The State of Bay–Delta Science 2016: An Introduction

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INTRODUCTION

The State of Bay–Delta Science (SBDS) is a synthesis of the current scientific understanding of the Delta, emphasizing progress made on key research questions and remaining knowledge gaps. The first edition of SBDS (Healey et al. 2008) provided a system-wide baseline for the state of scientific knowledge of the system, and a reframing of the interaction between science and policy. In this new edition, various authors summarize the state of science in relation to a dozen issues in the Delta. Paper topics were chosen after surveying senior scientists and managers in the Delta to identify the most topical science issues. This introductory essay is intended to set the stage for the topic-focused papers that follow by providing a brief description of the Delta and its ecology, linking issues to individual papers where appropriate. In addition, a previously published paper (Luoma et al. 2015) describes the challenges facing water and environmental managers in the Delta and a final paper (Healey et al., submitted) will synthesize the main advances in scientific understanding over the past decade and their policy implications. The primary audience for these papers is managers and

ABSTRACT

The State of Bay–Delta Science 2016 (SBDS) is a collection of papers that summarizes the scientific understanding of the Sacramento–San Joaquin Delta, emphasizing progress made during the past decade. It builds on the first SBDS edition (Healey et al. 2008). Paper topics for this edition address the most relevant scientific issues in the Delta identified by senior scientists and managers. The topical papers cover issues ranging from contaminants in the Delta to levee stability, and from Delta food webs to recent discoveries about salmon migration. These papers are written for a scientific audience. Two additional papers, one describing the challenges of managing water and ecosystems in the Delta and another that discusses policy implications of the recent scientific findings, are written for a general audience. The papers will be published in at least two issues of San Francisco Estuary and Watershed Science and will be available as a set electronically.
policy-makers, whereas scientists are the primary audience for the topic-focused papers. The papers will be published among at least two issues of San Francisco Estuary and Watershed Science but will be accessible as a set electronically.

THE DELTA

The Delta of the Sacramento and San Joaquin rivers is the hub of the statewide water redistribution system, one of the largest waterworks in the world, that is managed to achieve flood control, reliable water supply, and environmental conservation (Figure 1). This system, with the Delta at its heart, feeds the country’s most productive agricultural sector, and is critical to much of the rest of California’s $2.2 trillion economy, the eighth largest in the world (Luoma et al. 2015). Managing the Delta to prevent floods and meet the ever-changing demands for water is an ongoing problem of great complexity (Luoma et al. 2015).

THE PHYSICAL DELTA

Prior to 1850, the Delta was a 700-square-mile complex of low islands, shifting channels, large woody debris, and tule marshes (Whipple et al. 2012) that bedeviled early settlers but was the natural habitat of many species now in trouble. The historic wetlands and river floodplains have been transformed into a patchwork of islands protected by 1800 km of levees (Figure 2) that are used primarily for agriculture with some residential property. Only about 3% of the original marsh remains (Whipple et al. 2012). Before levee construction, the aquatic and terrestrial ecosystems of the Delta interacted strongly, and this interaction was critical to the ecology of native species (Wiens et al. 2016; Brown et al., forthcoming; Perry et al. 2016). The levees have isolated the aquatic and terrestrial ecosystems, contributing to the problem of native species conservation. But the levees protect valuable farmland, and are part of the infrastructure of the water export system (Luoma et al. 2015). Furthermore, many island surfaces have subsided several meters (Moore and Shlemon 2008) so that breaching the levees would create ponds, which are attractive habitat for non-native species, rather than wetlands beneficial to native species (Grossman 2016, this volume; Brown et al., forthcoming).

Freshwater flowing through the Delta forms a hydraulic barrier to the intrusion of seawater from San Francisco Bay (MacWilliams et al., submitted). If river flows drop too low, the hydraulic barrier is weakened and circulation driven by the tides can carry salt, dissolved organic material, bromide, and other chemicals to the water supply diversion points in the Delta (Schoellhamer et al., submitted; Deverel et al., submitted). Reservoir releases are crucial to maintaining the hydraulic barrier in summer and fall when rainfall is limited. During prolonged droughts, however, there is increased risk that reservoir supplies will not be sufficient to maintain the flows that keep salinity away from the interior Delta (Dettinger et al., submitted). Droughts (and floods) are a feature of California’s climate (Dettinger et al., submitted). Both unprecedented droughts and unprecedented floods are potential consequences of climate change, and both increase the risk of catastrophic failure of the levee system. The physical configuration of the Delta is both fragile and critical to the present program of water exports. It may also be a serious obstacle to conserving and restoring native species in the Delta.

THE ECOLOGICAL DELTA

The Delta ecosystem and the surrounding region supports more than 750 species of plants and animals. It is one of 25 international biodiversity hot spots listed as highest priority for conservation (Myers et al. 2000). The Delta provides year-round habitat for some species and important feeding and nursery habitat for others. Most ecological studies of the Delta have focused on the aquatic ecosystem, because of the obvious conflict between water exports and conservation of this ecosystem. Delta Smelt (Hypomesus transpacificus), has been a focal species in conflicts over water exports and, despite major changes in water allocation intended to benefit them, smelt are nearing extinction (Moyle et al. 2016). Four races of Chinook Salmon (Oncorhynchus tshawytscha) and Steelhead Trout (Oncorhynchus mykiss) move through the Delta on their seaward and spawning migrations. Acoustic tagging of smolts is revealing important details of where and when young salmon suffer mortality.
Figure 1 Map of California with Delta inset showing major water redistribution infrastructure and key landmarks
Figure 2  Map of the Delta showing the principal river channels, important landmarks, and major cities on the Delta margin.
in the Delta (Perry et al. 2016). Yet these species continue to decline. There has also been considerable study of wetland ecosystems and some study of upland ecosystems (e.g., Whipple et al. 2012; Golet et al. 2003), which further emphasizes the dramatic changes in ecosystem structure and function that have occurred in the past 200 years (Wiens et al. 2016). Conservation concern is focused on native species and more than 102 species in the Delta are of special concern.¹ All Delta ecosystems are highly invaded by alien species so that the majority of individuals or biomass found in any location are alien (Luoma et al. 2015; Brown et al., forthcoming). Cohen and Carlton (1998) described the Delta as the most heavily invaded estuary in the world, and Orsi and Ohtsuka (1999) described the zooplankton community of the Delta as essentially East Asian. A number of alien species, like Striped Bass (*Morone saxatilis*), were deliberately introduced and have become important components of the recreational economy of the Delta. Others, like the invasive reed, *Arundo donax*, are mainly a nuisance. Still others, like the overbite clam, *Potamocorbula amurensis*, have profoundly affected the productivity and food webs of the Delta (Brown et al., forthcoming). The presence of so many alien species is a major obstacle to recovering native species.

Historically, aquatic food webs were probably sustained by detrital organic carbon from the extensive and highly productive tidal marshes. The construction of leveed islands together with multiple species invasions have led to dramatic changes in open water food webs. Details of these changes are still being worked out (Brown et al., forthcoming) but it appears that shortages of suitable food may be an important contributor to native species decline. Invasive submerged and floating aquatic vegetation (Brazilian waterweed, *Egeria densa*; water hyacinth, *Eichhornia crassipes*) are also beginning to play an important role in Delta food webs with, as yet, uncertain consequences (Brown et al., forthcoming; Dahm et al., submitted).

In the early 2000s, four pelagic fish species—Delta Smelt, Threadfin Shad (*Dorosoma petenense*), Longfin Smelt (*Spirinchus thaleichthys*), and Striped Bass—declined abruptly in abundance, a phenomenon that became known as the Pelagic Organism Decline, or POD (Sommer et al. 2007). Analysis of potential causes of the POD showed that multiple stressors affected each species, and it was not possible to identify which were the most important. Water exports, which historically were believed to be the cause of native species declines, could not be singled out as the primary cause. Predation was considered to be the proximate cause of a great deal of mortality, but ultimate causes may have been changes in habitat configuration, reductions in available food, or exposure to toxic pollutants that made the species more vulnerable to predation.

Toxic pollutants are, perhaps, the most complex and least understood of the stressors. Many toxic substances enter the Delta from agriculture, industry, wastewater treatment plants, urban stormwater, and atmospheric fallout. Water samples from the Delta are sometimes acutely toxic, or cause behavioral or physiological impairment in test organisms. However, it is not yet possible to determine the degree to which chemical contamination contributes to species declines, in part because toxicity in the Delta is not routinely monitored (Fong et al., submitted). Understanding the sources, effects, and interactions of contaminants in the aquatic ecosystem remains a major gap in scientific understanding of the Delta.

THE SOCIO–ECONOMIC DELTA

The Delta is not just an ecosystem, nor a water conveyance system, nor solely an economic engine. Rather the Delta is a place where all of these services come together and are at risk (See the 2009 Delta Reform Act). As the hub of the statewide water redistribution system (Figure 1), the Delta underpins the state’s economy and is a component of the much wider water network that links the seven states of the Colorado River Pact and Mexico (Luoma et al. 2015). More than 570,000 people live in the urbanizing regions around the margin of the Delta and derive their livelihoods from the Delta or use the Delta for transportation and recreation (Figure 2). The Delta is a source of water to these people and millions more (Lund, submitted). This social dimension of the Delta is a critical consideration in every decision that affects the fate of the region. The Delta exists at many scales, therefore, from the local to the regional

to the national and international, and understanding the science of the Delta is important at all these scales (Wiens et al. 2016).

**SCIENCE OF THE DELTA**

The first edition of *The State of Bay–Delta Science (SBDS)* began with seven shifts in perspective, changes in the way that scientists understood the Delta and water supply that had emerged in the decade before its publication (Healey et al. 2008). These changes in scientific understanding have profoundly affected the long-term vision for the Delta and the way that it is managed (Delta Vision Task Force 2008; Delta Stewardship Council 2010). A similar evolution in understanding has emerged in the 8 years since publication of the first edition and we expect our understanding of the Delta and California water will continue to evolve and change as new studies are conducted. *SBDS* will be a living document, updated from time to time as sufficient new findings accumulate and, in particular, as the effects of climate change become more apparent. Meeting the new and unprecedented challenges to water supply and ecosystem conservation that climate change will bring will require both an intensification of scientific effort and much better coordination and planning of Delta science. The papers that make up the second edition of *SBDS* represent one step toward an integrated and policy-relevant compilation of science in the Delta.

**ACKNOWLEDGMENTS**

*SBDS* 2016 was initiated by Peter Goodwin in 2015 while he was lead scientist for the Delta Science Program. During 2016 the new lead scientist, Cliff Dahm, has helped guide the project. However, most of the work of communicating with authors, ensuring manuscripts were submitted, and managing the overall project logistics has been performed by Science Program staff, in particular Darcy Austin, Executive Editor for the project. Other important tasks, such as ensuring appropriate cross referencing of manuscripts and preparation of figures for this introduction were also performed by staff, particularly Maggie Christman, Martina Koller, and Megan Brooks.

**REFERENCES**


RESEARCH

Challenges Facing the Sacramento–San Joaquin Delta: Complex, Chaotic, or Simply Cantankerous?

Samuel N. Luoma*1, Clifford N. Dahm2, Michael Healey3, and Johnnie N. Moore4

ABSTRACT

Freshwater is a scarce and precious resource in California; its overall value is being made clear by the current severe drought. The Sacramento–San Joaquin Delta is a critical node in a complex water supply system that extends throughout much of the western U.S., wherein demand is exceeding supply. The Delta also underpins a major component of the U.S. economy, helps feed a substantial part of the country, is a unique and valuable ecological resource, and is a place with a rich cultural heritage. Sustaining the Delta is a problem that manifests itself in many dimensions including the physical structure of the Delta, the conflicting demands for water, changing water quality, rapidly evolving ecological character, and high institutional complexity. The problems of the California Delta are increasingly complex, sometimes chaotic, and always contentious. There is general agreement that current management will sustain neither the Delta ecosystem nor high-quality water exports, as required under the Delta Reform Act, so there is a renewed urgency to address all dimensions of the problem aggressively. Sustainable management of the Delta ecosystem and California’s highly variable water supply, in the face of global climate change, will require bold political decisions that include adjustments to the infrastructure but give equal emphasis to chronic overuse and misuse of water, promote enhanced efficiency of water use, and facilitate new initiatives for ecosystem recovery. This new approach will need to be underpinned by collaborative science that supports ongoing evaluation and re-adjustment of actions. Problems like the Delta are formally “wicked” problems that cannot be “solved” in the traditional sense, but they can be managed with appropriate knowledge and flexible institutions. Where possible, it is advisable to approach major actions incrementally, with an eye toward avoiding catastrophic unexpected outcomes. Collaborative analyses of risks and benefits that consider all dimensions of the problem are essential. Difficult as the problems are, California has the tools and the intellectual resources to manage the Delta problem and achieve the twin goals of a reliable water supply and an ecologically diverse Delta ecosystem.
INTRODUCTION

California, the most prosperous state in the nation, has an economy and a lifestyle built on water—and an illusion that freshwater is always abundant. However, the current drought, now entering its fourth year, has brought sharply into focus the fact that water is a scarce resource. With snowpack in the Sierras at a record low, Governor Jerry Brown has decreed serious water rationing, stating: “This is the new normal and we’ll have to learn to cope with it” (KTLA 5 News 2015).

Shortages of water and conflicts over water use are nothing new in California. The Delta of the Sacramento and San Joaquin rivers (the Delta; Figure 1) is at the center of these conflicts. Major state and federal water projects began delivering water from the Delta in 1949 to agricultural and urban users in the San Joaquin Valley and southern California. This redistribution of water stimulated economic growth, but the projects were soon plagued by conflict over whether, when, and how to transfer water from the Delta (Hanneman and Dyckman 2009). Conflict intensified with the listing under the federal and state endangered species acts of more than 50 native species found in the Delta (DSP 2013). As public concern grew, new policies were put in place to address environmental effects. These initiatives also led to improved understanding of the Delta, the listed species, and the complexities of managing the Delta to achieve a reliable water supply and a healthy ecosystem. Nevertheless, listed species continue to decline and dissatisfaction with water deliveries continues to grow. There is concern that the present approach to water operations is unsustainable in the face of widening demands and shrinking supplies. Frustration with management’s inability to satisfy all the demands for water has led to litigation, distrust among parties, and the threat of policy paralysis,

with cascading consequences for California, the semi-arid west, and the nation (Sidebar 1).

In this paper we look at multifaceted questions about water and environmental management in the Delta. Our goal is not to evaluate specific recent initiatives, but to provide a larger framework to guide implementation of these and future initiatives. We illustrate how the complexity of the Delta problem complicates management and leads to inefficiency and conflict. We give examples of trade-offs, disagreements, and the consequences of failure in managing these issues. We discuss why bold new approaches to managing Delta issues are urgently needed to address inefficiencies in water use, aging infrastructure, and the deteriorating condition of native species. We also show that it is important to ensure that those actions take full advantage of existing knowledge, are implemented incrementally where possible, and are accompanied by ongoing evaluations of outcomes and subsequent adjustments, as necessary. Our hope is that this paper will help managers and policy-makers better appreciate the complexity of water and environmental management in the Delta, and understand that there are ways to move forward.

THE PROBLEM

At its simplest, the problem of the Delta is similar to water challenges throughout the arid and semi-arid western U.S.: growing demands and over-allocated resources. For example, California has water rights that allocate over 500% of average annual river flows (Grantham and Viers 2014). Media reports often focus on the conflict over whether water should be exported from the Delta or left flowing through the Delta to San Francisco Bay to sustain listed native fish species. All this attention to flows and fish creates the impression that if only water managers in the major river basins would “get their act together,” the problem could be solved. But the problem of the Delta is more complex than a simple decision about allocating flows. It is a problem with many different dimensions (Table 1) and interactions that confound simple answers.

1 Widening demands for water are expected from projected population growth, economic growth, and demands to use water for the environment. The Delta Reform Act of 2009 states its “coequal” goals as “providing a more reliable water supply for California and protecting, restoring, and enhancing the Delta ecosystem.” “The coequal goals shall be achieved in a manner that protects and enhances the unique cultural, recreational, natural resource, and agricultural values of the Delta as an evolving place” (CA Water Code 85054). "Widening" means overall demand, not necessarily increasing demand per capita.
Figure 1  Water supply system in California (large map) showing reservoirs and canals that store and transport water from the wetter northern areas of the state to the drier southern and coastal areas. The Delta (inset) is at the heart of the system, pumping water to the south from two large pumping plants in the southern Delta. Reservoir volume and annual delivery is in millions of acre feet. Within the Delta, different zones are dominated by different uses and economic productivity. Agriculture is the most important economic activity in the Delta’s economy producing $800 million annually in crops (e.g., corn, alfalfa, tomatoes, wheat, and wine grapes). Adding all value-added activities (wineries, dairies, canneries, etc.), the Delta produces $2.6 billion in total economic output and 13,000 jobs for the counties encompassing the Delta, and $5.3 billion and 25,000 jobs statewide. Recreation is the second most important economic activity in the Delta, generating $312 million and over 3,000 jobs annually within Delta counties, and over 5,300 jobs and $353 million statewide. Natural gas from the Delta also produces more than 20% of California’s gas-powered electricity. (Modified from DPC 2012).
SIDEBAR 1

Delta Conflicts: Cascading Consequences

- Federal and state regulations curtail water exports from the Delta when legally protected species, such as salmon and Delta Smelt, are drawn into the pumps.
- Local restrictions on exporting water from the Delta affect the intricately balanced supply and demand of interdependent water transport networks throughout California and the Colorado River Basin.
- The Colorado River Basin Compact is a complicated deal that defines the water rights of users in the seven states that share the river. Southern California obtains water from both the Delta and the Colorado River Basin. California’s supply of Colorado River water was reduced with implementation of the compact. Reducing supplies to Southern California from the Delta increases their reliance (within the bounds of the agreement) on water from places like Lake Mead in the Colorado River system (Fleck 2012). Integration of interstate water infrastructure via these complicated agreements means that decisions about water exports from the Delta have cascading consequences for flows in the Colorado River, as well as endangered species conservation and water supply disputes throughout the Colorado River Basin.
- With so much at stake, it is not surprising that water managers argue that water disputes throughout the arid and semi-arid western U.S. cannot be resolved in the absence of decisions about managing the Delta (Austin 2015; Fleck 2012)

Table 1 The Delta problem: a nationally important but “wicked” problem with many dimensions and potentially contradicting solutions

<table>
<thead>
<tr>
<th>Dimension</th>
<th>Problem</th>
<th>Some characteristics of the problem</th>
</tr>
</thead>
<tbody>
<tr>
<td>Physical</td>
<td>Natural system seasonal and episodic</td>
<td>Strong seasonality of water supply; highly variable year-to-year; drought and floods the norm; changing climate; high earthquake damage potential.</td>
</tr>
<tr>
<td>Socioeconomic</td>
<td>Unsustainable demand from population, economy</td>
<td>Growth nearing limits of water supply; inadequate awareness that water is scarce; directly linked to the rest of the semi-arid West.</td>
</tr>
<tr>
<td>Water Supply</td>
<td>Increasingly vulnerable water infrastructure</td>
<td>Aging conveyance and levee systems stretched to limits; snowpack declining; groundwater exploited at an unsustainable rate; water used is out of balance and inadequately tracked.</td>
</tr>
<tr>
<td>Environment</td>
<td>Multiple stresses on ecosystem</td>
<td>Many native species at risk; scale of change massive, difficult or impossible to reverse; stresses difficult to manage, may act in combination, can change over space and time.</td>
</tr>
<tr>
<td>Ecosystem Restoration</td>
<td>Difficulty ensuring project success</td>
<td>Some projects help native species while others attract invasive species; benefits of water diversion mitigations questionable; successes, failures, and challenges inadequately tracked.</td>
</tr>
<tr>
<td>Institutional</td>
<td>Insufficiently unified vision for the Delta</td>
<td>Plethora of institutions with their own visions and contradicting missions; monitoring programs plentiful yet uncoordinated; management programs inconsistently coordinated and evaluated.</td>
</tr>
<tr>
<td>Science</td>
<td>Key uncertainties remain</td>
<td>Multi-institutional, collaborative approach requires new support; equal need for broadly applied science and research focused on immediate policy issues; data-sharing must be improved.</td>
</tr>
<tr>
<td>Management</td>
<td>Contradictions among solutions</td>
<td>Problems can be characterized in many possible ways; single-focus problem-solving can create unanticipated outcomes; management must be continual and adaptable.</td>
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</tbody>
</table>
Historically, the problem of water management was about supply: not enough water in the south and more abundant water in the north. California’s impressive water system was designed to address this supply problem. But California’s water problems can no longer be solved through supply management and traditional engineering solutions alone. Water supply and demand are increasingly out of balance, and the cornerstones of the water supply system are changing. Snowpack is declining with warming temperatures, groundwater is being mined at an unsustain- able rate, the infrastructure is aging, human demand for water continues to grow, and the Delta ecosystem continues to deteriorate. The accelerating pace of these changes introduces a new urgency into the need to find novel ways to manage the host of variables that affect water and the Delta ecosystem.

