. This is especially true in a system where so many changes have occurred and responses to change have covaried over the same, relatively short period. The Bay-Delta ecosystem is complex, and it is highly likely that the changes to the resources that are the focus of human interest stem from a multiplicity of drivers exerting different weightings under differing sets of environmental conditions. We need tools—basically an experimental approach—to try to assess the weighting of each of the main drivers to the response of resources of interest. As discussed under adaptive management, it is unlikely that we will be able to test all resources individually, and it is equally unlikely that we will be able to test all drivers experimentally.

Inferences based on correlation analyses, which so far have be the main tool applied to understanding the relationships between resources and processes in the Bay-Delta ecosystem, are limited because they do not inherently prove cause and effect. This is especially true in a system where so many changes have occurred, and responses to change have covaried over the same, relatively short period.

Key Papers: Nichols et al. (1986), Cloern and Jassby (2012)

More comparisons with estuaries around the world

One of the problems with ecosystem-level experiments is that they usually lack sufficient replication; there is only one system of interest and so it is difficult to set up controlled and replicated experimental protocols. This is especially true of estuaries and other coastal ecosystems. While most of the drivers are common to all estuaries, combinations of climate, flow regimes, tidal signature, geomorphology and the history of human intervention are such that it is nearly impossible to find multiple estuaries that are sufficiently similar to use as replicates.

Nevertheless, comparison between estuarine systems, if it is done rigorously, can help to identify broad patterns of the effects of certain drivers on certain estuarine process. Good examples of these syntheses are Cloern (1987), Cloern (2001) and Cloern and Jassby (2012). Applying conceptual models derived from these syntheses to management must be done cautiously, though, because specific interactions of the drivers with characteristics of a specific estuary influence the strength and possibly direction of the ecosystem response. Another example is provided by Burghart et al. (2013), who assembled data from eight Florida estuaries to create a water-clarity gradient that was interpreted using space-for-time substitution, and observed an abrupt (strongly nonlinear) decrease in the abundance of plankton-oriented species as estuarine waters became clearer. The abrupt decrease in plankton-oriented species was coupled with an equally abrupt increase in the abundance of benthic species. The operating process was proposed to be a shift in the partitioning of primary production between the plankton and benthos, as driven by the light environment (Radabaugh and Peebles 2012). By the time the Burghart et al. (2013) comparison was conducted, the estuaries in the region had already experienced invasions by two exotic bivalves, the clam *Corbicula fluminea* in oligohaline habitats and the Asian green mussel *Perna viridis* in open bay waters. Given that the Sacramento-San Joaquin estuary has undergone a decadal-scale decrease in turbidity, the fundamental ecological effects of a changing light environment should be further explored.

Syntheses based on estuarine comparisons are likely to be informative.

Given that the Sacramento-San Joaquin estuary has undergone a decadal-scale decrease in turbidity, the fundamental ecological effects of a changing light environment should be further explored.

Key papers: Cloern and Jassby (2012), Burghart et al. (2013)

Comparisons should also include comparisons of management approaches. The Southwest Florida Water Management District has been using a management approach for unimpounded rivers that limits withdrawals to a percentage of streamflow at the time of withdrawal (Flannery et al. 2002). The natural flow regime of a river is the baseline for identifying the effects of increased withdrawals; various streamflow parameters are then evaluated to determine changes in river flow regimes. This approach to water supply planning and management is designed to maintain the physical structure and ecological characteristics of unimpounded rivers. Relationships between freshwater inflow and estuarine characteristics are then examined to determine withdrawal limits that will not result in negative environmental impacts. This percent-of-flow approach was supported by initial findings that indicate a curvilinear response of isohaline locations to freshwater inflow and the influence of inflow on catch-per-unit-effort for a number of key organisms.