Human use of the Delta and surrounding lands has changed the landscape and water quality in ways that create serious environmental challenges (Figure 2). We know that multiple factors (e.g., water flows, water quality, invasive species, predation pressure, and habitat loss) interact to increase risks to native species. Despite measures to address individual stresses, the situation for many native species is increasingly dire (Sommer et al. 2007). Largely because of massive landscape transformations, the Delta cannot be restored to what it once was (NRC 2012). But the situation for native species can be improved, and there is a new urgency in taking advantage of whatever opportunities exist to do that. Exactly how to reduce the cumulative impacts of the stresses on the ecosystem is not clear (Baxter et al. 2010), but the need to address this multiplicity of problems and their interactions is as urgent as the need to address water-supply issues.

Another aspect of the problem is that more than 230 agencies, institutions, and stakeholders claim a role in water and environmental management but come with different core interests—and often conflicting visions of how the Delta should be managed. The resulting institutional fragmentation creates conflict and slows decisions. Addressing the water supply and ecosystem problems of the Delta will require management institutions that are both nimble and sufficiently coordinated to take bold, timely, and well considered actions.

Formally, the problem of water and environmental management in the Delta fits the definition of a “wicked” problem in the sense of Rittel and Webber (1973; Sidebar 2). Recognition of the Delta as a wicked problem presents a new way to think about management. Wicked problems have no single correct characterization and no single correct solution, only better or worse approaches to management of the situation. This means the Delta’s problems cannot be solved in the traditional sense, but they can be actively managed to minimize adverse outcomes and maximize beneficial outcomes (Healey 2008). Difficult political decisions and bold actions will be necessary, and this will require thinking outside the box, thinking holistically, making learning integral with doing, and finally and honestly embracing the equivalent value of water supply and ecological health. Addressing demand will be as important as addressing supply; restoring ecological function (as Moyle et al. [2012] suggest) will receive as much attention as re-engineering water-distribution infrastructure; and broadly coordinated actions will take precedence over individual institutional missions. The Delta Stewardship Council, the Delta Reform Act, and the Delta Plan provide an institutional and policy framework for this kind of operational innovation.

**SIDEBAR 2**

The Delta: A “Wicked” Problem

- If the problem were just about allocating flows, it might be solvable.
- Add in the complexity of moving water through a hydrologically and hydrodynamically complex Delta and it becomes complicated.
- Add the uncertainty of ecological responses and the institutional complexity of many actors with many visions and the problem becomes wicked (Dryzek et al. 2013).
- Then add the ever-changing water supply and ecological and economic contexts within which decisions must be made, and the problem becomes devilishly wicked.
Figure 2: The Delta ecosystem responds to factors both within the physical Delta and from regional stressors and drivers of change, including the ocean. This regional view highlights eleven major factors affecting the Delta and surrounding landscapes.
WHY IS THE DELTA PROBLEM IMPORTANT?

As the hub of a regional water-redistribution system, the Delta is a critical node in a complex network of dams, pumps, canals, drains, and reservoirs, all of which are managed jointly by local, state, and federal institutions to meet goals for flood control, water supply, and environmental conservation (Figure 1). This engineering marvel is one of the largest water-works in the world. Through California’s participation in the Colorado River Basin Compact, uncertainties about water availability from the Delta have consequences throughout seven western states and into Mexico.

Water is a fundamental driver of the economy of the western U.S. California’s economy is the most productive in the country (Figure 2; Sidebar 3). The water system is the lifeblood of this economic powerhouse and fuels the nation’s most productive agricultural sector. The Delta contributes to the California economy in myriad other ways. Commercial shipping moves through the Delta to and from the ports of Stockton and Sacramento, and several major rail lines cross the Delta. Natural gas is generated and stored in the Delta. Silicon Valley, the heart of America’s electronics industry, gets half its water directly from the Delta. California’s entertainment industry—America’s largest export—is also centered in cities dependent upon Delta water (Farhi and Rosenfeld 1998). Although the California economy has proved resilient to year-to-year water shortages in the past (Hanak et al. 2012), negative consequences of a more permanent water scarcity will be increasingly difficult to avoid (Howitt et al. 2014) and will carry over to the economies of the region, the nation, and the world.

The Delta is also of considerable ecological importance. With San Francisco Bay, it is home to more than 750 species of plants and animals. The California Floristic Province, of which the Delta and Bay are a part, is one of 25 hot spots of biodiversity across the world cited as the highest-priority areas for conservation of species (Myers et al. 2000). Some species are present year-round, like Delta Smelt, Sacramento splittail, salt marsh harvest mouse, and soft bird’s beak. Other species that are important culturally or economically, including salmon, and

SIDEBAR 3

The International, National, and Statewide Importance of an Economy Underpinned by Availability of Water from the California Delta

California’s Economic and Agricultural Profile:

- A gross domestic product of $2.2 trillion.
- The 8th largest economy in the world, equal to Brazil’s.
- Contributes 13% to the total economic output of the United States.
- Ranks 1st in the nation for patents.
- Outpaces all other states in venture capital investment with 41% of all companies in the U.S. receiving venture capital from California.
- Has the highest rate of employment by U.S. subsidiaries of foreign companies.
- Exports $174 billion of products annually ($48 billion from computer and electronics goods) for 11% of total U.S. exports.
- Imports more than $230 billion in goods from other states and countries.
- The entertainment industry in California accrues over $47 billion per year.
- California produces more food than any of the 50 states, with $45 billion in sales per year, including:
  - 40% of annual national agricultural production;
  - 45% of all the fruits and vegetables, including:
    - 98% to 99% of U.S. almonds, walnuts, and pistachios.
    - 90% to 95% of broccoli, strawberries, grapes, and tomatoes.
    - 74% of all lettuce.
- Produces many crops year-round supplying the nation with fresh produce throughout the winter.
- Because California produces most of the fruits and nuts and a high percentage of vegetables consumed in the U.S., restrictions on water for agriculture in the greater Delta affect the availability and price of these agricultural products throughout the U.S. and elsewhere.
- If production relocates because of water shortages in California, some of the conflicts over water will also relocate.

sturgeon, use the Bay and Delta seasonally. Migratory waterfowl and shorebirds use the Bay and Delta as a feeding and nursery habitat during only a brief part of their lives, but these species could not exist without these systems. The presence of migratory species connects the Delta to ecosystems as distant as Alaska, the Pacific Ocean, and South America, just as the water distribution system connects the Delta to regions far to the south and east. The Delta is truly an internationally connected ecosystem with contributions to local and state enterprise, to regionally valuable fisheries, and to global biodiversity.

Finally, the concept of the Delta as a place, enshrined in the 2009 Delta Reform Act, makes tangible the human dimension of issues such as water export and management, environmental management, and habitat restoration. All these activities go on in a real place, a place where people live and play, a place with a rich cultural history. More than 570,000 people live in the greater Delta itself, mostly in the urbanizing regions around the margin of the Delta (Secondary Zone, Figure 1). Many derive their livelihoods directly from the Delta. Most of the rest use the Delta for transportation, recreation, and as a source of water. The importance of this social dimension of the Delta is a critical consideration in every decision that affects the fate of the region.

THE DELTA: A STUDY IN COMPLEXITY

Physical System Complexity

The Delta began forming about 10,000 years ago when rising sea level slowed the outflow of the Sacramento and San Joaquin rivers through Carquinez Strait. Sediments accumulated east of the strait and created a complex of low islands, shifting channels, large woody debris, and tule marshes (Whipple et al. 2012) that bedeviled early settlers but were the natural habitat of many species now in trouble.

Human activity has transformed the original complex wetlands and river floodplains into a 3,000 km² patchwork of approximately 57 islands separated by 1,100 km of sloughs and winding waterways (CDWR 2015). It is the largest delta on the Pacific coast of North America (almost the size of the state of Rhode Island). The islands of the central Delta are used primarily for agriculture, although there is a small amount of residential property. Only remnants of the original marsh remain, and many of these are highly managed (Ferner 2012).

The physical character of the Delta is at the center of some of the most complex and contentious aspects of the Delta problem. The islands of the Delta are protected by 1,800 km of levees (Figure 3). The levees are aging and at risk of failures from numerous causes. In the 1990s, 160 levee breaches occurred, and breaches continue at a high rate (Bates and Lund 2013). Delta islands have subsided, particularly in the center and western portion of the Delta where the surfaces of many islands are now 5 m or more below sea level (Moore and Shlemon 2008), increasing the risk of levee failure. Droughts and floods also increase the risk of levee failure, and this risk will likely increase as these events become more frequent and more severe with climate change. Rising sea level, another consequence of climate change, further increases the risk of levee failure. Finally, the levee system is highly vulnerable to earthquakes. There is an estimated 60% probability that an earthquake of magnitude 6.5 or greater will occur in or near the Delta sometime in the next 35 years (Moore and Shlemon 2008). Levee maintenance is costly and upgrading levees to address the growing risks is costlier still. Ultimately, prioritization of maintenance and land uses will be necessary, and incremental approaches to this have been proposed (DSC 2015). But the levee system is also interconnected, making solutions more complex. Breaks or intentional breaches in one levee could increase the risk of levee failure elsewhere in the Delta. If any of these risks results in multiple, simultaneous levee breaks, there would be cascading consequences for water transit, water exports, local economics, and, use of islands to benefit the ecosystem.

A most important consideration in the discussion of levee maintenance is that the levees are an essential part of the California water-distribution system. Delta channels are designed, in part, to channel Sacramento River water from the north Delta to the south Delta, where it is exported via massive pumps to the Central
Valley and southern California (Figure 1). This makes the levees critical to all the human uses of Delta water. One of the greatest concerns of Delta water managers is that multiple levee failures would allow a massive salinity intrusion that would threaten the many agricultural crops and urban water supplies that rely on high-quality water exported from the Delta. Desalinization is not economically feasible to remove salt from irrigation water because of the volumes (and thus cost) involved. Water treatment facilities can remove salt from drinking water, but at a considerable increase to the cost of drinking water. In addition, there is a potential risk to human health from carcinogens that form during water treatment when the source water contains higher levels of organic matter and bromide (Richardson and Postigo 2012).

Under the current levee configuration, river flows out of the Delta provide a flow barrier that prevents intrusion of seawater from San Francisco Bay. If river flows drop too low, circulation driven by the tides (the strongest hydrodynamic force in the Delta) can carry salt, dissolved organic materials, bromide, and other chemicals to the water supply diversion points.
in the Delta. Reservoir releases are crucial in maintaining river flows in summer and fall when rainfall is limited. Thus, exports of high-quality water from the Delta depend upon a complex interaction among climate, reservoir operations, and levee configurations. For example, during prolonged droughts, there is increased risk that reservoir supplies will not be sufficient to maintain the flows that keep salinity away from the interior Delta. At the present time, after 4 years of drought, reservoir supplies are shrinking, the flow barrier is weakening, and water managers are adjusting levee configurations, each with their own problems, to ensure the quality of freshwater delivered from the Delta (Rubissow–Okamoto 2014).

**Water Supply Complexity**

The complexity of Delta water issues partly revolves around widening demand for water from a supply that is not only limited but also highly variable and growing increasingly uncertain. California’s water supply is based upon four pillars: surface water, snowpack, groundwater, and the massive human-built infrastructure that stores and redistributes water from these sources. The human-built system is effective in managing seasonal variability and regional redistribution of water. Large storms that occur in late fall, winter, and early spring are a major source of California’s water supply, contributing 30% to 45% of all precipitation in central and northern California (Dettinger et al. 2011). These storms are associated with atmospheric rivers, bands of warm, moist air from the subtropics that sweep across the Pacific and make landfall as a series of high-intensity rainstorms (with snow in the high mountains). These intense storms are a mixed blessing, sometimes providing much-needed water and at other times causing significant flooding and property damage.

California precipitation comes both as rainfall and snowpack from the high mountains. Rainfall runs off immediately, and water managers must decide whether to store this water in reservoirs for water supply or to release water to reduce future flood risk (Knowles et al. 2006). Snowpack provides a critical second source of water. California reservoirs begin to release their stored water as precipitation declines in late spring. These reservoirs are then refilled by snowmelt from carefully metered mountain snowpacks. Typically, snowpack provides just under half of California’s water supply (Dettinger 2015), allowing seasonal redistribution to proceed into the late fall when the rains normally begin again.

Year-to-year variability in precipitation is a predominant feature of the California climate and is by far the greatest in the U.S. (Dettinger et al. 2011). If one or two large winter storms do not materialize, the year will be dry; if there is an additional large storm or two, the year will be wet. Complex cycles of ocean climates2 contribute to a tendency for wet or dry periods to occur over multiple years, adding another layer of complexity to the water supply picture (Cayan et al. 1998). The reservoirs were originally designed to buffer the effect of precipitation variation, but as demand has grown the system has become increasingly less flexible. The capacity of reservoirs in the Sacramento and San Joaquin basins is about 1.1 times average annual runoff (Lund et al. 2007). Thus, reservoirs allow water managers flexibility for within-year water management but no longer provide much flexibility for dealing with multi-year droughts. Finally, long-term trends in California’s water supply associated with climate change portend growing uncertainty in water supply and uncertainty about strategies for coping with increasing variability (Sidebar 4).

For decades, groundwater has provided the back-up to lessen the effect of surface water variability. Of the total California water supply, about 40% comes from groundwater wells (CDWR 2014). But in some regions groundwater is being used faster than it is being replenished. For example, groundwater supplies in the Central Valley have decreased by about 79 million acre feet since the early 1960s (CDWR 2014; Famiglietti et al. 2011). The current drought has greatly exacerbated the issue. From spring 2013 to spring 2014, before the worst of the present drought hit, groundwater levels dropped in 88% of the wells in California, with 22% of those wells dropping by more than 10 feet in that 1 year. As groundwater levels drop, costs increase, availability declines, and

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2 The El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) phenomena are two examples of ocean climate influences [http://www.pmel.noaa.gov/tao/elnino/el-nino-story.html].
land subsidence occurs (Faunt and Sneed 2015). These signs of groundwater depletion add considerable uncertainty to the supply picture for the future. Exact measurements of groundwater reserves and the cost to access and use groundwater under different future climatic scenarios are crucial to understand the implications of current rates of groundwater depletion. In 2014, California passed legislation requiring that groundwater reserves be measured and groundwater use regulated. Implementation of this new law will require increased study and monitoring of the groundwater system at local, regional, and statewide scales.

Water Quality Complexity

A broad array of nutrients and potentially toxic chemicals enters the Delta from agricultural runoff, and there exists a long legacy of mining and industrialization in the watershed (van Geen and Luoma 1999; Sidebar 5). Today, more than 100 industries, wastewater treatment plants, and urban stormwater discharges release waste streams to the Bay and Delta (van Geen and Luoma 1999). The waste streams are mostly treated, but the Bay and Delta are, nevertheless, listed under the federal Clean Water Act as impaired because of the presence of a variety of toxic contaminants. People are advised not to eat striped bass, white sturgeon, and some diving ducks caught in the Bay and Delta because they may contain high concentrations of mercury, selenium, PCBs, or DDT breakdown products.

The complex spectrum of chemicals entering the Delta is continually changing over time as regulations, industry processes, and consumer preferences change. Federal and state regulations (e.g., the Clean Water Act, passed in the 1970s) have made substantial progress in reducing inputs of some toxic chemi-

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**SIDEBAR 4**

**Uncertainties About California’s Future Water Supply Arising from Climate Change**

- More precipitation falls as rain in late winter and less as snow in mid-winter.
- Spring snowmelt occurs earlier because of higher temperatures.
- Less snow and earlier melting means less water is stored as snowpack and more uncertainty about water availability from reservoirs in the late summer and fall.
- Higher temperatures yield less runoff from the same rainfall amount.
- Average precipitation increases in northern California but decreases in southern California.
- More frequent extremes: prolonged drought, floods from atmospheric rivers.
- Greater dependence on groundwater as a buffer from extremes.
- Increasing costs and decreasing availability of that buffer as groundwater is over-exploited.

*(Sources: Cloern et al. 2011; Dettinger and Cayan 2014)*

**SIDEBAR 5**

**Contaminants in the Delta and San Francisco Bay**

- Mercury from historic mining sources contaminates food webs.
- Selenium from Central Valley irrigation drainage and Bay refineries affects reproduction of native predator species in the Bay.
- Organic chemicals remaining in sediments from historic use accumulate in food webs, including DDT and its breakdown products, and polychlorinated biphenyls (PCBs).
- Pharmaceuticals, flame retardants, and personal care products from waste treatment facilities disrupt endocrine systems of aquatic organisms and birds.
- Multiple, changing pesticides from agriculture and urban uses cause toxicity at least near their points of release.
- Nutrient inputs from wastewater treatment facilities and other sources affect Delta food webs.
- Nitrogen, phosphorous, and other nutrients stimulate nuisance or toxic algal blooms and water weeds, as turbidity of water declines.
cals (metals, some organic compounds) into the Bay and Delta (van Geen and Luoma 1999) and reversed adverse ecological effects around what were once contamination hot spots (Hornberger et al. 1999). Nutrient input remains a source of concern, although management has improved in some areas (Sidebar 6). Newly emerging contaminants pose another concern, and include pharmaceuticals, flame retardants, and personal care products that are shown to cause endocrine disruption in fish and other organisms. There is evidence of toxicity to invertebrates at the base of the food web, at least near the sources of inputs for some pesticides (Weston and Lydy 2010) and PCBs (Janssen et al. 2011). In addition, selenium causes reproductive effects on some native fish (Stewart et al. 2013). Finally, the fate of chemical wastes is interwoven with the physical characteristics of the modern Delta. Many aspects of water quality are affected by river inflows, Delta hydrodynamics, connections to the Bay, and changing temperature and turbidity. All of these interact with each toxic chemical to create variable exposures over time and space. In short, there is cause for concern about the potential for adverse effects from toxic contaminants, even though exact risks are difficult to assess and are confounded with the effects of other stressors.

**Ecological Complexity**

Before European colonization, the Delta was a vast, 3,000 km² complex of low, forested islands, tule marsh, and meandering channels (Figure 3). Parts of the Delta flooded and drained with each tidal cycle, and most of the Delta flooded during the spring, after which parts dried out during the long period of low river flow in the summer and autumn. The tidal and seasonal cycles of flooding, draining, drying, erosion, and deposition created and sustained the Delta. This was the environment in which native species evolved and in which they thrived. The life cycles of many native species were cued to these natural rhythms. As tides rose and inundated island marshes, fish would

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**SIDEBAR 6**

**Nutrient Inputs: An Example of a Changing Issue with Regional Implications**

- The waterways of the Delta are enriched with nitrogen, phosphorus, and other nutrients that come from natural sources, agricultural inputs, and wastewater treatment facilities.
- Nutrients typically fuel the growth of phytoplankton (open water algae) and aquatic plants that form the base of the food web in the Delta. Plant productivity determines the availability of food resources to zooplankton, aquatic invertebrates, and fish.
- Annual primary production of the phytoplankton in the Delta has typically been low compared with other estuaries because of limited light penetration into turbid waters and the low residence time of water in the Delta (Jassby et al. 2002). Feeding by bottom-dwelling animals that filter the water column also reduces phytoplankton availability to the pelagic food web.
- Summer blooms of a harmful algae (*Microcystis aeruginosa*), that began in 1999, are a new concern (Lehman et al. 2005), for the first time raising the specter of ecological problems from nutrient inputs.
- The problem has been accentuated by an increase in the clarity of the water that allows more light penetration. This occurred as the residual sediments from hydraulic mining passed through the ecosystem, and dams captured sediments that originated upstream.
- Nutrient availability, especially ammonium from wastewater treatment plants, facilitated the invasion of two non-native aquatic plants (Brazilian waterweed, and water hyacinth), which are now well established in the Delta (Santos et al 2009). Both grow well in high-nitrogen environments if light is available, and are effective at using ammonium as a source of nitrogen.
- Programs are being initiated to reduce nitrogen discharges. A sustained commitment to experimental nitrogen-removing technologies illustrates that creative new ways to address stressor problems exist. Although it is uncertain to what degree nitrogen reductions alone will shift trajectories for native species, it is an example of bold, prudent action with a low probability of cascading negative outcomes.
invade the marsh along tidal channels, feeding on the abundant food resources of the marshes before retreating into the main Delta channels as the tide ebbed. Shorebirds would also populate the emerging mud flats to probe for food. Fish species such as splittail were adapted to the seasonal flooding, moving onto the floodplains to spawn during the spring floods and retreating to the main river channels with their young as the flood receded.

Very little of this historic ecosystem remains (Figure 3). The modern Delta is a patchwork of leveed islands separated by channels. These islands do not flood on tidal or even seasonal cycles, unless levees fail. Little wetland habitat remains, and what does is not subject to the extent of flooding and drying that characterized the historic wetlands.

Beyond transformation of Delta habitats, human development imposes a wide array of additional drivers of environmental change (Figure 2) with effects that vary among species, locations, and with time. The severity of the cumulative effects of these stressors is manifested in the estimate that 80% of native fish species are in decline (Hanak et al. 2011). Many of the risks from individual stressors are understood, but the relative importance of each stressor to the cumulative consequences is difficult to pinpoint. Moreover, natural cycles and climate change constantly shift the baseline conditions in the ecosystem (Cloern and Jassby 2012), adding to the complexity of determining why changes are occurring. As a result, predicting the outcome when water operations, land forms, or the levees are changed is uncertain, at best.

Since passage of the Central Valley Project Improvement Act of 1992, federal and state agencies have focused attention on how to sustain viable populations of native species in the Delta while still maintaining water exports from the Delta. Early attention focused on prevention of mortality at the export pumps (Sidebar 7) and management of flows through the Delta for the benefit of native species. More than a decade of litigation has been driven by uncertainties about the effectiveness of the regulations that curtail exports, and how these curtailments and other water management operations, in real time, negatively affect the populations of legally protected fish species. Even defining water allocations for the environment versus human use has been a source of controversy (Sidebar 8).

As more has been learned about the Delta ecosystem, it is clear that recovery of native species will require cumulative effects from all stressors to be addressed. A good example of the dire circumstances that characterize the Delta ecosystem is the recent sharp decline of several native fish species, termed the pelagic organism decline or POD (Sidebar 9). Statistical studies, improved conceptual models, and improvements in quantitative modeling of the environment all point to multiple causes for the POD, and perhaps a broad change in the overall ecological regime of the Delta. Initial studies of the POD were focused on declines in abundance of a few species.

**Sidebar 7**

**Preventing Mortality of Legally Protected Fish Species in the Delta**

- One focus of Delta management is regulations that curtail water exports when legally protected species, such as salmon and Delta Smelt, are drawn into the pumps.
- At a larger scale, water project operations also affect water movement and water quality throughout the greater Delta changing, for example, cues that fish such as salmon use to direct their seasonal migration from spawning rivers to the sea and back.
- Today only 5% of the young salmon that enter the Delta in their seaward migration survive to enter the ocean (del Rosario et al. 2013). That proportion dropped from 40% in the 1990s.
- It is difficult to determine unambiguously how much of this mortality is caused by water operations, how much by habitat change, or how much by interactions with other causes of mortality, such as predation by non-native species (Figure 2).
- Survival of migrating chinook salmon has been improved to 86% to 94% by scientifically supported actions in the Columbia River system (northwest U.S.; Muir et al. 2001). This means improving migratory survival is feasible, and is an example of an opportunity to improve the situation for native species.
such as Delta Smelt or longfin smelt and their link to water diversions. But broader conceptual models (e.g., IEP MAST 2015) led to the recognition that more species and other events were involved with this change. The idea that focusing action on one problem will allow relaxation of the regulation of others has underlain much of the contentious dialogue about Delta restoration. The POD studies and others show that concerted action on multiple fronts offers the best opportunity for progress.

It is difficult to pin down the causes of events such as the POD, in large measure because today’s Delta is essentially an alien habitat to the hundreds of native species that try to live there. Under these circumstances, it is no surprise that many native species are struggling to survive, and that many factors are implicated in their low population numbers. The Delta cannot be returned to the way it was 200 years ago. The great challenge is to figure out how to provide enough suitable living space in the modern Delta for these species to persist (Moyle et al. 2012). The challenge is increased by the continually evolving nature of the ecosystem as new species arrive, and as land use and climate change (Sidebar 10).

Institutional Complexity

Because managing water and environment is inherently complex, the tendency is to break the perceived

### Sidebar 9

**The Pelagic Organism Decline**

- Four pelagic species—two native species (Delta Smelt and longfin smelt) and two introduced species (juvenile striped bass and threadfin shad)—declined to record low numbers in only a few years beginning in 2002–2004.
- The collapse of these populations occurred despite management actions intended to improve conditions in the Delta, and relatively moderate hydrological conditions at the time.
- Before this event, most attention had focused on water exports as the principal cause of the declining abundance of native species. Careful re-examination and re-analysis of data was catalyzed by the dramatic change in fish populations.
- Although different stresses (Figure 2) figured most prominently in different studies, all showed that direct effects of water exports was only one factor—and perhaps not the most important factor—in this most recent species decline in abundance.
- Conceptual models, rooted in ecological theory, are developing ideas about how a number of drivers of change interact to cause precipitous declines in species. These models are qualitative and generalized, but do provide a useful framework for organizing and synthesizing both data and ideas related to the conservation of pelagic fish species.
problem down into what seem like manageable pieces and address each piece more or less independently. The result has been a plethora of agencies, departments, and commissions at federal, state, regional, and local levels of government, each dedicated to addressing one or more components of water and environmental management (Figure 4). Private interests, like the State Water Contractors, and non-governmental organizations, like the San Francisco Estuary Institute and the Nature Conservancy, are also involved. The repeated crises in management of the Delta have only served to increase this institutional complexity (DSC 2013).

When so many institutions with different mandates are involved in management of a critical resource such as the Delta, integration and coordination are critical. Although there are notable examples of long-standing cooperation and integration among state and federal agencies (the Interagency Ecological Program, for example), there are also notable examples of decision-making that is fragmented and uncoordinated, leading to inefficiency and poor outcomes (NRC 2012). One consequence of the fragmentation of responsibility and authority over the Delta is the increased difficulty of addressing Delta problems. The complexity provides a multiplicity of ways for individuals and organizations that are dissatisfied with water or environmental management to seek redress for their dissatisfaction through litigation. The Delta Reform Act of 2009 attempted to address this complexity by establishing the Delta Stewardship Council with responsibility for achieving the coequal goals of a more reliable water supply for California and protecting, restoring, and enhancing the Delta ecosystem. Time will tell whether the Council can achieve sufficient cooperation or has sufficient authority to address institutional complexity.

THE ROLE OF SCIENCE

As we noted earlier, the Delta is one of the most studied ecosystems in the world. A growing understanding underpins ongoing adaptations in managing the Delta. Throughout the decades of conflict over water issues, all parties have recognized that advancing the state of scientific knowledge is fundamental to making constructive progress. As we enter an era of increasing uncertainty about climate and water supply, commitments to multi-institutional science that informs policy beyond agency decisions are critical. Continuing advancement of scientific understanding and effective integration of science into management will require science that embraces differences of scientific opinion, structures science in a way that is useful for management decisions,

### SIDEBAR 10

**Non-Native Species and the Restoration Conundrum**

- Aquatic and terrestrial habitats are heavily invaded by non-native species delivered by international shipping, recreational boating, the horticulture and pet industries, agriculture, or deliberate introduction.
- San Francisco Bay and the Sacramento–San Joaquin Delta have been described as the most heavily invaded estuary in the world (Cohen and Carlton 1998). Cohen and Carlton (1998) showed that 40% to 100% of species found in various aquatic habitats was non-native.
- The consequences of invasions by exotic species can be dramatic.
- A 1986 invasion of the overbite clam changed phytoplankton dynamics in Suisun Bay, and eliminated what was once a large spring bloom of plants that was essential to native food webs.
- Introduced predators, like striped bass and largemouth bass, have grown to large populations in the Delta, and their predation on native fishes is thought to contribute to the decline of such species.
- Restoration of shallow-water habitats is often plagued by invasive plants and invasive predators instead of fostering more habitat for native species. Currently, it is unclear how best to reduce populations of invasive species or how to increase the certainty that new habitat will be best suited for native species.
Figure 4 Complexity diagram of actors (red circles) and institutions (blue squares) involved in water governance of San Francisco Bay (Source: Lubell et al. 2014).
and quantifies uncertainty. Looking into the future of complex problems like the Delta will require scientific models that can simulate the consequences of different management approaches. Such models have been developed for water operations; are in their early stages for the ecosystem (DiGennaro et al. 2012) and climate change (Cloern et al. 2011); and have been used to envision alternative futures for the Bay–Delta (e.g., Lund et al. 2010). The understanding necessary to integrate and strengthen these models is growing rapidly, but is scattered among agencies and research institutes and needs to be brought together. Challenges remain in merging models of various types, and in ensuring that the model output is sufficiently reliable for management. But if carefully implemented and interpreted, such models can provide valuable guidance to policy, management, and science (Healey et al. 2008).

Continuously improving models and scientific understanding of the Delta problem is necessary but not sufficient to manage successfully the complex technical, political, and resource challenges facing the Delta. There will always be uncertainties that surround any action. Difficult political choices will be necessary. Adaptive management is the preferred approach to implementing management actions in the face of uncertainty. Regular monitoring and evaluation of the Delta’s response to management is the best way to detect unexpected outcomes and adjust management actions to deal with uncertainties. Although a number of monitoring and assessment programs exist to aid in such evaluations, there is not as yet a unified set of performance criteria for the key dimensions of the Delta problem. As adaptive management becomes more fully implemented, such criteria must be developed, implemented, and reported on regularly. Effective adaptive management also requires collaboration, communication, and transparency among all interest groups as well as a willingness to overcome the institutional barriers to collaborative decision-making. Recent commitments to collaborative decision-making are encouraging (e.g., the Collaborative Adaptive Management and Policy Team) but sustaining those initiatives has always been a challenge.

CONCLUSIONS: COPING WITH COMPLEXITY

The Sacramento–San Joaquin Delta is at the hub of an interconnected water-delivery system that feeds the impressive economy of California and also influences the economies of most of the western U.S. At the same time, the Delta is an ecological resource of international significance with a rich social and cultural history. The challenge in managing the Delta is to preserve all these important functions in the face of a widening demand for water that frequently exceeds available supply, including demand from a growing population, a growing economy, valuable agriculture, and a unique environment. The challenge is enhanced by climate change, which is raising temperatures, changing storm patterns, and reducing snowpack, leading to an increasingly uncertain supply of water and changing environmental conditions. Unsustainable mining of groundwater (Bredehoeft and Alley 2014) is increasing costs and decreasing the availability of a source of water that has long provided a buffer against drought. Water managers no longer have the flexibility they once had in dealing with the multi-year droughts that are inherent to the California climate. Managing the water supply system alone is complicated. But add in the imperative to sustain the ecological and social values of the Delta and every decision becomes considerably more complex. The current arrangement for addressing this combination of complexity, uncertainty, and change is unsustainable, as evidenced by both declines in native species and dissatisfaction with water deliveries.

From an environmental perspective, the ecosystem of the Delta is vastly transformed from its original state, making life difficult for a host of native species. Multiple interacting factors affect the well-being of native species. Some of these factors are well understood, but their interactions and cumulative consequences are not, making it impossible to make definitive statements about what is causing native species to decline. As a result, predicting the outcome when water operations, land forms, or the levees are changed is uncertain, at best. Nevertheless, opportunities exist to conserve and restore aspects of the native system and to structure the rest of the Delta to make it more hospitable to native species. Realizing
those opportunities without jeopardizing water supply is the ultimate challenge in managing the Delta.

Many of the approaches used in water-scarce environments elsewhere are under-utilized in the Delta. While adjustments to the infrastructure as it ages are essential, opportunities exist to simultaneously redefine bold action as we pursue proven (although not always initially popular) ways to work more effectively with what we have (http://www.energy.ca.gov/wet/). Examples include the following:

- Groundwater recharge and conjunctive use offer storage potential beyond that available for surface waters (CIWR 2015).

- Initiatives to promote water reuse, water recycling, and desalination in selected circumstances are under-utilized and can help address the imbalance between demand and supply (ACWA 2015).

- Priorities for maintenance and upgrades of the levees can be built from growing understanding of physical vulnerabilities, climate change, economics, and water transit needs (DSC 2015).

- Making water conservation a continual, long-term, statewide investment is a necessary part of accepting water scarcity (USEPA 2015; NatGeo 2014).

- Greater attention to both the tributaries and the Bay in Delta planning, including wetlands restoration, offer opportunities for both protection from sea level rise and ecosystem restoration (Save the Bay 2015).

- Continuing the precedent of improving water quality from tributary inputs and within Delta sources can help counter the expansion of exotic species (Brown and Caldwell 2015).

- Risk reduction for catastrophic Delta infrastructure failure can include investing in targeted levee improvements, addressing additional stresses from sea-level rise, and planning for climatic extremes such as atmospheric rivers and long-term droughts.

- Making the “One Delta, One Science” concept a reality will improve the underpinning for political actions in the face of uncertainty (DSC 2013).

Complex, wicked problems like the Delta rarely yield to the simplistic solutions directed at only one dimension of the problem. The lack of flexibility resulting from the already complete allocation of a shrinking water supply, combined with the serious deterioration of the native ecosystem, will reduce the effectiveness of many traditional engineering solutions in the Delta. History shows that large-scale, irreversible, physical changes in the water system are particularly risky (Sidebar 11) unless they promote flexibility and are implemented incrementally (Sidebar 12). Incremental, as used here, does not imply “small,” but “implementation in stages,” such that lessons learned from early increments can be used to improve design of later increments. While economics alone may not always support such an approach, it is time to recognize that other dimensions of the issue also must carry weight.

New approaches to scenario-building and modeling can help managers explore the potential outcome of major management initiatives and anticipate problems before they arise. Modeling and scenario-building needs to be a collaborative, multi-institutional activity. As we enter an era of increasing uncertainty about climate, water supply, the fate of the Delta’s native ecosystem, and institutional complexity, multi-institutional collaborative approaches will become increasingly important.

Water scarcity has defined and will continue to define the future of the Delta and all that is linked to it. California has risen to the challenge of water scarcity in the past to build an economy and a society that is, in many ways, the envy of the world. The present problem of water scarcity seems more complex and less amenable to traditional engineering solutions than in the past. But California has the tools and the intellectual resources to manage the problem and to achieve the twin goals of a reliable water supply and an ecologically diverse Delta ecosystem.
SIDEBAR 11

Implementing Inadequately Understood Engineering Solutions: The San Luis Drain Example

An example of implementing a simple solution to a complex problem is the issue of irrigation drainage in the Central Valley.