New types of ecosystem modeling

Hydrodynamic models of the Bay-Delta are approaching a level of sophistication where they can accurately predict a range of ecologically important water properties (currents, net flow, residence time or "water age," particle movements, dispersion of dissolved and particulate materials). We are thus poised to begin integrating conceptual models of biological processes with hydrodynamic models. Specific efforts in this direction have provided useful tests of conceptual models—for example, Kimmerer's work on the role of vertical migration of zooplankton in a tidally oscillating environment in maintaining their populations in the LSZ (Kimmerer CABA presentation, Kimmerer et al. 2014) and the Kimmerer et al. comparison of X2 and habitat suitability indices (cited above). During the workshop, Lisa Lucas presented a simple conceptual model of the effect of residence time ("water mass age") on phytoplankton dynamics in an environment where benthic grazing is significant. This model (Lucas and Thompson 2012) was combined with field data on clam distributions and static estimates of residence time to identify areas in the Bay-Delta that support net positive growth of phytoplankton. A next step is to more fully integrate this module into a hydrodynamic model that captures the temporal variability in vertical mixing and residence time under different flow regimes. Eventually this "***Delta ecosystem model***" should capture clam population dynamics in order to model grazing pressure. Similar modules that capture the interactions between nutrients and phytoplankton growth could be added to assess the relative contribution of these factors to phytoplankton production. The results could be used to test assumptions about the strengths of variables and formulations used in the model, to predict delivery of phytoplankton carbon to the LSZ and to test hypotheses about the effects of various management actions (increased flow, decreased flow, the value of stratification versus flow, etc.) on Delta and LSZ function.

New monitoring technologies

New sensors have been developed that allow long-term monitoring of various physicochemical variables on a nearly continuous basis. These sensor packages could be added to existing continuous monitoring packages to provide more highly resolved data on variables such as pH (an important physiological and chemical variable, especially in the fresher end of the LSZ) or nutrient concentrations. Additional tools for monitoring biota are also becoming available (FlowCam, use of ADCP to assess fish movements, etc.); these could be added to established monitoring sites. It may be desirable to expand the distribution of monitoring packages to capture more examples of the different habitats in the Bay-Delta. These data can be used, for example, to drive or verify the Delta ecosystem model described above.

Benthic indicators

Analyses of benthic invertebrate communities have been widely used as bioindicators in assessment and monitoring studies worldwide (Dauer 1993). There are several reasons why these organisms are good indicators of environmental stress or change. Because of gravity, particulate materials tend to end up in bottom sediments. Materials from watersheds and freshwater are transported downstream to the estuarine and coastal-ocean seafloor. Algae, vascular plants, and smaller planktonic and non-planktonic animals tend to contribute to the detrital food chain after they die, where their collective biomass is used by benthos. Transported pollutants are usually bound to organic matrices (Long et al. 1995), and therefore benthic organisms have elevated exposure to pollutants through their niche (food) and habitat (benthic living spaces). Benthos are relatively long-lived and tend to be sessile (limited or no ability to move around), and so they integrate the effects of pollutants over long temporal and spatial scales. Benthic invertebrates (primarily worms, bivalves, and crustaceans) are sensitive to change in environmental conditions—and pollutants in particular—thus, loss of biodiversity is an excellent indicator of environmental stress. Bioturbation and irrigation of sediments by burrowing benthos affects the mobilization and burial of foreign (xenobiotic) materials. Finally, because they are sessile and simply can't swim away, benthos must tolerate everything that happens in the overlying water column. In fact, benthic suspension- and filter-feeders sample the overlying water continuously between temporally structured sampling events, and thus integrate environmental effects over the long-term, including periods between sampling events.

There are also ecological models that provide a scientific basis for interpreting the effects of ecological disturbances, whether they are natural or anthropogenic in origin. These models include single species, community level, and statistical models. One of the most important concepts is the succession model proposed by Rhoads et al. (1978) and Pearson and Rosenberg (1978). They applied theories of ecological succession and its relation to productivity and community structure to suggest ways to assess risk due to dredge-spoil disposal and organic waste enrichment. The underlying concept in both papers is that distance from a source is analogous to time since disturbance. The idea is that succession after a natural disturbance proceeds in a predictable way over a given time period, thus successional stages will be distributed in an analogous way with distance from a source of pollution. In both cases, disturbed communities have pioneer species (r-selected life-history strategies among small, surface-dwelling infauna that are numerous but have low diversity) and undisturbed communities have climax species (k-selected life-history strategies among large, deeper-dwelling infauna that have low abundances and high diversity). One important prediction of this theory is that un-perturbed sediments will have a diverse assemblage of deeper-dwelling organisms than a polluted or disturbed environment. Thus, we have a scientific justification for using community structure and biological diversity as an endpoint or biology-based metric.