- As a part of the Central Valley Water Project in the 1950s, governments were obligated to deal with the return drainage that resulted from the export of water from the Delta.
- The simplest solution was to build drainage infrastructure under the agricultural fields and a canal (the San Luis Drain) to take the drainage to San Francisco Bay.
- The first increment of that system was completed in the 1980s with the drainage canal temporarily terminating near Kesterson Wildlife Refuge.
- Soon after the drainage disposal began, severe deformities were observed in birds, including birds that were part of the international Pacific Flyway. Later studies showed a massive ecological disaster, which was eventually attributed to heretofore, unknown selenium contamination in the drainage (Presser 1994).
- Later studies showed that a similar, if not worse, outcome was likely if the drain was extended to the Bay (Presser and Luoma 2000).
- Dealing with this problem has been much more expensive than the San Luis Drain itself. Adverse effects of irrigation drainage products such as selenium will always be an important consideration in any plans that change water-redistribution systems. The selenium problem cannot be solved, but it is being incrementally managed by land retirement and multiple, local in-valley treatment systems. The San Luis Drain was a multi-million dollar "stranded investment" that resulted from a poorly understood, simplistic engineering "solution" to a complex problem with many dimensions.

SIDEBAR 12

An Example of Incrementally Approaching a Complex Problem

Most Delta restoration projects have not been in place long enough to draw conclusions about the approaches being used. But the Kissimmee River in south Florida provides an example of how an incremental approach to restoration can work. Key elements of this widely proclaimed restoration success are listed here (see Dahm et al. 1995):

- River channelized for flood control from 1962–1971 at a cost of $38 million
- Collapse of key bird and fish communities
- Mounting interest and public pressure for restoration
- Pilot project to reroute some canal water back onto floodplain from 1984–1988 with positive responses from birds and fish
- Design phase for a large-scale restoration in the early 1990s with a rigorous evaluation program
- Testing sediment plug from old spoils piles to see if the channelized river could be rerouted onto the old floodplain in 1996; plug functioned as designed
- Construction of Phase One restoration 2000–2001 for about 30 kilometers of river and 3,200 hectares of wetland
- Initial restoration largely successful
- Currently carrying out Phase Two of restoration
- Restoration costs to date approaching one billion dollars
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ABSTRACT

Much of the water supplied in California for agriculture and cities is taken directly from the Sacramento–San Joaquin Delta (Delta) or indirectly from surface and groundwater diversions upstream. These water supplies have great economic and social value, and considerable ecosystem effects. Long thought of as the major source of water for economic growth in California, the reliability of water supplied from the Delta is threatened by drought, flood, climate change, earthquakes, growing water demands, and deteriorating conditions for endangered species and native ecosystems. Research in recent years has improved understanding of how management of the Delta ties together the quantity and quality of water available statewide. These ties run from the Sierra mountains and coastal streams, through the Central Valley, to the San Francisco Bay Area, and over the Tehachapi Mountains to southern California. For decades, Californians counted on reducing Delta outflows to supply water for growing water demands in its watershed and in water importing areas. With greater competition for water, concern for environmental effects, and a changing climate, the reliability of such supplies is now diminishing. This must lead to tighter accounting and modeling of water supplies in the Delta and throughout its watershed. This paper reviews issues about Delta water supplies, operations, regulations, and reliability; the economic value of supply; costs of unreliability in quantity and quality; and several directions for further scientific and technical work on water supply reliability.

KEY WORDS

Water supply, reliability, Sacramento–San Joaquin Delta

INTRODUCTION

The Sacramento–San Joaquin Delta (Delta) was created beginning about 6,000 years ago, after the last ice age, by sea level rise progressively drowning the confluence of the Sacramento and San Joaquin rivers, just upstream of San Francisco Bay. In modern times, the Delta has become the most central single feature of California’s extensive and diverse water supply system. Consequently, water supply reliability is one of the state’s “co-equal” goals for the Delta. Although the Delta’s importance to the water supply system is widely known, its roles are...
complex and often misunderstood. Public and policy discussions of potential diversion tunnels under the Bay Delta Conservation Plan (BDCP) and Water Fix proposals have enlivened interest in the Delta and its peculiarities in local and statewide water supplies (CDWR 2015b). This paper reviews California’s water system, the Delta’s roles in this system, and the fundamental economic demands for water from this system. It also reviews how the Delta ties the state together and is a source of conflict. Then I offer some fundamental observations about the Delta and water supply, along with some directions and limitations for the use of science to improve our understanding and management of water supplies both diverted from Delta outflow, and flows within the Delta.

OVERVIEW OF CALIFORNIA’S WATER SYSTEM

California has one of the world’s most extensive and interconnected water systems, supporting over 38 million people and almost 9 million acres of irrigated agriculture within a $2.3 trillion economy that is tightly connected globally by trade, migration, and communication (Luoma et al. 2015). The supplies and demands of this water system are substantial and diverse, with supplies being particularly variable across seasons and years, as the ongoing drought highlights.

Figure 1 shows the distribution of runoff, which is the source of all surface and ground waters, across California. In space and time, water availability is highly mismatched with water demands. About 90% of California’s runoff comes from 40% of its land surface, predominantly in northern and mountainous areas. Most human water demands are located in drier areas the in central (agriculture; 80%) and southern (cities; 20%) parts of California (CDWR 2013).

Figure 1 California’s uneven geographic distribution of water availability and water use. (Source: Hanak et al. 2011.)
California’s runoff occurs predominantly in winter and spring (Figure 3), sometimes with substantial flooding. This timing is seasonally opposite of most agricultural and urban water demands, which are mostly from crop and landscape irrigation during California’s long dry summers. Climate warming and diminished spring and summer snowmelt will likely concentrate California’s annual runoff more into winter months (Lettenmaier and Sheer 1991; Gleick 1987).

The flat topography and high land values of coastal urban areas make local storage of high winter runoff difficult and expensive. Even the Bay Area, with relatively high annual precipitation, imports most of its water from distant mountain reservoirs (diverting runoff that otherwise would have gone through the Delta).

In response to this mismatch of hydrology and human water demands, California has built an extensive system of water-management infrastructure, depicted in Figure 2 (Hundley 2001; Hanak et al. 2011). The Sacramento–San Joaquin Delta is the greatest single hub in this water system. California’s extensive water management infrastructure is owned and operated by a wide range of hundreds of local, federal, and state agencies, as well as several private companies, and is subject to regulations by a myriad of federal, state, and local governments. Over time, the system has become increasingly inter-tied, as improvements in reliability often come more cost-effectively from more flexible operations and from sharing water. The institutional complexity of California’s water system often exceeds even its great physical complexity (Luoma et al. 2015; Lubell et al. 2014).

CLIMATE AND WATERSHED-BASED SUPPLIES

California has a highly variable climate, with great seasonal differences in precipitation and runoff (Figure 3) and large fluctuations between years (Figure 4). Much of the year-to-year precipitation variability derives from the few storms each year that provide much of the annual runoff totals. Across the Delta watershed, half or more of the average precipitation arrives in less than 10 to 15 wet days per year (Dettinger et al. 2011). Just a few storms can cause the difference between a dry and a wet year. Broadly speaking, water year totals range from about 50% to 60% of average to almost 200% of average with unusual frequency in California. Some of this year-to-year variation reflects global-scale climate processes such as El Niños and La Niñas (Schonher and Nicholson 1989) and their multidecadal counterpart, the Pacific Decadal Oscillation (PDO) (Mantua et al. 1997). However, more variation is from year-to-year variations in the number and size of large storms, most often as atmospheric rivers (Dettinger and Cayan 2014). The same storms that supply much of the state’s water supplies also routinely cause large, dangerous floods. Flood- and water-supply management are inextricably joined in California (Lund 2012). The Delta is at the downstream end of both.

Much of California’s climate is Mediterranean, with colder wet winters and dry hot summers. Seasonal snowpacks in California’s high mountains store and

![Figure 2](http://dx.doi.org/10.15447/sfews.2016v14is3art6)
shift large amounts of runoff from the winter to the spring and early summer (Figure 3). Higher temperatures reduce snowpack and accelerate snowmelt, reducing snowmelt-related shifts in seasonal runoff. Climate warming will diminish the State’s snowpack to an extent that will depend on global greenhouse-gas concentrations and the resulting warming of California’s climate (Anderson et al. 2008).

California’s Mediterranean climate and long dry summers mean that each year California has a longer and more severe drought than the eastern U.S. has ever seen in history. The unimpaired flow\(^1\) available to supply water demands varies greatly seasonally and between years, as depicted in Figure 3. This absence of precipitation during the main growing season, along with the state’s strong year-to-year swings of precipitation, drought and floods (Figure 4), motivated development of California’s extensive water system.

About three-quarters of the water flowing towards the Delta is from inflows from the Central Valley’s surrounding mountains, so-called “rim flows” into the Central Valley and Delta. These flows are largely regulated by reservoirs, mostly along the Central Valley’s rim, for water supply and flood control, with other purposes including hydropower, recreation, and, increasingly, support for ecosystems. The most common operations is to partially fill large reservoirs during the wet winter, while reserving considerable space to manage floods, and then allow this flood space to refill with snowmelt during the spring. These flood control and water supply operations of large reservoirs greatly dampen the seasonal effects of flows from watersheds on Delta inflows and outflows. Such reservoir operations can be improved with better coordination and forecasts (Graham and Georgakakos 2010; Maher 2011; Rheinheimer et al. 2016).

A warming climate is already shifting inflows from the spring to winter, as precipitation falls more as rain and less as snow, and snowpack melts earlier (Aguado et al. 1992; Cayan et al. 2008). Even if climate change brings similar amounts of precipitation, larger evaporation and evapotranspiration rates are likely to reduce streamflows (Null et al. 2010), including the effects of additional clear days (Viviroli et al. 2011).

Upstream flows also are affected by upstream watersheds conditions and activities, particularly forest conditions, which are themselves affected by climate, human land and fire management, and natural hydrologic and ecosystem processes (Bales et al. 2011; Brown et al. 2005; Ursino and Rulli 2011). Wildfire and watershed conditions also can affect water quality downstream (Smith et al. 2011; Bladon et al. 2014; Dahm et al 2015). Aggressive fire suppression has increased the density of trees in upstream forests, further increasing evapotranspiration, and reducing streamflow averages from natural conditions (McIntyre et al. 2015; Westerling et al. 2006). Restoration of mountain meadows also increases evapotranspiration losses and decreases average streamflows, while somewhat evening downstream flows (Hammersmark et al. 2008). The large reservoirs around the Central Valley’s rim diminish the effects of upland activities on the timing of water supplies and floods to the valley floor. These rim reservoirs also support major upstream diversions upstream of the Delta that shift

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\(^{1}\) “Unimpaired flow” is the streamflow that would occur with today’s land uses, without upstream dams or diversions; it is the water available today for management. “Natural flow” is what streamflow would be without upstream diversions and land use changes—such as changes in watershed land use and levees—and has some relevance for riparian water rights and studies of natural ecology. Sadly, these terms are often used interchangeably and inappropriately. Both estimates are subject to differences in hydrologic judgment and calculation.
seasonal and some inter-annual flows into the Delta and supply consumptive upstream agriculture and cities which diminish total Delta inflows.

WATER SYSTEM OPERATIONS IN THE DELTA

Water supply operations in the Delta consist of various water diversions, pumps, and gates in the Delta, as well as releases of water from reservoirs upstream, which are often coordinated to regulate Delta water quality. The operation of Delta pumping and gates varies with water demands, Delta inflows, and Delta water-quality conditions (driven by hourly, daily, and seasonal conditions of Delta tides, inflows, and diversions), and regulations to support endangered fish species. Figure 5 shows major annual Delta inflows and outflows for a wet water year (2011) and a critically dry year (2015).

Local agricultural and urban water diversions, and local water quality—particularly in the western Delta—reflect strong tidal and seasonal variations in water quality. For example, until the early 1900s, the city of Antioch directly diverted Delta water year-round. Beginning in the early 1900s, upstream irrigation diversions reduced summer Delta outflows and increased salinity in the western Delta. Over time, despite the operation of upstream reservoirs, Delta and upstream diversions have greatly reduced Delta outflows (SFEP 2015). Today, Antioch still diverts some of its water use directly from the Delta in wetter months; Contra Costa Water District (CCWD) supplies the remainder of its water (Brown and Caldwell 2011). Likewise, CCWD’s intake on Mallard Slough (opposite Chippens Island) is only used in wet periods. As western Delta salinities often vary substantially over the tidal cycle, local diversions sometimes can continue by varying diversion locations and quantities with the tides.

The CCWD delivers about 100 taf yr\(^{-1}\) to its customers, with 160 taf of storage capacity in Los Vaqueros Reservoir, giving it considerable operational flexibility for seasonal and over-year storage and blending of different water qualities. This flexibility is enhanced by interties to the East Bay Municipal Utility District’s (EBMUD) upstream Mokelumne River diversions and storage. Agricultural water operations in the Delta divert much more water than local cities and vary considerably among the different
Delta islands, having different elevations, soils, and cropping patterns (Siegfried et al. 2014).

Since the 1950s, water projects in the Delta began pumping large amounts of water. Upstream water diversions also have grown, apparently reducing average annual Delta inflows (and outflows) at a faster rate than direct Delta pumping, as seen in Figure 6. (Upstream diversion growth is not at a statistically significant positive rate, given the high inflow variability. Central Valley unimpaired inflows, not shown, have a slight statistically insignificant increasing historical trend.) Delta project diversions (for the State Water Project [SWP], federal Central Valley Project [CVP], and CCWD) have significantly increased since the 1950s, except for the most recent decade, when droughts, environmental regulations, and reductions in CCWD demands have significantly reduced Delta project diversions.

Large SWP and CVP Delta diversions have more effect on Delta flows than smaller diversions and are regulated more by state and federal agencies. These larger diversions also have more access to...
water-storage capacity, allowing water diverted in wetter times to be carried over to drier times. The SWP and CVP have much larger intakes in the southern Delta at Banks and Jones pumping plants, respectively. These projects (with a combined physical Delta pumping capacity of about 15,500 cfs, but less permitted capacity) often store water in the off-stream San Luis Reservoir (2.1 maf capacity). Total water project diversions vary considerably by year and season (Figures 6-10). Importing water users also have access to several million acre-feet of groundwater and surface water storage capacity in the southern Central Valley, Southern California, and the Santa Clara Valley to help even out supply availability.

The large southern Delta pumping plants often create reverse net flows (compared to their natural net flow directions) in Old and Middle rivers and other parts of the Delta. The highest pumping rates can even reverse the ebb tide flow, creating uni-directional flow for days or weeks, resulting in concern for native fish species, particularly during migration periods. South Delta pumping brings large quantities of higher-quality Sacramento River water into the southern Delta that benefits water exporters as well as local urban and agricultural diverters. San Joaquin River inflows are generally small, especially in the irrigation season, and often are less than local agricultural diversions in the south Delta in summer.

Various agencies and court decisions limit Delta water supply operations. The State Water Resources Control Board (SWRCB) is the major state agency that regulates in-Delta and upstream water diversions. Federal agencies and court rulings primarily establish endangered species limits on water operations. Regulations on Delta operations include limits on pumping.
rates (usually established by water rights or fish agencies), standards for Delta flows and salinity at various locations to protect Delta water diversion quality, and additional regulations to protect native fish species (SWRCB 2010). These regulations include minimum outflow requirements, regulation of outflows reflected in the location of the 2,000 mg L\(^{-1}\) (or 2 psu near-bed salinity) salinity concentration (so-called X2 requirements), ratios of exports to inflows, reverse flows in Old and Middle rivers, and salinity conditions at particular locations in the Delta (SWRCB 2010).

Changes in regulatory standards in recent years and decades have reduced the reliability of Delta water exports. Early restrictions by the SWRCB were for water quality—primarily salinity—in the western and central Delta. The Federal listing of several Delta fish species as threatened or endangered has added legal restrictions on water export pumping, particularly seasonal limits on negative flows in Old and Middle Rivers in the Central Delta, according to Biological Opinions established by federal fish agencies (the National Oceanic and Atmospheric Administration [NOAA] and the U.S. Fish and Wildlife Service [USFWS]). SWRCB permitting, water quality, and minimum flow conditions now address a broader range of environmental objectives. For those managing Delta project diversions or seeking to transfer water across the Delta, project pumping is reduced or limited by a combination of: (1) Delta outflow requirements, (2) additional Delta outflow required to increase south Delta exports (“carriage water”), (3) required releases from storage from SWP and CVP storage reservoirs under their Coordinated Operating Agreement, and (4) other salinity, flow, and endangered species regulatory limits (both internal Delta flows and Delta outflows). Since 2007, newer environmental restrictions have greatly limited water exporter’s abilities to capture unregulated winter inflows to the Delta, and forced the more aggressive operation of reservoir storage to supply water for Delta diversions (SWRCB 2010).

In drought, some environmental flow requirements are relaxed, and additional limitations emerge as a result of water right limits and curtailments. Future water exports from the Delta could be shaped as much or more by the health of native fish species as by climate, drought, and water demands.

WATER DELIVERY RELIABILITY ESTIMATION

The reliability of water supplies for agriculture and cities is often difficult given California’s highly variable hydrology, and its complex environmental and water rights regulations. Given the system’s complexity and the wide range of conditions it must prepare for, the reliability of water deliveries are estimated using computer-based simulation models (USBR 2016). The most routine estimation of water delivery reliability directly from the Delta is the California Department of Water Resources’ biannual SWP delivery capability report series (CDWR 2015).

Traditional modeling estimation of reliability employs a historical record of unimpaired streamflows (perhaps modified for climate change) to represent hydrologic variability. These flows are then entered into a simulation (computer) model that includes representations of water demands, regulations, operating policies, and infrastructure for a specified current or projected future time. As an example, Figure 8 shows the simulated volume of total Delta water project diversions that could occur in each water hydrologic year shown on the x-axis, assuming 2015 regulatory and water demand conditions and the existing and proposed infrastructure and operation of major Delta water projects. The resulting water delivery volumes are then plotted in terms of their frequency, as in Figure 9. For example, with existing facilities and operation, total Delta water project diversions are predicted to exceed 5 maf yr\(^{-1}\) in about 56\% of years, whereas under the BDCP facilities and operation proposal (similar to the more recent WaterFix proposal), exports exceed this amount in about 62\% of years. In these examples, the proposed alternative changes represented in the figures would raise the ability to divert water from the Delta in wetter years, and decrease water exports in about 25\% of drier years compared with current infrastructure (CDWR 2015).

Figure 9 also includes levels of Delta project exports that actually occurred during the severe drought years of 2014 and 2015. These are substantially below the worst levels of exports expected from the simulation modeling, which omitted the easing of Delta outflow requirements during these drought years. Although modeling of water supply reliability is important and offers insights to decision-makers,
Figure 8  Total Delta water project deliveries (SWP, CVP, and CCWD) estimated over historical hydrologic conditions (water years) with 2015 level of development and regulations (black solid line with diamond markers) and with BDCP Alternative 4 H3 conditions. (Source: Data from CDWR 2015.)