A persistent concern is that benthos control plankton dynamics through their grazing activities, and that these dynamics are disrupted by invasive species (Nichols et al. 1985, 1986). For example, reduced phytoplankton biomass during periods of persistently low river flow and high salinity results from increased grazing losses to introduced benthic suspension feeders (e.g., *Mya* and *Macoma* clams) that are normally excluded from-this region by winter freshets. In light-limited environments without bivalves, shallow, hydrodynamically "slow-water" habitats generally have greater phytoplankton biomass and productivity than deeper, "fast-water" habitats (Lucas and Thompson 2012). But shallower, slower environments can have less phytoplankton biomass than deeper, faster ones if benthic grazing is strong. The finding that benthos control the overlying water column when water residence time is low (fast water) is contrary to findings in more saline estuaries with a smaller LSZ along the Texas coast, because increased flow increases the feeding and productivity of all suspension feeding benthos, not just bivalves (Montagna and Li 2010; Kim and Montagna 2009, 2012). The difference occurs because LSZ salinity ranges 0.5–6 in the Delta outflow area, but can range 5–15 in other estuaries where river flow rates are much lower and residence times are longer, such as in many Texas estuaries. In fact, slow-moving water in Texas promotes growth of deposit feeders, not suspension feeders. This is particularly noteworthy since *Potamocorbula* clams are suspension feeders that are living in an estuary where reduced outflows (slower water) are viewed as a stressor. The key variable is water residence time (or "water age") (Sheldon and Alber 2002, 2006). A good example of the importance of the variability in water age in different parts of an estuary is provided by Meyers and Luther (2008), who show that the residence times in different grid cells in Tampa Bay, FL can vary spatially from a few days to 90 days.

Benthic organisms do not seem to have received the same level of scrutiny as pelagic organisms, in the Delta in particular, with the possible exception of clam abundance because of the perceived significance of clams to benthic grazing. Other benthos may be important as food resources and as contributors to important ecosystem processes ranging from bioturbation and nutrient regeneration to important predators. Decapod abundance and distribution are examples of potential predators on the clams, and the Panel was presented little information on epibenthos in general. Because decapods are arthropods, body burdens of pesticides (which are easily collected and integrate over fairly long time scales and fairly small spatial scales) may serve as a means of assessing the effect of toxins on zooplankton, and thus contribute directly to a better understanding of the factors responsible for organism declines in the estuary. The same is true for barnacles and other filter feeding organisms in the LSZ. Stable isotopes are another tool that can be used to trace pathways and fate of carbon and nitrogen through the ecosystem. In particular, identifying trophic links with clams is very important. The Panel is surprised not to have been presented with this type of information, given its common use in other systems.

More studies of <u>Potamocorbula</u> (and <u>Corbicula</u>)

Potamocorbula is clearly an important organism in the ecosystem. We know relatively little about its ecophysiological characteristics other than distribution and feeding rates. For example, studies conducted in the 1990s showed that the South San Francisco Bay population of *Potamocorbula* was derived from the Suisun Bay introduction. There is evidence now (Thompson's CABA presentation) that these populations may have diverged in some ecologically important traits. This can be tested. Other important population biology parameters of *Potamocorbula* that would be needed for a food-web model coupled to a hydrodynamic model (proposed Delta ecosystem model) is information on larval dispersal (verified from field data), duration of the larval stage, temperature and salinity tolerances and growth response of the larvae, food preferences of the larvae, larval behavior (vertical migration, for example), and ideally, the susceptibility of the larvae to predation.

Fish condition and food-web analysis

Funding should be provided to perform routine analyses of fish samples other than simply counts and sizes of fish collected by monitoring programs. While there have been some short-term examinations of specific stressors or condition, routine sampling is largely absent. Parameters of interest would be expansion of the efforts to collect data for fish condition indices, reproductive states, and toxin loads (body burdens). The role of pesticides in the collapse of the Bay-Delta ecosystem has received little attention apart from the POD studies (Scholz et al. 2012); routine measurements of the body burdens of key pesticides could be informative (pesticide use in the Bay-Delta watershed has changed with time). It may be necessary to couple pesticide surveys with laboratory experiments to calibrate physiological and reproductive responses to body burdens.

As with clams, multiple stable isotope surveys of fish and lower trophic levels would be useful for identifying the dominant biomass pathways that support fish at different life history stages. These have been conducted in the past, but more advanced methods such as compound-specific isotope analysis (e.g., analysis of source vs. trophic amino acids) can help overcome confounded interpretations that arise from stable-isotope analysis of bulk tissues. C:N ratios are a common byproduct of stable isotope analysis that can be used as a proxy for condition (as lipid content).

During investigations of biomass pathways, specific consideration needs to be given to the possibility that benthic microalgae are becoming more important contributors to the estuarine food web as the estuarine water becomes clearer with time. Although estimates by Jassby and Cloern (2000) are sometimes cited as support for the idea that benthic microalgae are not important in the Sacramento-San Joaquin estuary, Jassby and Cloern were primarily addressing food-web drivers in the interior Delta (including tidal fresh water) rather than the estuary in the general vicinity of Suisun Bay. Moreover, Jassby and

Cloern only considered benthic microalgal production on mud flats that were exposed to air during the tidal cycle—benthic microalgal production below the low-tide line was not considered. In contrast, Jassby et al. (1993) considered seafloor surfaces with >1% surface light to be benthic microalgal habitat. In North San Francisco Bay for the year 1980, they estimated the benthic microalgal contribution to total autochthonous primary production to be 28% versus 72% for phytoplankton (Table 3 in Jassby et al. 1993). After more than 30 years of decreasing turbidity, it seems probable that these proportions have shifted in favor of benthic and other attached microalgae, and that this shift would be reflected in the biomass pathways of the Sacramento-San Joaquin estuary.

Although some of the effort for the above analyses could be accommodated by IEP monitoring crews, it appears from statements made to the Panel that the time required of the monitoring crews to implement additional studies is constrained by the demands of maintaining the ongoing monitoring programs in their current configuration. Thus, in order to obtain more information from the monitoring program, new positions need to be provided to hire persons with the requisite expertise to make new measurements. In addition, time needs to be made available for knowledgeable senior personnel to commit to activities other than those constrained by the reporting requirements of the monitoring program, including time to conduct *ad hoc* sampling, gear testing, method development, data analysis, and general data-product development.

Molecular techniques to examine population dynamics.

New techniques derived from the fusion of molecular biology with environmental sciences are bring brought to bear on Bay-Delta problems to some extent, but these approaches could be used more widely to answer a number of important ecological questions. For example, the Panel was surprised to learn that despite the importance of Delta Smelt in the Bay-Delta, important aspects of their reproductive biology remain obscure. It may be more difficult to determine this now that populations are so low because detecting eggs and larvae will be difficult, but should not be impossible. Cryptic larvae and eggs captured in plankton tows can be readily identified using molecular genetic techniques. The samples may already exist to do this, depending on how plankton tows taken during monitoring exercises are preserved, though it may be more productive to set up a dedicated monitoring plan tied to tracking populations of potential spawners. Similar tools could help identify the distribution of cryptic stages of other important organisms, for example clam larvae.

7. Question 5.

How should Delta outflow be measured and managed to better reflect the flows necessary to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

To what extent does managing winter-spring outflow by X2 reflect the flows necessary to protect estuarine fish? Are there other approaches to managing winter-spring outflow that could improve our ability to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

How should summer-fall outflow be measured and managed to better reflect the flows necessary to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes? Are there other approaches to managing summer-fall outflow that could improve our ability to protect estuarine fish, estuarine fish habitat, and other important ecosystem attributes?

There is very strong (even unequivocal) evidence that specifying outflow requirements and objectives specific to seasons (specific months) is a rational and scientifically justified approach. As summarized in SWRCB (2010 – Development of Flow Criteria), there is solid evidence that high outflows during various combinations of winter-spring months benefit a variety of species. Table 2 of that report lists the species, life stage, mechanism, and the seasons when flows are most important. High winter-spring flows into the Bay-Delta (low X2) have been shown or argued to act as cues for fish spawning migrations, to improve reproductive success, and to increase survival of juvenile anadromous species migrating seaward. High winter-spring outflows also benefit a variety of species through early-life-stage dispersal, access to floodplain habitat, and reduced entrainment.