Figure 9  Estimated probability distribution of total Delta project water deliveries (CVP, SWP, and CCWD) for 2015 and BDCP Alternative 4 H3 conditions. (Source: Data from CDWR 2015.)
actual results are likely to vary, and the results of different modeling efforts are unlikely to entirely agree.

The original plans and contracts for the SWP included additional project facilities (particularly a large 22,000 cfs peripheral canal around the Delta) to increase water exports. With additional environmental regulations in recent decades, the reliability of Delta export water deliveries has deteriorated substantially. These changes from expectations in the late 1970s and early 1980s have led to the use of a more integrated portfolio of water management alternatives—especially groundwater use and banking, conservation, and water markets—to bolster reliability. For agriculture in the southern Central Valley, the loss of Delta exports has led to increased groundwater overdraft.  

WATER QUALITY

Water quality is a critical aspect of water delivered from the Delta. Contaminants of major concern include salinity and disinfection by-product precursors (especially bromide and dissolved organic carbon) (CCWD et al. 2005; Chen et al. 2010). Salinity affects crop yields in the Delta and in water-importing areas, and limits urban use and reuse of some Delta waters (Medellín-Azuara et al. 2014, 2008; Shoups et al. 2005). Removing dissolved salt from diverted water would require expensive water treatment for urban users and would be prohibitively expensive for most agriculture.

<table>
<thead>
<tr>
<th>Location/ Constituent</th>
<th>Mean TDS (mg L⁻¹)</th>
<th>Mean EC (µS cm⁻¹)</th>
<th>Mean Chloride (mg L⁻¹)</th>
<th>Mean Bromide (mg L⁻¹)</th>
<th>Mean DOC (mg L⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sacramento River at Greens Landing</td>
<td>100</td>
<td>160</td>
<td>7</td>
<td>0.018</td>
<td>2.5</td>
</tr>
<tr>
<td>North Bay Aqueduct at Barker Slough</td>
<td>192</td>
<td>332</td>
<td>26</td>
<td>0.015</td>
<td>5.3</td>
</tr>
<tr>
<td>SWP Clifton Court Forebay</td>
<td>286</td>
<td>476</td>
<td>77</td>
<td>0.269</td>
<td>4.0</td>
</tr>
<tr>
<td>CVP Tracy Pumping Plant</td>
<td>258</td>
<td>482</td>
<td>81</td>
<td>0.269</td>
<td>3.7</td>
</tr>
<tr>
<td>CCWD Intake at Rock Slough</td>
<td>305</td>
<td>553</td>
<td>109</td>
<td>0.455</td>
<td>3.4</td>
</tr>
<tr>
<td>San Joaquin River at Vernalis</td>
<td>459</td>
<td>749</td>
<td>102</td>
<td>0.313</td>
<td>3.9</td>
</tr>
</tbody>
</table>

Source and notes: CCWD et al. 2005, Sampling period varies, but generally is between 1990 and 1998. mg L⁻¹ = milligrams per liter, TDS = total dissolved solids, EC = electrical conductivity, µS cm⁻¹ = microSiemens per centimeter, DOC = dissolved organic carbon.

Contaminants and water quality for urban and agricultural water supplies vary considerably in the Delta with location and over time (Table 1, Figure 10). Organic carbon in Delta waters is from drainage of peat soils in the Delta and other sources upstream (CCWD et al. 2005; http://www.water.ca.gov/waterquality/drinkingwater/). Bromide is mostly from sea water, and secondarily from San Joaquin River drainage.

Major sources of salt in Delta waters include sea water from San Francisco Bay and agricultural drainage from the San Joaquin River, as well as some salts from upper watersheds and other agricultural and urban wastewaters. Agriculture in the Delta concentrates salt from irrigation water and discharges it back into the Delta. (http://www.water.ca.gov/waterquality/drinkingwater/; CCWD et al. 2005) Delta outflows remove most salt from the Delta. Export pumping removes some salt, to the detriment of export water quality. As sea levels rise, more Delta outflow or changes in Bay and Delta geometry will be needed to repel ocean salinity (Fleenor et al. 2008).

ECONOMIC MOTIVATION FOR DELTA WATER AND LAND MANAGEMENT

Most human management of water and land in the Delta supports the economic purposes of land owners and water users. Most water diverted from the Delta upstream and within the Delta is for commercial agriculture, with almost all remaining diversions...
being for urban uses, as summarized in Table 2 (Lund et al. 2010).

The demand for water from the Delta is significantly affected by changes in water users and the availability and cost of other water supplies. Higher agricultural commodity prices bring pressure to increase crop production and irrigation. Growing populations displace some agricultural land and irrigation, but can increase urban water demand, although at a decreasing per capita rate depending on urban water conservation, landscaping, and population densities. Reductions in per-capita urban water use will continue to dampen growth in urban water use, and perhaps reduce it a bit (Wilson et al. 2016). For southern California and the San Francisco Bay Area, which are the largest urban users of Delta water, non-Delta supplies include generally expensive options for wastewater reuse, local stormwater collection, brackish and ocean water desalination, less expensive brackish desalination, and investments and costs for water conservation. For the southern Central Valley, alternative water supplies are quite scarce, given its dry climate, immense agricultural water demands, and the long-standing natural reuse of most return flows, wastewater and stormwater in local aquifers and streams. As net groundwater use in the southern Central Valley is reduced to eliminate groundwater overdraft, some water uses will be discontinued and demand will likely increase for other water sources, especially the Delta (Nelson et al. 2016; Dogan 2015).

Tanaka et al. (2011) examined the costs of changes in Delta exports in terms of scarcity costs from diminished water use and operating costs for alternative water supplies. Although these annual costs could be as high as $3 billion per year for ending Delta exports entirely, California’s water system appears able to withstand some significant changes in Delta water availability, albeit at substantial expense. Sunding (2013) also estimated the economic benefits of improvements in water supply reliability for some long-term Delta conveyance options as part of the Bay Delta Conservation Plan (BDCP) effort. The economic effects of reduced Delta water availability are dampened by the allocation of scarce water to the most economically valuable agricultural and urban uses by farmers, water utilities, and water markets.

The development of land with levees and drainage systems for farming and towns also has changed water flows and reduced wetlands and aquatic habitat (Medellin–Azuara et al. 2012). Historical Delta land reclamation for commercial agriculture reshaped the Delta, and continues to drive how Delta...
lands are managed (Thompson 1957). Agricultural land in the Delta is often sold for $4,000 to $10,000 per acre, less than much of California’s agricultural land, although urban land has much higher economic value (California ASFMRA 2013). (For example, Metropolitan Water District of Southern California (MWDSC) recently bought 20,000 acres on four subsided islands for $8,500 per acre.) Some subsided Delta lands have been abandoned because of high seepage and costs and risks of levee repair compared to the revenues available from these lands (Thompson 1957; Deverel et al. 2015). Upstream leveeing and drainage of lands have brought several other problems through accelerated flood waves, reduced seasonal habitat for migratory fish and birds, reduced supplies of sediment to the Delta (Wright and Schoellhamer 2004), and reduced some potential for groundwater recharge. The economics of land and water use in the Delta and elsewhere drives most Delta management.

**Drought**

Droughts have always brought innovations for water management in California, such as the development of irrigation systems in the late 1800s, the development of major dams after the 1930s drought; the development of urban water conservation, water markets, and integrated urban water planning from droughts in the 1970s and 1980s; and the development of new Delta governance and some improvements in water conservation and data from the 2009 drought. The current drought has similarly tested the current water system, highlighting weaknesses, bringing focus and attention to problems, and motivating solutions. So far, this most recent drought has led to major legislation and policy changes statewide in groundwater overdraft, urban water conservation, and water rights administration (Hanak et al. 2015).

For the Delta, the current drought has brought consideration of salinity barriers for several parts of the Delta (the first since 1977), installation of a major salinity barrier at False River in 2015, reduced freshwater inflows; greatly reduced Delta water exports; lower counts of Delta Smelt and other native fishes; expanded areas of water hyacinth, Brazilian waterweed (*Egeria densa*) and other invasive species (clams, fish, other plants, phytoplankton, and zooplankton), and greater attention to the Delta overall. This drought also has brought higher temperatures in the Delta, a likely precursor of future climate warming (Williams et al. 2015), with likely effects on ecosystems, water quality, and perhaps on human water demands.

Direct water diversions from the Delta fell dramatically during this drought, with levels not

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**Table 2** Gross water diversions from the Delta and economic value (about 1998–2009)

<table>
<thead>
<tr>
<th>Use</th>
<th>Average annual gross use (taf yr⁻¹)</th>
<th>Typical incremental economic value ($ af⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upstream diversions for agriculture</td>
<td>9,650</td>
<td>$100–$280</td>
</tr>
<tr>
<td>Upstream diversions for cities</td>
<td>1,450</td>
<td>$500+</td>
</tr>
<tr>
<td>In-Delta agriculture</td>
<td>1,150</td>
<td>$120–$180</td>
</tr>
<tr>
<td>In-Delta cities</td>
<td>150</td>
<td>$500+</td>
</tr>
<tr>
<td>Export agriculture</td>
<td>3,400</td>
<td>$150–$550</td>
</tr>
<tr>
<td>Export cities</td>
<td>1,700</td>
<td>$500+</td>
</tr>
<tr>
<td>Total upstream</td>
<td>11,100</td>
<td>—</td>
</tr>
<tr>
<td>Total in-Delta</td>
<td>1,300</td>
<td>—</td>
</tr>
<tr>
<td>Total export</td>
<td>5,430</td>
<td>—</td>
</tr>
<tr>
<td>Total diversions</td>
<td>17,830</td>
<td>—</td>
</tr>
<tr>
<td>Total Delta outflow</td>
<td>17,140</td>
<td>—</td>
</tr>
</tbody>
</table>

a. Includes San Francisco Public Utilities Commission (SFPUC) and EBMUD diversions.

Sources: CDWR, C2VSim, Dayflow, and CALVIN and SWAP model results.
seen since the 1976–77 drought. These reductions are more than would be anticipated based on water delivery capability studies (Figures 9 and 10), and reflect some of the difficult-to-predict problems and opportunities involved in real-time drought management. For water users in southern California, the southern Central Valley, and San Francisco Bay Area, the drought led to serious and sometimes unprecedented cut-backs in Delta supplies and much higher market values for water (Howitt et al. 2015). The CVP was unable to meet all exchange contractor demands, and many contractors were given zero allocations for the first time (http://www.usbr.gov/mp/cvo/deliv.html).

Economically, most in-Delta, upstream, and water importing areas have managed well during the drought, with economic suffering far less than the proportions of lost water supply (Medellin–Azuara et al. 2015; Howitt et al. 2015; Hanak et al. 2015). Access to groundwater, system flexibility, water market transfers and exchanges, and preparation create much of this robustness. New laws requiring groundwater users to implement sustainability plans should improve long-term water supply reliability, but require some additional land falling (Nelson et al. 2016). Better and more comprehensive accounting of consumptive use, employing remote sensing, or other cost-effective approaches in the Delta and statewide, could improve management. The drought experience indicates that short interruptions of Delta supplies, if properly prepared for and not too frequent, can be endured economically with sizable local yet modest statewide costs (Howitt et al. 2015).

Drought effects on water quality and ecosystems have not yet been well investigated. It seems likely that the higher temperatures during this drought and the longer residence times resulting from reduced inflows, reduced Delta project pumping, and installation of a major Delta barrier will have affected many aspects of Delta water quality and the movement and success of different aquatic plants and animals. These effects might ultimately affect the quantity and quality of water supply deliveries.

**CHALLENGES FOR DELTA WATER SUPPLIES**

For decades, water diversions upstream or within the Delta have been the primary source for expanding agricultural and urban water use in California (Figures 11 and 12). Water use upstream of the Delta continues to grow (Figures 6 and 12). But, after decades of continuous increases—interrupted only by occasional droughts (Figures 6 and 7)—recent years have seen large reductions in Delta water exports, because of drought and environmental regulations.

The current Delta water supply system is unsustainable. Climate change, rising sea levels, and additional environmental regulations can be expected to reduce the ability of Delta water projects to divert water from the Delta (Anderson et al. 2008; Fleenor et al. 2008). Fleenor et al. (2008) estimated that a 1-foot sea level rise would require almost 500,000 acre ft yr⁻¹ of additional Delta outflow to meet salinity requirements, about a 10% reduction in overall Delta project diversions. Larger sea level rises, a warming climate, and drier extremes would increase difficulties in sustaining
upstream and in-Delta efforts to restore native ecosystems; diminishing supplies from Delta tributaries, the Colorado River, and other water sources; growing profitability of agriculture; and growing urban populations (Wilson et al. 2016; Nelson et al. 2016; Dogan 2015). Agriculture and cities in the southern Central Valley currently overdraft groundwater by 1 to 2 maf yr\(^{-1}\) as a major water source (Faunt et al. 2009). The growing economic value of Central Valley agriculture and state policy to end groundwater overdraft under the Sustainable Groundwater Management Act (SGMA), will raise economic demands for additional water from the Delta and upstream sources (Nelson et al., 2016; Dogan 2015). Greater urban water use efficiencies, local water supply development, reduced irrigated land area from salinization and urbanization, greater water rights enforcement, and recent Delta diversion levels. Figure 13 shows the results of a federal study that examined change and variability in total Delta exports for different climate conditions (USBR 2016), with climate change more likely to reduce Delta exports. Most climate change predictions show some loss of runoff for California, if only from additional evapotranspiration from warmer watersheds, particularly south of the Delta (Woodhouse et al. 2016; Null et al. 2010). Increased wetland surface area from habitat restoration or inundation with sea level rise, might somewhat increase evaporative losses of freshwater and reduce available water supply, depending on pre-restoration land and water use.

Figure 12 Historical Delta water availability, diversions, and outflows. Note the discontinuity in the later 1990s, probably reflecting use of incompatible water accounting systems. (Source: California Water Plan 2013, Figure D7.)

<table>
<thead>
<tr>
<th>Period</th>
<th>Average Annual Total (MAF)</th>
<th>Outflow</th>
<th>in-Delta</th>
<th>Exports</th>
<th>Delta Watershed</th>
</tr>
</thead>
<tbody>
<tr>
<td>1930 to 1949</td>
<td>25.80</td>
<td>81%</td>
<td>5%</td>
<td>0%</td>
<td>14%</td>
</tr>
<tr>
<td>1960 to 2005</td>
<td>31.71</td>
<td>67%</td>
<td>4%</td>
<td>4%</td>
<td>24%</td>
</tr>
<tr>
<td>1950 to 1969</td>
<td>34.34</td>
<td>51%</td>
<td>5%</td>
<td>15%</td>
<td>29%</td>
</tr>
<tr>
<td>1970 to 1989</td>
<td>32.65</td>
<td>46%</td>
<td>4%</td>
<td>17%</td>
<td>31%</td>
</tr>
</tbody>
</table>

Note: Data from 1930–1997 comes from the Delta Vision Blue Ribbon Task Force 2008 Final Report and data.
environmental regulations will somewhat reduce overall growth in Delta water demands.

The physical reliability of water quantity and quality for diversion is threatened by increasing water demands for ecosystem management and growing numbers of endangered species (which reduce flexibility in water diversion operations), risks of levee failure from subsidence, floods, earthquakes and sea level rise, more stringent water quality and drinking water standards, increasing upstream water diversions, and climate warming (Lund et al. 2010; Anderson et al. 2008). If the trajectory of recent decades continues, the average quantity of water available for diversion from the Delta will likely diminish, particularly in drier years (Figure 13).

As water becomes less available from the Delta, the incremental economic value of water diversions from the Delta will likely rise. Overall, economic demand for Delta diversions seems likely to grow, while inflow to the Delta seems likely to decrease. Delta management will attract growing scrutiny, and, as a result, continuing to expand exports from the Delta could entail increasing difficulties.

![Figure 13 Annual exceedance plot of total Delta exports for an ensemble of climate scenarios with current trends socioeconomic conditions. (Source: USBR 2016.)](image)

**OPTIONS AND OPPORTUNITIES FOR DELTA WATER SUPPLIES**

Many suggestions have been made to address these changes in water supply reliability for and from the Delta. Water conservation, wastewater reuse, water markets, desalination, conjunctive use of surface and groundwater, surface storage expansion, shifting Colorado River water to urban uses, and changes in Delta water diversion infrastructure (including major diversion tunnels) have been suggested most seriously (CDWR 2013, 2015b). Many of these activities are being implemented, some with considerable success.

Given growing upstream uses and water uses for ecosystems and ending groundwater overdraft, greatly expanding Delta and upstream water diversions would involve great expense in increasing export conveyance and storage capacities for increasingly rare and diminished “surplus” water in wet years. Modest expansions—and even retention—of existing Delta diversion quantities, relative to “No Action” cases, are envisioned under WaterFix, but require expensive and controversial changes in regulations and infrastructure for moving water across the Delta in coordination with upstream and downstream water storage (CDWR 2015b). Water availability and regulations seem likely to limit the ability to use expanded storage and conveyance infrastructure under most hydrologic conditions (Lund et al. 2014).

So far, the greatest successes have come from a portfolio approach to water management, initiated locally and regionally. An orchestrated portfolio of water supply and demand management actions and incentive policies often provides more reliable, economical and environmentally-effective performance (Table 3) (MWDSC 2015; SCVWD 2010). Such a mix of actions and policies also increases flexibility for water systems to adapt to changing conditions, such as droughts, new environmental regulations, and climate change. This more integrated resource management approach often involves internal and external partners who coordinate activities over a range of conditions using water markets, prices, and contracts. The success of urban areas in the current drought is substantially as a result of these mostly local and regional activities.
Agriculture also has benefitted during the drought from these actions, but has relied more on pumping groundwater (Howitt et al. 2015).

Even with a portfolio approach, the Delta faces strategic issues of cross-Delta water conveyance, levees, and ecosystem management, all of which affect water supply reliability. The Delta might be California’s best example of how major changes in basic conveyance infrastructure, changed diversion locations, operations, channel geometries, etc., if properly done, have potential to provide both water supply and ecosystem improvements (although not necessarily for all water users and all species) (Lund et al. 2010). The effectiveness of water storage initiatives throughout the Central Valley is reduced without better ability to move water across the Delta (Lund et al. 2014; Dogan 2015). The value of cross-Delta conveyance capability increases substantially with a drier, warmer climate and the end of groundwater overdraft (Harou et al. 2010; Dogan 2015; Buck 2016).

Challenges to Delta water supplies are substantial and growing, and will require improvements in scientific and technical analysis and information for local, state, and federal decision-making. Changing conditions and a wide range of long-term and short-term options make it important to realistically and quantitatively assess mixed alternatives for the Delta to supply water in the future.