While outflow objectives must be considered for the entire year, the evidence for specifying specific targets for months during the summer, and especially the fall (e.g., Delta Smelt habitat and X2) is more uncertain and is highly controversial (NRC 2010). We agree with the statements in SWRCB (2010) and made by others at the meeting that summer-fall outflow objectives should be developed with an AM approach. However, we are not recommending that AM replace outflow-based objectives. Given the current legal situation, this seems to be the most viable pathway forward. It is not ideal that it may require legal proceedings to force new studies because the success of such collaborations is based on trust. Also, we caution that while Delta Smelt are very important, other species are affected by outflow during the summer and fall seasons, and they should be included in the AM studies and analyses. These studies offer an opportunity for developing a sound scientific basis for managing summer-fall outflows into the future, and must be funded and peer-reviewed at sufficient levels to ensure the results are of sufficient scientific credibility and generality to be effective in resolving some of the outflow issues.

Provided certain conditions are met, managing outflows, whether directly or via X2, provides a coarse level of protection to estuarine fish and ecosystem health. One condition that needs to be met is the acknowledgment that outflow is a highly aggregated measure and that the same outflow can result in different endpoints: quantity and quality of habitat for different species. That is, X2 and outflow are useful, but incorporate the effects of many factors and subsume a great deal of variability and uncertainty. This makes outflow a good indicator of general conditions in the estuary, but not always with fine enough resolution to determine precisely described conditions for individual species or ecosystem traits. Basically, using outflow and X2 can help manage some species and general aspects of the ecosystem, but does so with considerable uncertainty about the response of individual species in a specific year.

A second condition that must be met is that evaluation of species and ecosystem targets (i.e., success or not, as in the standard AM procedure) should occur using multiple years (either with data or model projections), rather than requiring the targets be met based on field data measurements the first and every year thereafter. Despite the extensive monitoring that occurs in the Delta—which is impressive and must continue—quantifying the responses of populations (not just abundance indices) to changes in outflow has a substantial degree of uncertainty. This uncertainty arises from the aggregate nature of outflow as a measure, the complicated interactions of outflow with population dynamics, and from the inherent variability of the system relative to a finite frequency of sampling at a finite number of locations (i.e., the prescribed survey designs).

When outflow is used to protect or improve estuarine health, expectations should be realistic: (1) habitat use by different organisms is seasonal, thus the same amount of outflow will have different effects at different times of year, (2) many populations require more than one year to respond, (3) abundance indices may not accurately reflect the true population responses, and (4) room for adaptive management should exist within prescribed outflow management practices.

A third condition required for successful use of outflow is to allow for some deviations from meeting the individual species and ecosystem targets, which then relates to managing expectations. The ecosystem is dynamic and fish populations are notorious for responding in non-linear and sometimes counterintuitive ways to changes in their environment. As previously discussed, the variance in resource abundance indices explained by X2 or outflow varies greatly across species. Species will respond with differing sensitivity and magnitudes to changes in outflow. Even under ideal conditions, there are also trade-offs among species and ecosystem traits in the responses to different outflow values, and changes in an abundance index does not mean the same changes should or will occur in the population abundance. Successful use of aggregate measures, such as X2, involves management of expectations on the speed and magnitude of the responses of individual species and the system as a whole, how well the monitoring data can be expected to detect these responses, and how clearly responses can be attributed to management actions versus other factors. Lack of success for some species in the short-term may indicate a true non-benefit, or may result in an effective action being falsely dismissed as ineffective, when it is in fact effective but not for all species in every year.

A fourth condition for the successful use of outflow to protect species and ensure ecosystem health is to find the appropriate balance between flexibility and prescriptiveness for specifying outflow objectives. A high degree of prescriptiveness provides a very clear way to determine compliance or not (i.e., tractability). However, a high degree of prescriptiveness also requires data and information that has a relatively high level of certainty; otherwise, inefficiencies can be introduced (e.g., small responses costing a lot of water) by some of the required actions. Further, highly prescriptive rules can lead to very unnatural transitions of ecosystem conditions (e.g., step function changes in outflow and salinity distributions). This might be addressed by prescriptively tying outflow to some index of inflow from the watershed to ensure flexibility, though tractability might suffer. The right balance between flexibility and prescriptiveness should result in cost-effective actions that protect species and the ecosystem without losing the tractability and accountability associated with highly specific and rigidly defined outflow objectives. AM offers one approach for generating the information needed to rationally balance prescriptiveness and flexibility.