### ANALYTICAL CAPACITY

Water agencies in California spend substantially on analysis of water problems and management. A major difficulty in water supply management and analysis is the lack of a common and complete water accounting and model development across state agencies and programs (Escriva–Bou et al. 2016; Grantham and Viers 2014). Water use by surface water rights holders has been reported poorly, if at all, until very recently. Only applied and not consumptive use is reported. Groundwater use remains unreported, and is estimated partially, using different models and water balances. Return flows from users back to the system are coarsely and differently estimated by different agencies. Lacking a common authoritative framework to account for water use and availability, different state programs develop different partial accounting frameworks and models, conflicts deepen, and transparency diminishes. A common accounting framework and improved data collection would improve the usefulness and transparency of water data for analysis, enforcement, and management.

Most regional and many local water agencies maintain and use computer models of their water systems to provide a more coherent picture of their water use and management. State and federal agencies maintain dozens of computer models of water management and water availability, with little coordination.

### Table 3 Elements of modern water supply system portfolios

<table>
<thead>
<tr>
<th>Supply</th>
<th>Demand Management</th>
</tr>
</thead>
<tbody>
<tr>
<td>• System re-operation</td>
<td></td>
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<tr>
<td>• Surface and groundwater reservoirs</td>
<td></td>
</tr>
<tr>
<td>• Conveyance and interconnection</td>
<td></td>
</tr>
<tr>
<td>• Conjunctive use of surface and groundwater (recharge and pumping)</td>
<td></td>
</tr>
<tr>
<td>• Expanded conveyance &amp; storage</td>
<td></td>
</tr>
<tr>
<td>• Urban reuse</td>
<td></td>
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<tr>
<td>• New water treatment</td>
<td></td>
</tr>
<tr>
<td>• Wastewater reuse</td>
<td></td>
</tr>
<tr>
<td>• Ocean or brackish desalination</td>
<td></td>
</tr>
<tr>
<td>• Contaminated aquifers</td>
<td></td>
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<tr>
<td>• Stormwater capture</td>
<td></td>
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<tr>
<td>• Source protection</td>
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<table>
<thead>
<tr>
<th></th>
<th>Incentive Policies</th>
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</thead>
<tbody>
<tr>
<td>• Agricultural water use efficiencies, recharge, and reductions</td>
<td></td>
</tr>
<tr>
<td>• Urban water use efficiencies (e.g., plumbing codes) and reductions</td>
<td></td>
</tr>
<tr>
<td>• Ecosystem water use efficiencies and allocations</td>
<td></td>
</tr>
<tr>
<td>• Pricing</td>
<td></td>
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<tr>
<td>• Markets and exchanges</td>
<td></td>
</tr>
<tr>
<td>• Subsidies, taxes</td>
<td></td>
</tr>
<tr>
<td>• Education</td>
<td></td>
</tr>
</tbody>
</table>
Modeling efforts have focused predominantly on surface water, and been rooted in traditional infrastructure development and operations, despite the growing importance of groundwater, conjunctive use of surface and groundwater, water conservation, and water markets. Most water supply analysis has been based on comparison of results with and without proposed changes in facilities or operations. This comparative approach was attractive in the era of large-scale infrastructure development, because it required less attention to model testing against field data, but is unlikely to serve the state well in the future (Close et al. 2003). The tighter water management required for implementing the Sustainable Groundwater Management Act (SGMA), water rights curtailment during drought, and environmental flows imply a performance standard for water supply modeling success based on field conditions, rather than comparisons of model results.

Today, two modern groundwater models exist for the Central Valley: the state’s California Central Valley Groundwater–Surface Water Simulation Model (C2VSIM) (Brush et al. 2013) and the U.S. Geological Survey’s Central Valley Hydrological Model (CVHM) (Faunt 2009). Despite their great advances over previous models, large differences remain between them, mostly reflecting fundamental uncertainties in estimating water availability and use. Implementing the state’s new Sustainable Groundwater Management Act would seem to require reconciling and improving these representations, which also will affect understanding Delta water availability and demands, and aid in managing environmental flows and surface water rights.

Unfortunately, fragmentation, inconsistency, and poor or delayed documentation of modeling, both among and within state agencies, often leads to opacity and confusion. For water supply system models, the state Department of Water Resources Modeling Support Branch has the CALSIM II, unreleased CALSIM III (now nearly 10 years in development), and CalLite models (Draper et al. 2004; Islam et al. 2011). The same Department’s Planning branch has a proprietary WEAP (Water Evaluation And Planning) model for the California Water Plan. Operators of the State Water Project employ both a large spreadsheet model and their own different version of CalSim II, which are not publicly available. The State Water Resources Control Board employs a different WEAP model for developing environmental flow criteria. The federal Bureau of Reclamation employs the CALSIM models (which it co-develops with CDWR), but has a separate version of CalLite which runs on a different proprietary software platform. Each of these models involves different groups of consulting hydrologists and engineers, often with their own model versions and without substantial documentation. Although this diversity of modeling capability supports a range of modeling approaches, the degree of fragmentation and compartmentalization of information seems to contribute to the opacity and expense of Delta water supply operations modeling and discussions, and diminished development and use of insights from modeling analysis. The difficulty in modifying CALSIM II to analyze new scenarios has led some consultants to develop spreadsheet-based models of the CVP–SWP system.

Typically, model simulations are reasonably reliable for near-average hydrologic conditions, but are usually less reliable for extreme hydrologic conditions or finer time-steps and locations. Simulated operations in drought, for example, often show unrealistically low storage levels while maintaining deliveries, but actual operators commonly hedge to store more water (delivering less) in case drought conditions worsen. These deficiencies in modeling may lead to errors in identifying and estimating project benefits and beneficiaries in a comparative analysis of with and without project conditions.

The California Water and Environment Modeling Forum (CWEMF) has fostered useful conversations and presentations on model and data development among technical audiences, including external reviews of some models. However, this largely voluntary effort has not yet been effective in fostering systematic coordination and quality improvements in model and data development and applications. The many state agencies and programs have no common discussion or clear strategic vision of the models and data needed and how they should be developed and maintained to support water management and policy. As the need for more common and widely-accepted water accounting becomes more important, state agencies need to lead in developing a common multi-agency technical data and modeling program.
Modeling results will inevitably differ. Modeling results for BDCP alternatives in 2013 often differed by 200 to 700 taf yr\(^{-1}\) in average water deliveries and flows, when evaluated by different models and modelers (MBK Engineers and Dan Steiner 2014). This reflected differences in model errors and differences in the professional judgments of the consultants. This divergence also reflects something of the uncertainties inherent in using model runs to estimate water supply performance over long planning horizons. The economic incentives to contest even unavoidable modeling error are apparent when a modeling difference of 10 taf yr\(^{-1}\) for water valued at $300 af, is worth $3 million per year or a long-term, present value of $60 million at 5% interest.

The size and unavoidability of technical uncertainty and variability in water availability is illustrated by statistical uncertainty in the average flow of the Sacramento River, California’s largest river. The measured long-term average annual flow of the Sacramento River is 21.7 maf, but there is a 32% statistical chance that the true long-term average differs more than 1 maf yr\(^{-1}\) from this value.\(^2\) Such unavoidable uncertainty has considerable economic value.

Water policy and management in California, and the Delta, must deal with sizable and unavoidable uncertainty. To discourage unnecessary and counterproductive squabbles, state regulators and water rights administrators must act with reasonable and firm authority and due process. This reasonableness must include adequate dedication to continuous improvement in modeling, data, and water accounting. Technical information and insights should be developed transparently, better organized, and better articulated and documented. Decision-making processes should be organized to better support and assimilate the development, analysis, and discussion of promising technical and policy alternatives.

\(^2\) With an average annual flow of 21.7 maf, and a standard deviation of observed annual flow of 10 maf over roughly 100-years of record, the standard deviation of the mean is 1 maf/yr ($\pm SD_{\text{sample}}/\sqrt{n}$), so there is a 68% confidence range the true mean annual flow of the Sacramento River is 20.7-22.7 maf/yr, and a 92% chance that the average is more than 100 taf in error, neglecting climate change.

### OBSERVATIONS AND DIRECTIONS FOR WATER SUPPLY SCIENCE AND ANALYSIS

Several observations, questions, and directions for further work seem apparent for policy-makers, scientists, and technical managers.

1. California is a mostly dry place where climate and geography have motivated development of extensive water supply and flood management infrastructure to serve economic purposes of human settlement and agricultural production. This infrastructure now also must serve newer, and sometimes conflicting, environmental objectives.

2. The Sacramento–San Joaquin Delta is California’s major hub for transferring water from wetter to drier regions. Outflows from the Delta have been reduced mostly by diverse upstream diversions and secondarily by in-Delta diversions, which are mostly from state and federal water projects (SWP and CVP) (Figures 6 and 13). Internal flows in the Delta are significantly affected by channel geometries and the location and operations of major water project pumping plants.

3. California’s nearly statewide water network makes most of the state interdependent for water supply, water quality, flood management, and ecosystem performance, including remote upstream areas and water users in the Bay Area, southern California, the Central Valley, and particularly the Delta. Local interests tend to protect local interests and avoid comprehensive solutions (Madani and Lund 2012). Statewide efforts are needed to help bring local interests together while regional and statewide interests are addressed.

4. The complexity and diversity of California’s water supply system makes it rich in physical possibilities for management—and remarkably reliable, robust, and adaptable to change and disruptions (Tanaka et al. 2011; Harou et al. 2010; Medellín–Azuara et al 2015; Howitt et al. 2015). Many agencies and the state benefit from diverse portfolios of water sources, water use management, and coordinated operations to improve water supply reliability, reduce costs, and reduce effects on Delta and upstream ecosystems. For example, when a SWP contractor
in northern California conserves water or is protected from floods by a state reservoir, the water retained becomes available for others in the SWP, including those in Southern California.

5. The Delta, as a hub, is perhaps the water system’s most vulnerable component (Lund et al. 2010; Tanaka 2011; Dogan 2015; Buck 2016). Its management and condition often affect economic and ecosystem performance throughout California. The management of such a central element will always engender controversies and conflicts among upstream, in-Delta, and water-importing regions.

6. The failure to authoritatively and systematically quantify actual water rights, diversions, and return flows (including groundwater flows) is a major shortcoming and hinders management of water supplies (Escriva–Bou et al. 2016).

7. Fragmentation of water measurement, accounting, and analysis often leaves the state with inconsistent, insufficient, and often incomprehensible analyses of water management performance and alternatives.

Some directions for scientific and technical work on the reliability of water supplies taken directly and indirectly from the Delta include:

1. Better and more systematic estimates are needed of water supply reliability, given many ongoing and impending changes from sea level rise, climate change, Delta levee conditions, worsening conditions of native species, upstream diversions, and difficulties making decisions with a myriad of governmental agencies and interests. This challenge calls for the integrated attention of physical, social, and biological sciences and engineering—and more effective coordination among different water agencies, uses, interests, sources, and facilities.

2. More explicit estimates of trade-offs among California’s water supply and environmental objectives for the Delta would better inform policy discussions. Delta conflicts are not only between water supply reliability and ecosystem health, they also involve in-Delta flood safety, property protection, land development, recreation, and other local issues. Water supply interests often involve different upstream, in-Delta, and water export diversions and water quality. Ecosystem interests are spread across recreational fisheries and upstream, anadromous, and resident native species. In some cases, it might be best to pursue compromise–reconciliation approaches to jointly provide ecosystem and economic benefits from land and water use (Moyle 2015; Rosenzweig 2003).

3. Explicit analysis and comparison of portfolios of water management activities would better inform governmental agencies and interests of the trade-offs from different management activities and policies. Such work should identify more promising sets of activities and give insights about their selection, implementation, and adaptive management as conditions change. This approach should explicitly integrate economic and environmental performance objectives using models that integrate Delta water management and groundwater and surface water management in and outside of the Delta. Many components for such analyses are available, but need to be made more transparent and accessible, technically integrated, and consistently supported across agencies.

4. An authoritative water accounting framework is needed for the state to quantify water rights, diversions, and discharges or return flows to support management of the Delta, groundwater, environmental flows, and surface water (Escriva–Bou et al. 2016). This will require development of more authoritative means for estimating water balance components as part of such a framework, such as evapotranspiration, groundwater, etc. This framework must be common across state agencies.

5. Better organization and documentation of data and analysis, and more consistent and authoritative development and availability of such data are needed (Escriva–Bou et al. 2016). Information for water accounting remains underdeveloped and often incoherent. A disadvantage of decentralized water management and governance is the fragmentation of data, information, and analyses. More common documentation and data management would be helpful overall, and make analyses more comparable and insightful.
6. The sociology of water problems is often harder than their more purely technical aspects. More explicit thinking about the organization of Delta governance and analysis might be helpful. Game theory has some potential to improve understanding of Delta management and water supply conflicts and for crafting more durable adaptive management strategies (Madani and Lund 2012).

Organizing Delta discussions and policy processes to be more data-driven and performance-driven might be helpful. Providing information for such discussions is a major problem for technical managers in the state—and starts with reporting water use. Agency decision-making processes also need to be organized to better ensure integrated scientific and technical results, and insights into their deliberations.

CONCLUSIONS

All water supply systems require cooperation and engender some forms of conflict. This game theory aspect applies to water supply reliability and the Sacramento–San Joaquin Delta as well. The many conflicts of the Delta reflect the differing benefits to local and statewide interests from a functioning Delta. Scientific and technical work on the Delta and its management, which have implications for water supply reliability, can better highlight the benefits to all interests from cooperation and relative trade-offs in management and policy. Although better technical and scientific work should aid in the difficult strategic and operational decision-making needed for the Delta, it cannot fully overcome the fundamental trade-offs necessary for decisions involving different benefits and costs to different interests. Decision-makers must actively seek and support scientific insights to inform these decisions. Developing and using science for decision-making requires changes in both science and decision-making.

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Factors and Processes Affecting Delta Levee System Vulnerability

Steven J. Deverel,1 Sandra Bachand,2 Scott J. Brandenberg,3 Cathleen E. Jones,4 Jonathan P. Stewart,3 and Paolo Zimmaro3

ABSTRACT

We appraised factors and processes related to human activities and high water, subsidence, and seismicity. Farming and drainage of peat soils caused subsidence, which contributed to levee internal failures. Subsidence rates decreased with time, but still contributed to levee instability. Modeling changes in seepage and static slope instability suggests an increased probability of failure with decreasing peat thickness. Additional data is needed to assess the spatial and temporal effects of subsidence from peat thinning and deformation. Large-scale, state investment in levee upgrades (> $700 million since the mid-1970s) has increased conformance with applicable standards; however, accounts conflict about corresponding reductions in the number of failures.

Modeling and history suggest that projected increases in high-flow frequency associated with climate change will increase the rate of levee failures. Quantifying this increased threat requires further research. A reappraisal of seismic threats resulted in updated ground motion estimates for multiple faults and earthquake-occurrence frequencies. Estimated ground motions are large enough to induce failure. The immediate seismic threat, liquefaction, is the sudden loss of strength from an increase in the pressure of the pore fluid and the corresponding loss of inter-particle contact forces. However, levees damaged during an earthquake that do not immediately fail may eventually breach. Key sources of uncertainty include occurrence frequencies and magnitudes, localized ground motions, and data for liquefaction potential.

Estimates of the consequences of future levee failure range up to multiple billions of dollars. Analysis of future risks will benefit from improved description of levee upgrades and strength as well as consideration of subsidence, the effects of climate change, and earthquake threats. Levee habitat ecosystem benefits in this highly altered system are few. Better recognition and coordination is needed among the creation of high-value habitat, levee needs, and costs and benefits of levee improvements and breaches.
INTRODUCTION

Since the 1860s, Delta levees have been rebuilt and strengthened in response to failures and threats of failure. Resource availability and subsidence have hampered efforts to repair and reinforce levees. The present-day 1,800-km levee system built largely upon original structures imperfectly protects subsided islands and associated agriculture, homes, urban areas, and infrastructure from the continuous threat of inundation and ensures the movement of water from northern California to areas south of the Delta, including parts of the San Francisco Bay area (Figure 1). Levees provide habitats and protect wetlands on selected Delta islands (e.g., Twitchell, Sherman) (Figure 2). The presence of levees substantially determines Delta land use.

The Delta Risk Management Strategy (DRMS) study concluded that under business as-usual practices, the Delta region is unsustainable because of threats to levee integrity (URS Corporation and Jack R. Benjamin and Associates, Inc. 2009). Levees can be reinforced and augmented to reduce risk but all risk cannot be eliminated (Arcadis 2016). Principal threats to levee stability include earthquakes, seepage, high water, and subsidence (Figure 1). The consequences of levee failure include loss of agricultural production, water supply disruption, habitat loss and species alteration, water quality changes, infrastructural destruction and disruption, and loss of life (Figure 1). Before ongoing mitigation efforts, the DRMS study estimated costs ranging into the billions of dollars would result from losses associated with levee failure and island flooding.

Moore and Shlemon (2008) previously described the modern Delta levee system. The DRMS study provided a comprehensive compilation of data and analysis related to levee risk (URS Corporation and Jack R. Benjamin & Associates, Inc. 2009). Levee science and practice have evolved since Moore and Shlemon (2008) and the DRMS study. Hundreds of millions of dollars have been spent on levee upgrades. Threats to levees and the relation of levees to aquatic and terrestrial habitat have also been further quantified. A reappraisal of key issues is necessary.

Our objective was to assess processes and factors that affect levee-system vulnerability and sustainability through a reappraisal of threats and consequences of levee failure. We also summarized ecosystem effects, and potential benefits and integration of ecosystem function; explored mitigation and monitoring; and defined critical uncertainties and information needs. We focused on three levee threats: high water, earthquakes, and subsidence.

BACKGROUND

Human activities—farming and drainage of peat soils, flood control—and levee upgrades (Figure 3) have affected the stability and failure of levee. After initial levee construction from manually-harvested peat blocks in the 1860s and 1870s, floods in 1878 and 1881 destroyed many levees and sapped much of the previous decade’s optimism for reclamation (Thompson 1957, 2006). By the 1890s, peat levees were mostly overlain by stronger and more robust levees built from mineral material extracted from adjacent channels (Figure 3). By the 1940s, reclamation of about 1800 km² of Delta land for agricultural use was complete (Thompson 1957).

The legal Delta encompasses about 3,600 km² where lowlands are surrounded by levees; about 630 km are “project” levees which are part of the State Plan of Flood Control, and about 1,200 km are “non-project” levees, owned and maintained by local levee-management agencies (Figure 4). Arcadis (2016) identified 125 Delta islands to be included in the Delta Levee Investment Strategy (DLIS) analysis. Because the majority of the land in the Delta is used for agriculture, we focused on levees that surround farmed islands.

Farming and Drainage

Drainage of peat soils for farming resulted in oxidation of organic matter, which is the primary cause of subsidence. Delta peat soils formed from decaying wetland plants and the build-up of below-ground biomass (Atwater 1982; Shlemon and Begg 1975; Drexler et al. 2009a). During the 6,000 to 7,000 years before the 1850s, about 5 billion m³ of tidal marsh sediment accumulated in the Delta, and, since the 1850s, half of this volume has disappeared (Deverel and Leighton 2010; Mount and Twiss 2005). The present-day area of peat soils is smaller than the area of historic wetlands (Figure 5) that resulted
Figure 1  Factors affecting levee stability

Figure 2  Levee and on-island habitat
Subsidence rates have decreased substantially since the first half of the twentieth century due to cessation of burning, reduced wind erosion and depletion of soil organic carbon as illustrated by the average elevation data from Bacon Island from Deverel and Leighton (2010).

Figure 3  Timeline of Delta levee related events (A) failures, (B) annual peak discharge and (C) subsidence on Bacon Island. Subsidence rates have decreased substantially since the first half of the twentieth century due to cessation of burning, reduced wind erosion and depletion of soil organic carbon as illustrated by the average elevation data from Bacon Island from Deverel and Leighton (2010).
in the formation of Delta peat soils; peat soils have disappeared close to the margins of the Delta. Much of this remnant peat is preserved epidermal parts of rhizomes of bulrush (*Schoenoplectus* spp.) and reeds (*Phragmites* spp.) (Atwater 1982).

Subsidence rates have decreased substantially since the first half of the twentieth century because of the cessation of burning, reduced wind erosion resulting from the planting of alternative crops, and depletion of soil organic carbon (Figure 3). Present-day rates range from a few mm yr$^{-1}$ to over 1.8 cm yr$^{-1}$ where elevations are $-2$ m or less (Deverel et al. 2016). Based on data from extensometers, greenhouse-gas emission measurements, and modeling, subsidence is occurring primarily where organic soils are at elevations at or below $-2$ m (Figure 6) (Deverel et al. 2016), which represents about 24% of the legal Delta or about 70,700 ha.

In his review of the history, causes, and costs of the flooding of Delta islands from the early 1900s to the 1980s, Prokopovitch (1985) concluded that levee foundation instability resulting at least partially from subsidence became an important factor in island flooding after the 1940s. Duncan and Houston (1983), CDWR (1982), and the U.S. Army Corps of Engineers (1982) generally agreed. For example, foundation failure caused 12 of 18 failures that occurred from 1950 to the early 1980s (Prokopovitch 1985). Subsidence lowered island interiors and created height differences between island surfaces and adjacent surface-water levels (Figure 3), which contributed levee internal failures. Mount and Twiss (2005) stated that during the 1950s and beyond, the percentages of failures from levee structural problems increased because of the subsidence of organic soils, which augmented hydraulic forces against levees.

**Flood Control**

The benefit for Delta levees of flood control resulting from dam construction on California’s major rivers, which began in the 1930s, is uncertain. Prokopovitch (1985) stated that before dam construction, greater numbers of levees failed from overtopping, relative to after the 1940s. The decrease in levee failure from overtopping may have been lessened because of reduced Delta inflow. McDonald et al. (2008) speculated that reservoirs increased flood protection somewhat by reducing flows. However, some of the flood attenuation reservoirs provided may have occurred anyway because of floodplain storage. Numbers of Central Valley levee breaks were not significantly different during the first half of the twentieth century (before major dam construction) compared to the last half of the twentieth century (after major dam construction) (Florsheim and Dettinger 2007).

The National Flood Insurance Act of 1968 and establishment of 100-year floodplain elevations likely provided a solid basis for levee upgrades. Before the establishment of 100-year floodplain elevations, land-owners used approximate elevations of previous floods to attempt to increase levee heights. Establishment of flood control and flood insurance thus provided benchmarks for levee upgrades and maintenance, and likely reduced levee overtopping.

**Investment in Levee Upgrades**

Large-scale state investment in levee upgrades began after the June 1972 Brannan–Andrus levee failure and the authorization of Subventions$^1$ funding (Figure 3). The Brannan–Andrus failure resulted in curtailment of Delta exports because of increased salinity, and additional reservoir releases were required to re-establish Delta exports (Cook and Coleman 1973). Along with recognition that structural disintegration$^2$ caused increasing numbers of failures, the California Department of Water Resources (CDWR) (1982) reported the generally substandard levee conditions.$^3$ Subsequently, additional funds were authorized for Special Projects which widened the sphere of levee improvements, and thus began the current epoch of upgrading Delta levees to meet applicable standards.$^4,5$

Assembly Bill 360 (1996) expanded Special Projects to require aquatic habitat enhancement, with the stated goal of preserving the 1996 Delta. Priority went to eight western Delta islands$^6$ and protection of Thornton and Walnut groves. After 2006, CDWR expanded the interpretation of Special Projects, which included a reduction in local cost share for levee improvements throughout the Delta. Assembly Bill 1200 mandated the completion of DRMS study to assess major risks to Delta resources and to develop measures to reduce risk (URS Corporation
Figure 4  Levees and islands, project and non-project islands and legal Delta, soil organic matter content from Deverel et al. (2016)
Figure 5  Overlay of organic soils from Deverel and Leighton (2010) on former historical habitats from Whipple et al. (2012)
Figure 6  Land surface elevations where organic soils prevail (based on LiDAR data (CDWR 2007)), and approximate delineation of artesian areas from Deverel et al. (2015)
and Jack R. Benjamin & Associates, Inc. 2008b). The Delta Reform Act requires the Delta Stewardship Council to develop risk-based priorities for state spending on Delta levees (Arcadis 2016): the Delta Levee Investment Strategy (DLIS). A continuation of the original Levee Subventions and Delta Flood Protection Program, the Delta Leves System Integrity Program received funding from both the Disaster Preparedness and Flood Prevention Bond Act of 2006 (Proposition 1E) and the Safe Drinking Water, Water Quality and Supply, Flood Control, River and Coastal Protection Bond Act of 2006 (Proposition 84). Over $700 million of state funds have been invested in Delta levee maintenance and improvement since the mid-1970s. After the 1970s, the state’s cost share for levee improvements generally increased, and ranged from 50% to 100%.

Estimates for necessary future standards-based Delta levee improvements range from $3.8 billion to $4.28 billion (DSC 2015). Suddeth et al. (2010) estimated from half a million dollars per kilometer to over $2.48 million per kilometer to meet PL 84–99 standards. As part of the CALFED Levee Integrity Program Plan, Murray, Burns and Kleinlen, and Kjelson, Sinnock, and Neudeck estimated from $0.2 to $0.6 million per km to meet PL 84–99 standards (2000).

Investments since the mid-1970s augmented levee structure and resulted in improved conformance to levee standards, but data are inconsistent about corresponding reductions in numbers of failures. The DRMS study estimated that mean annual frequency of island failures was 1.39 for the period between 1980 and 2006, compared to 0.80 during 1950 to 1979 (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008). They attributed the increased rate of island flooding to cumulative effects of subsidence, sea level rise, and higher peak storm inflows. Delta peak inflows were higher during 1980–2006. Extending the DRMS study analysis to 2015, we estimated an annual frequency of 1.2. Hopf (2011) re-assessed Delta levee failures and concluded that levee performance was better than indicated by DRMS. By separating out some levee failures, Hopf (2011) reported an annual failure rate of 0.75 since 1972. Storm events associated with the high Delta in-flows since 1980 correspond to simultaneous flooding of multiple islands and tracts: 6 in 1980, 11 in 1983, 9 in 1986, and 11 in 1997 (Figure 3).

Recent levee upgrades increased compliance with relevant levee standards. About one-third of Sacramento–San Joaquin Delta levees (“project levees”) complied with federally-authorized flood-control projects (Figure 4). Local reclamation districts own and maintain on behalf of private land-owners or the State of California the remainder of “non-project levees.” The CDWR used Light Detection and Ranging (LiDAR) survey and reclamation district-engineer-supplied survey data to estimate that 84% of the total levee length complied with the Water Code’s Hazard Mitigation Plan (HMP) standards, and 61% complied with the higher PL84–99 standards. At least 80% of levee lengths on 72% of individual islands complied with the HMP, and 39% of levee lengths on 80% of individual island complied with PL84–99. Most of the non-compliant islands were concentrated in the central Delta. However, there was some uncertainty in the mapping because of gaps in the LiDAR data.

Although HMP compliance numbers generally appear encouraging, individual island flood protection requires 100% compliance. Moreover, standards specify geometric requirements and are not performance-based, nor do they account for the probability of floods larger than those prescribed by their water-surface metric. Ongoing consolidation of underlying peat and levee creep can contribute to future non-compliance and the need for ongoing upgrades to meet guidelines, which include provisions for overtopping, but not necessarily for other levee failure causes such as earthquakes and under-seepage. Additional data analysis is needed to confirm that levee upgrades since 2007 have increased compliance with standards.

Levee upgrades may have resulted in reduced estimated failure probabilities. The DRMS study estimated the range of mean annual probability of failure from combined flooding, dry-weather, and seismic threats for 2005 conditions from greater than 7% (for Sherman, Tyler, Venice, and New Hope) to less than 1% for islands on the periphery of the Delta (URS Corporation and Jack R. Benjamin and Associates, Inc. 2008a). A 5% annual probability of failure translates to a 72% probability of failure.
during a 25-year period. For most of the central and western Delta, estimated mean annual probability of failure generally ranged from 3% to 7% (53% to 84% probability of failure in 25 years).

For 2012 conditions, Arcadis (2016) estimated similar annual probabilities of failures for hydraulic- and earthquake-related failures ranging from 1% to 5% for most of the Delta. Probabilities in the central and western Delta generally ranged from 2% to 5%. Earlier studies (CDWR 1982; USACE 1982) estimated annual failure frequencies from overtopping and stability failure for 1974 levee conditions to be generally higher than the DRMS estimates for much of the central Delta, where land-surface elevations are below -2 m (Figure 6). The DRMS estimated annual failure frequencies were lower than the CDWR (1982) estimates for 13 of the 27 islands. These reductions in failure probability may have resulted from investments in levee upgrades and different analytical methods, but the extent to which DRMS study and the DLIS considered upgrades when estimating probabilities is unclear.

**HOW LEVEES FAIL**

Levees fail in multiple ways from external and internal processes. Moss and Eller (2007) listed six failure modes: destabilizing inertial loading, sliding along a preferred failure plane, slumping or spreading, seepage, erosion, and overtopping during extreme high-water events. URS Corporation and Jack R. Benjamin & Associates, Inc. (2008a) summarized stressing events: floods, earthquakes, animal burrowing, and wind waves, which correspond to failure modes of under- and through-seepage, overtopping, seismic deformation, and erosion (Figure 7). A normal-condition “sunny day” failure stressing event was also listed and associated with failure modes of through- or under-seepage, slope instability, erosion, animal burrowing, and falling trees. Levee construction inconsistencies exacerbate failure mechanisms where, for example, crest heights of two adjoining levee sections differ. These create potential locations for seepage, overtopping, and erosion (Seed et al. 2006). Duncan and Houston (1983) and Foott et al. (1992) suggested that differential settlement and deformation of peat soils resulted in cracks in levees and greater susceptibility to failure during high-water events. Seepage is a common failure mechanism in fluvial depositional environments such as the Delta (Moss and Eller 2007). Coarse-grained materials allow for flow underneath levees, which can erode foundation materials. The thickness of marsh deposits affects the magnitude of under-seepage (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008a). Flooding may also increase seepage onto adjacent islands, and the need for increased drainage and pumping requirements (Arcadis 2016).

Biogenic agents (e.g., rodent holes, tree roots, or other biological activity that create conduits) can also lead to destabilizing seepage and erosion of levee fill materials (Figure 2). The magnitude of vertical exit gradients$^{12}$ indicates the potential for under-seepage to erode levee foundational materials. Exit gradients greater than the critical gradient (the point where seepage forces approximately equal soil resistance) can result in piping or “boils,” as water moves with sufficient force to transport soil particles and levee foundation materials. Based on observations and theoretical considerations, the critical exit gradient is about 1.0, and is proportional to peat thickness (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008a).

The CDWR's Northern Non-Urban Levee Evaluations (NULE) generally classified the under-seepage susceptibility adjacent to Delta project levees as high to very high for levees in the western Delta (URS Corporation and Lettis Fugro–William & Associates 2011). The Southern NULE analysis included project levees in the southern Delta adjacent to Roberts Island and Union Island where levees were also classified mostly as having high or very high under-seepage susceptibility (Kleinfelder et al. 2011). The DRMS study also mapped most of the central Delta as highly to moderately vulnerable (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008a).

**THREATS TO LEVEE STABILITY**

Delta levees do not conform to the traditional definition. Unlike Central Valley levees, which are designed to prevent flooding only during high water, Delta levees withhold water continuously. Functionally, Delta levees are similar to dikes: embankments built to prevent flooding from the sea similar to those protecting Dutch polders.
(which are dams without spillways). The ability of levees to effectively withhold water from subsided islands depends on the strength of the levee and its foundation relative to external threats (e.g., subsidence, seepage, animal burrows, hydraulic pressure gradients, seismic shaking, high water). When external forces cause internal stresses that exceed the cohesive and frictional strength of the levee or its foundational materials, levee failure can occur. We appraised three threats to levee vulnerability: seismic shaking, high water, and subsidence.

Seismic Shaking

**Historic Observations**

What can past observations of earthquake effects in the Delta tell us about present levels of seismic risk?

As we explore this subject it is important to recognize that large earthquakes are rare events with recurrence intervals often spanning hundreds to thousands of years; therefore, observations made over relatively short time intervals (tens to hundreds of years) inadequately predict future performance. For example, the 2010 Haiti earthquake that killed over 100,000 people was the first large event in the region since 1860 (DesRoches et al. 2011), and the tsunami created by the 2011 Tohoku earthquake in Japan was the largest in the region since the Jogan earthquake in 869 AD (Sawai et al. 2012). Past observations and personal experience of earthquake effects are, therefore, clearly inadequate to assess the risk to the 150-year-old Delta levee system. Nevertheless, such observations are present in the literature, and offer some clues about Delta seismic vulnerabilities.

An oft-repeated perception among some Delta interests is echoed by CALFED: “historical information indicates that there has been little damage to Delta levees caused by earthquakes.” The literature indicates otherwise. Although Delta levees were relatively new in 1906 and low in height, Finch (1985) and Youd and Hoose (1978) reported that the 1906 earthquake caused the foundation of a railroad bridge on the Middle River to drop about a meter and become misaligned, and the pier supporting the bridge over the main San Joaquin River sank “several inches.” The embankments associated with those structures were similar in composition to the present Delta levees (Finch 1985). Hopf (2011) investigated evidence that might link the earthquake (which occurred in April 1906) to subsequent levee failures. He discovered that six islands flooded in
June and July of 1906 (Hopf 2011). Although these floods could be earthquake-related, high-water events that spring and summer were also documented. For example, The San Francisco Call on July 6, 1906 described “alarming high conditions on the San Joaquin River due to warm weather.” Prokopovitch (1985) stated that the 1906 earthquake may have weakened levees, resulting in the 1907 flooding of 53 major Delta Islands. Indeed, the largest numbers of failures on record occurred during the winter of 1906–1907. The record also contains evidence of damage to levees in their essentially contemporary state. Finch (1985) reported 15 instances of observed Delta levee damage from relatively modest earthquakes from 1979 to 1984 with epicenters in Livermore, Coalinga, Pittsburg, and Morgan Hill. Reported damages included lateral and vertical levee displacements, rotational slip (see Figure 7), and cracks.

We lack recordings of strong ground motion in the Delta region, because local seismicity has been low since seismographic instruments were installed (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008). This lack of strong shaking should be interpreted as a fortuitous outcome of a random process rather than evidence of a lack of seismic risk. The DRMS study assessed the stability of levees under a range of earthquake scenarios (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008b) using simplified numerical models. Several factors allow these hazards to now be more reliably characterized, including improved regional seismic source and ground motion models, as well as improved insights from physical model studies and investigations of levee failures in similar geologic conditions elsewhere in the world. The three essential components for assessing levee seismic vulnerability are seismic source characterization, ground motion modeling, and fragility assessment for levee sections and systems. The first two are often combined in a probabilistic seismic hazard analysis (PSHA) (McGuire 2004); PSHA maps show ground motion levels with specified probabilities of exceedance within a particular time-period. A common reference value is 10% probability of exceedance in 50 years, which is the ground motion level expected to be exceeded, on average, every 475 years. Larger probabilistic ground motion levels correspond to a higher seismic risk. We attempted to address the following questions:

1. What does recent data indicate about seismic ground motion in the Delta region?
2. What are the threats to levees from earthquakes?

**Seismic Ground Motions**

**Seismic Sources**

Several faults are present in and near the Delta: Pittsburgh–Kirby Hills, Orestimba, Los Medanos, Clayton–Greenville, and others (Figure 8). The Midland Fault underlies portions of the western Delta (Harwood and Helley 1987; Johnson 1990; Unruh et al. 2007; Unruh and Hitchcock 2009; Weber–Band 1998; Unruh et al. 2016) (Figure 8). Additionally, more distant larger faults, such as the Hayward and San Andreas, also contribute to the Delta’s seismic hazard. These faults are included in the Uniform California Earthquake Rupture Forecast Model Version 3 (or UCERF3; Field et al. 2013, 2014), which considers the potential for future earthquakes on unknown and known sources. The characteristics of unknown sources are linked to the locations and rate of small earthquakes in the region (M <6); this aspect of UCERF3 is referred to as the background (or gridded) seismicity model.

Based on geological evidence, UCERF3 documentation indicates deformation rates across the Coast Range faults ranging from 0.1 to 4 mm yr\(^{-1}\) based on geological evidence and earthquakes rates with moment magnitude M >6 ranging from \(7 \times 10^{-5}\) to \(5 \times 10^{-3}\) events per year (approximate return periods of 200 to 14,000 years). UCERF3 represents the slip rates for the Midland Fault that underlies the west Delta as 0.4 mm yr\(^{-1}\) and a M >6 earthquake rate of \(7 \times 10^{-4}\) events per year (return period of 1,400 years). Unruh et al. (2016) suggests lower levels of activity, with an estimated slip rate range of 0.03–0.13 mm yr\(^{-1}\); this would reduce the earthquake rates cited above on this fault by a factor of three to ten.