A fifth condition is the clear expression of both the magnitude of change in outflow and the resulting expected change in species or ecosystem indicators. The derivation of many of the X2-abundance relationships involved log transformations on the Y and/or X axis (Kimmerer 2002a, Kimmerer et al. 2009) and on top of this, X2 is non-linearly related to outflow (Kimmerer et al. 2013). A clear statement of the expected return for the changed outflow (benefit-cost assessment) is needed in order for outflow to be used effectively in ecosystem management. The Panel is arguing for transparency in expressing changes in outflow and changes in species indices so that everyone is using the same, intuitively understandable information. There are also situations of a large benefit-cost (e.g., often near the origin of the response curve), where one gets a large change in the index for a relatively small increase in X2 or outflow, or for changes in outflow for certain species in certain water-year types.

Even when all of these conditions are met, the abundance relationships with outflow (or X2) are correlations, sometimes quite strong and robust, but they are still correlations. In the case of using outflow in the Delta ecosystem, as in many other ecosystems, correlations can be misunderstood and over-interpreted because they are specific to a set of conditions and they do not provide information on causality. It is easy to criticize correlations; however, the X2 (or outflow)-abundance correlations documented for some species in the Delta clearly reflect some irrefutable level of dependency between outflow and species indicators. In general, correlations are associated with a domain of observations under a set of conditions; large changes in the ecosystem (e.g., due to effects of an introduced species) can change those conditions and render formerly strong correlations weak and predictions based on earlier conditions highly uncertain or invalid under the new conditions. Also, correlations can appear to be simple and direct but often reflect many steps in a complicated set of processes and mechanisms. An example is the conceptual model relating outflow to the population dynamics of Longfin Smelt (Figures 3-5, Rosenfield 2010); outflow appears in many places in the conceptual model and thus there are many pathways that relate outflow to environmental conditions and biological processes that ultimately combine to affect population abundance and spatial distribution. Longfin Smelt is typical and is not cited here as an extreme example, as conceptual models relating stage-specific population abundance to outflow for many species (e.g., Delta Smelt, IEP MAST 2013) would likely share similar complexity of environmental conditions and biological processes. Without a very long data record for field observations sufficient to tease out effects of multiple factors (which is impractical) and a strong basis of experiments and process-level studies (not just monitoring of abundance indices), correlation-based indicators have inherent uncertainty that can result in projections with various levels of inaccuracy or even unexpected responses.

Use of outflow objectives on a monthly to seasonal basis does not capture all of the desired dynamics that ensure protection of species and ecosystem health.

Use of outflow objectives on a monthly to seasonal basis does not capture all of the desired dynamics that ensure protection of species and ecosystem health. Two examples are turbidity and episodic flow-related events. For example, Delta Smelt show elevated concentrations in turbid water, and their spawning migration is correlated with the first flush events during December to February (IEP MAST 2013). Such dynamics can be codified into objectives, but need to be dealt with differently than monthly-to-seasonal outflow objectives. For outflow-based management to be protective, it requires the inclusion of additional non-outflow objectives.

The calculation and interpretation of unimpaired or more natural flow regimes should be revisited to establish an agreed-upon set of benchmark flows. The use of some version of unimpaired flows to set Delta outflow objectives is useful for establishing more natural outflow conditions and to ensure effective protection of species. However, without widespread agreement concerning how these benchmark flows are to be calculated and interpreted, they simply add more confusion to the discussions. A hydrologic frame of reference for outflows is absolutely critical to specifying outflow objectives that are rational and effective.

Expressing outflow (and X2) in terms relative to conditions in key habitat features, such as the LSZ, Suisun Marsh, and the intermittently flooded habitat at the intersection with the shoreline and with conditions in specific sub-embayments is helpful. In a sense, not only expressing X2 in kilometers, but also having several axes that show habitat volumes or areas and habitat types or features helps to provide context for flow or X2 objectives. Those who are very familiar with the Bay-Delta system and hydrodynamics already know this, as evidenced during the presentations and conversations, but making it explicit and part of the documentation of outflows and X2 objectives would help others less familiar with Bay hydrodynamics and would keep the discussions focused on the resources the objectives are meant to protect, and would simplify discussions by facilitating the use of a common terminology.

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