Earthquake rates for local faults can be difficult to conceptualize in the abstract, but in 2014 nature provided a tangible reminder of the capacity of similar fault systems to produce damage with the M 6.0 South Napa Event. That event occurred on
the West Napa Fault (Figure 8) (Brocher et al. 2015), which is one of a series of Coast Range faults that have slip and event rates within the aforementioned ranges. The event produced ground motions in the Delta region, but was too distant to produce damage. Ground motions during the Napa earthquake were observed to attenuate more quickly with distance than predicted by the ground motion models used in the DRMS study (Erdem et al. 2016). Currently, it is unclear whether the rapid attenuation was a specific characteristic of the Napa event, or whether it is a more general characteristic of ground motions in the region. Future research is needed to clarify if this could potentially result in regional modifications to ground motion models for the Delta.

Ground Motion

Figure 8 depicts the spatial distribution of peak ground velocity in the Delta region based on analyses performed by the USGS seismic hazards mapping team in collaboration with the authors (Zimmaro et al. 2016). We used the UCERF3 source model (time-independent, branch-averaged model, known as Mean UCERF3). We applied ground motion models developed in the Next Generation Attenuation (NGA)-West2 project (Bozorgnia et al. 2014). The results apply for a 10% probability of exceedance in 50 years and for site conditions that correspond to relatively firm soil conditions that are present at depth below surficial soils in the Delta (peats, fluvial deposits). The mapped ground motions range from 70 cm s\(^{-1}\) in the west Delta to 30 cm s\(^{-1}\) near the eastern margin of the Delta in West Sacramento. The most proximate Coast Range faults, background seismicity, and the Midland Fault are the principle contributors to the hazard (Figure 8). Distant (but more active) sources near the plate boundary (San Andreas and Hayward faults) make minor contributions. Ground motions provided by separate hazard runs using updated parameters for the Midland Fault (with reduction of the Midland Fault earthquake rate by a factor of 4.7) are reduced on average by ~2% within 20 km of the surface projection of the fault, with a maximum discrepancy of about 6% in the immediate vicinity of the central segment of the fault relative to those shown in Figure 8.

Local variations in thickness and stiffness of relatively soft, shallow soil layers may alter mapped motions. For example, Tokimatsu and Sekiguchi (2007) described ground motions during an earthquake in Nigata, Japan on organic soils that were appreciably stronger (exceeding 1g) than those on nearby inorganic soils, indicating that organic soils can amplify strong ground motions. These results are consistent with the results of simulations of amplification (even at high amplitude) for conditions consistent with those in the Delta (Kishida et al. 2009a). Even in the absence of strong site amplification, motions presented in Figure 8 are sufficient to induce ground failure in levee fills and underlying foundation soils.
Seismic Threats to Levees

A substantial seismic threat to levees during and immediately after an earthquake is liquefaction: the sudden loss of strength in granular (cohesionless) soils resulting from an increase in the pressure of the pore fluid and corresponding loss of inter-particle contact forces. Liquefaction occurs in loose, saturated cohesionless soils such as sands, non-plastic silts, and, in some cases, gravel. Delta levees can be comprised of liquefiable soils and founded on liquefiable soils. Liquefaction can cause levees to slump, spread, and crack as a result of shear strains, which result in shape changes (Figure 9), and volumetric strains that lead to settlement as pore water is expelled from liquefied sand. Liquefied sand may also flow out through cracks, resulting in additional volume loss and settlement. In some cases, the strength of the liquefied soil is too low to support the levee, resulting in a levee collapse.

The size and spatial continuity of the liquefiable stratum is an important consideration. Small liquefiable lenses surrounded by non-liquefiable soils may result in negligible deformations, whereas large, spatially-continuous liquefiable zones will likely cause significant deformations. Earthquake-induced liquefaction has caused levee deformations during past earthquakes in various locations around the world (though not in the Delta) (Miller and Roycroft 2004; Sasaki 2009; Green et al. 2011; Sasaki et al. 2012; Kwak et al. 2016a).

Liquefaction analysis begins with evaluating a soil’s susceptibility to liquefaction. The liquefaction resistance of susceptible soils is then computed using semi-empirical correlations that consider the density or stiffness of the soil (e.g., Idriss and Boulanger 2008; Kayen et al. 2013). The seismic load imposed on the soil (i.e., the demand) is computed based on earthquake ground motion, and the factor of safety against liquefaction is computed as the resistance divided by demand. Liquefaction-triggering procedures are formulated for level-ground conditions, and corrected to account for the influence of static shear stress on liquefaction resistance (Boulanger 2003) and demand (Athanasopoulos–Zekkos and Seed 2013). Once liquefaction triggering is predicted, settlements and lateral deformations are estimated using semi-empirical engineering evaluation procedures that have been calibrated with case history data (Youd et al. 2002; Faris et al. 2006), or using dynamic numerical simulations.

Levees not subject to liquefaction may be susceptible to earthquake-induced deformations (Figure 10). Earthquake shaking may weaken clays and plastic silts. However, the strength loss is not as significant as for liquefiable soils (Bray and Sancio 2006; Boulanger and Idriss 2007). This weakening combined with inertial loading from seismic shaking may result in levee slope deformations.

Because of uncertainties, levees seismic response is commonly treated probabilistically. Fragility functions define the probability of a levee exceeding a damage level; this probability depends on ground shaking, geologic conditions, groundwater levels, and other predictive variables. Fragility functions have historically been derived based on a combination of engineering evaluation procedures and expert opinion. More recently, Kwak et al. (2016) developed empirical fragility functions based on observations of damage to Japanese levees shaken by earthquakes in 2004 and 2007. A key finding was that high groundwater and soft geologic conditions increased levee fragility. The Japanese levees are infrequently loaded flood-control levees, and the groundwater table was either beneath the levee base or near the levee toe during the earthquake. In contrast, Delta levees are continuously loaded (i.e., groundwater levels are above the base of the levee) and are therefore more fragile than the Japanese levees because saturated levee fill can liquefy. The fragility functions of Kwak et al. (2016a) indicate high probabilities of levee damage using the ground motion levels shown in Figure 8, despite the relatively favorable (compared to the Delta) hydrological and geotechnical conditions. This suggests that damage to the Delta levees is highly likely at the 475-year ground-motion return period.

Although fragility functions quantify the probability that any particular levee segment or reach will be damaged by shaking, the probability that a levee system will survive equals the probability that all segments survive the earthquake. Levees comprise a series system in which failure of any single component constitutes system failure, especially when they are constantly loaded, as
several mechanisms. Cracks that form as a result of earthquake shaking may alter the flow of water through the levee by providing seepage channels. Also, earthquake-induced cyclic loading may increase settlement rates of levees founded on peat after an earthquake (Shafiee et al. 2015). This mechanism could result in a loss of freeboard, or formation of cracks in the days and months after an earthquake. Existing methods of analysis, and fragility relationships, do not consider these longer-term threats to levees.

Hopf (2011) and Prokopovitch (1985) noted an increased rate of levee failures after the 1906 earthquake. We can only speculate on whether earthquake-induced embankment cracking, perhaps combined with accelerated settlements, may have contributed to these failures. The levees are now significantly taller and under more hydraulic load than they were in 1906, and are thus more susceptible to earthquake damage. Furthermore, the

in the Delta. The probability of system failure depends significantly on correlation of seismic demands and seismic resistances among levee segments. Various approaches have been adopted to compute the probability of system failure, given spatially correlated demands and capacities (Wolff 2008; URS Corporation and Jack R. Benjamin & Associates, Inc. 2009; Kwak et al. 2016b). Kwak et al. quantified these spatial correlations for the Japanese levee system and explicitly accounted for spatial distribution and correlation. Other researchers selected a “characteristic length” where the levee system is divided into reaches that are considered uncorrelated (e.g., Wolff 2008; Jongejan and Maaskant 2015; VNK 2015).

Aside from the immediate threat to levees that liquefaction and related phenomena present, other longer term threats may also affect levees. Levees that are damaged but do not immediately release water may eventually breach because of

Figure 9  Levee deformation mechanisms due to liquefaction. Liquefaction occurs in loose saturated cohesionless soils such as sands, non-plastic silts, and in some cases, gravels. Delta levees often contain liquefiable soils and/or are founded upon foundation material containing liquefiable soils.

Figure 10  Levee deformation mechanisms in non-liquefiable soils
1906 earthquake was on the San Andreas Fault, which is approximately 80 km from the Delta. Seismic hazard is controlled by smaller and much more proximate faults (Figure 8), which threaten the Delta levees in a manner that has not yet been experienced.

When the factors that affect seismic risk in the Delta are considered, it is important to recognize key sources of uncertainty:

1. Source models are uncertain about earthquake rates and the M range of events that can be produced. The use of background seismicity models also indicates a lack of completeness relative to observed events.

2. Ground motion models estimate the mean and variability of ground motion parameters using global data for active crustal regions, but the local applicability of such models for the Delta is uncertain (Erdem et al. 2016).

3. Ground failure potential, and associated seismic fragilities, lack spatial certainty primarily because of the lack of data on the soil materials comprising levee fills and their foundations, which in turn controls liquefaction susceptibility.

The current DLIS uses seismic fragility models developed during the DRMS study. We recommend re-visiting this approach given more recent knowledge in source modeling, ground motion modeling, potential levee failure mechanisms, and system risk assessment. Such an analysis should consider the aforementioned uncertainties inherent to each component.

High Water

What are future threats to levees from high water? High water is the primary external force that resulted in 200 to 300 levee breaches in the Delta and associated rivers since the mid-1800s (Florsheim and Dettinger 2005, 2007). Arcadis (2016) consistently identified hydrologic failures as the primary Delta failure mechanism. Since 1951, 81% of Central Valley levee breaks occurred during floods generated by wintertime atmospheric river (AR) storms (Florsheim and Dettinger 2015), which are projected to increase in number and frequency in the warming future (Dettinger et al. 2016). These winter storms with heavy rains that reach higher up into the mountain watersheds than most have historically driven central California’s largest floods. When these storms and floods coincided with extreme winter tides, storm surges and high wind waves have caused levee failure and flooding in the Delta. These processes, part of El Niño Southern Oscillations, represent significant Delta hazards (Cayan et al. 2006; Bromirski and Flick 2008). As an example, the day Mildred Island flooded (January 27, 1983), West Coast sea levels were elevated by an El Niño, and high tides reached record levels in San Diego and Seattle (CSLRCOW 2012).

Field experience indicates consistently that these low-pressure storms are associated with high water levels in the Delta such as those that occurred in 1982–1983, 1998, and 2006. Increased seepage onto islands has been observed during these events.

The reported linkage of levee failures to high-river flows (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008a), suggests that projected increases in high-flow frequency associated with climate change will increase levee failure rates. In this case, overtopping (i.e., water flowing over the crown of the levee), is the primary failure mode (Figure 7). Overtopping failure results from the erosion and scour of levee materials during high-water-stage events during which there is insufficient time to complete flood barriers or during which the barriers settle (USACE 2000).

Climate change undermines the assumption of stationary processes upon which most water resource and levee management paradigms and tools are grounded (Dettinger et al. 2016) by forcing changes in water-resource behavior that exceed the underpinning probability functions. Cloern et al. (2011) concluded that there is a strong link between California’s warming climate and coastal flooding and storms, and the rate of global warming will increase with higher greenhouse gas emissions (Dettinger et al. 2016). Global circulation models described in Cayan et al. (2008) and hydrologic models trained with data to predict Sea level rise (Cloern et al. 2011) indicate an increased incidence of extreme flooding events and the risk of coastal flooding: increased frequency of extreme water heights above the 99.99th percentile of water elevation over the historical rate. This translates to increases of 1,200 to 2,000 hours per decade by mid-
century, and 15,000 to 30,000 hours per decade by the end of the century (Cloern et al. 2011). Water levels in the Delta are influenced by the tide level at Golden Gate and variable Delta inflow. An increase in the average sea level at the Golden Gate will affect water levels in the Delta, and thus likely increase hydraulic stress on the levees and contribute to an increase in the annual likelihood of levee failure. Quantification of this increased threat requires further research and data collection.

Historic hydrologic and levee failure data and climate-change modeling provide evidence for increased potential future levee failure associated with high water. For example, Arcadis (2016) indicated that the expected number of flooded islands for a 100-year flood event is approximately 16 of 109 leveed Delta islands and tracts (14.6% of the islands), with one standard deviation ranging from approximately 8 to 24 flooded islands. URS Corporation and Jack R. Benjamin & Associates, Inc. (2008b) estimated a unit probability for exceeding 10 island failures at 100 years. Suddeth et al. (2010) stated that, based on current flood and seismic failure probabilities, the median Delta island has a 95% probability of failure by 2050, and a 99% probability of failure by 2100. These estimates were based on likelihoods of failure without major investments in levees.

How do earthquake and high-water threats compare? High-water threats and a large earthquake will likely result in multiple levee failures. High-water events, which we conclude will increase in frequency with time, are more probable in any year than major earthquake threats, although the damage will generally be less. Earthquakes will result in larger numbers of island failures, more extensive damaged levee reaches, and more widespread repair. The likelihood of a large earthquake is rare in any given year, but the potential economic consequences are much larger than high-water events, which are expected to occur more regularly.

### Subsidence

How does subsidence affect levee probability of failure? Bachand et al. (submitted) investigated the effect of subsidence on current and future failure risk, and the potential benefit of strategic placement of shallow aquatic systems such as rice fields and wetlands. They did this by modeling changes in seepage and static slope instability relative to horizontal sliding failure typical of Delta levees (Duncan and Houston 1983) (Figure 7). They used 13 simplified scenarios in which levees met or exceeded the HMP requirements with varying levee heights and marsh deposit thicknesses relevant for the entire Delta. The simplified geology consisted of marsh deposits (peat, soft clay, and silt) underlain by sand. They used median soil property values for the analyses, and explored via a sensitivity analysis the effects of varying values within the ranges of the most critical variables.

Mechanistic models (Geo-slope 2012a, 2012b) (Figure 11) were used to calculate factors of safety against static and seepage failures, and to assess the sensitivity of the calculations to variability in soil-property values. Bachand et al. (submitted) used modeling results to calculate relative probabilities of failure (RPF) as a function of levee height (H) and marsh deposit thickness (T). Equations describing RPF for seepage and static slope stability were fit to the modeled results for H and T in meters:

$$ R_{\text{Seep}} = \frac{1.114}{\left(1 + e^{0.8919(1-H)/1.945}\right)} $$

$$ R^2 = 0.998 $$

$$ R_{\text{Slope}} = -1.3543 + 0.009152(T) + 0.04816H $$

$$ R^2 = 0.992 $$

$R_{\text{Seep}}$ is greater than $R_{\text{Slope}}$ when marsh deposits are thinner than about 6 m. The two failure modes were combined into a total RPF using the following equation (Hubel et al. 1990; URS Corporation and Jack Benjamin & Associates, Inc. 2008a):

$$ R_{\text{Total}} = 1 - \left(1 - R_{\text{Seep}}\right) \times \left(1 - R_{\text{Slope}}\right) $$

Calculated $R_{\text{Total}}$ varied as a function of levee height and marsh deposit thickness (Figure 12); predicted failures became more likely as marsh deposits thinned to less than 3 or 4 meters, and risk was exacerbated where levees were higher. The increasing probability of failure with thinning peat...
results from increasing seepage forces, and increasing risk with levee height is from increasing static loads. The RPF results were applied to the subset of levees likely to be affected by subsidence over time: levees underlain by marsh deposits and associated with islands greater than 60 hectares.

Levee height (H) and marsh deposit thickness (T) were calculated based on LiDAR elevation maps and a marsh deposit base elevation grid (Deverel et al. 2015). Deverel and Leighton (2010) also constructed a Delta-wide grid of an estimated 2050 ground surface elevations and estimated subsidence rates, which were used to estimate changes in ground surface elevations, and values of H and T, over time. H was also increased as a function of sea level rise, estimated to be 0.33 m by 2050 (Deverel and Leighton 2010). Levees were divided into segments of about 4 m and elevation, subsidence, and sea level rise data were used to calculate RPF for each segment over time. To calculate a failure occurrence rate, Bachand et al. (submitted) reviewed levee failures and determined that 9 of 15 failures since the mid-1970s were likely related to seepage or slope stability issues. Based on this information, they calculated an instability failure rate of 0.22 failures per year for 1994, the mid-year of the observed failure period. Bachand et al. (submitted) estimated a failure rate of 0.25 failures per year in 2016, and 0.32 failures per year by 2066 from subsidence and Sea level rise without change in land use or levee structure. Growing rice or wetlands adjacent to 600 km of levees is projected to stop subsidence (Deverel et al. 2016) and thus reduce the failure rate; Bachand et al. (submitted) estimated the failure rate to be 17% lower than if subsidence were allowed to continue.

To achieve 50% of the risk reduction, rice or wetlands would be required adjacent to 190 km of the levees where the greatest risk increase is expected, i.e., levees where marsh deposit thickness is less than 3 m and that have estimated subsidence of about 1.4 m or more during 50 years (Bachand et al., submitted). This magnitude of subsidence would occur in highly organic (>25% organic matter content) soils at least 1.4 m thick (Deverel and Leighton 2010). Bachand et al. (submitted) suggested that guidance for seepage berm width be used to estimate effective rice field width. The primary uncertainty in this analysis is the distance from the levee where subsidence affects levee stability for current and future conditions.

The seepage model results indicated that subsidence will increase seepage rates and RPF. For model runs with levees more than 4.5 m high and peat less than 3.5 m thick, 1.5 m of subsidence increased seepage flow by about 67%. This is consistent with data presented in Deverel et al. (2015), which indicates that subsidence resulted in increased seepage-affected marginally or non-farmable areas 10-fold from 1984 to 2012—a key factor was the presence of artesian conditions, which were delineated in most of the areas where there are organic soils (Figure 6).

Delta levee practical experience indicates that failure occurs typically at the interface of the marsh deposits and levee fill where there can be through-seepage and internal erosion. A potential solution is a vertically-placed cut-off slurry wall17 that extends through the center of the levee to prevent seepage. However, a cut-off slurry wall is expensive, and is likely impractical on a wide scale in the Delta.

Bachand et al. (submitted) used traditional equilibrium factor-of-safety calculations related to slope stability and seepage to estimate RPF.
Figure 12  Overlay of current island boundaries on historic channels (red). Non-historic channels are indicated in yellow and were excavated in areas where there were not historic channels as mapped by Whipple et al. (2012).
However, relying solely on traditional calculations for highly-deformable peat soils is uncertain because deformations may affect levee stability before critical factor-of-safety values are reached (Foott et al. 1992). Tension-induced differential consolidation from peat deformation outside of the radius of a typical slip circle led to levee crack formation, which Foott et al. (1992) attributed to dewatering of the peat. Subsidence that results in drainage ditch deepening beyond the radius of the typical slip circle (see Figure 7) may affect levee failure, and merits further investigation.

**Consequences and Risk**

**Consequences of Levee Failure and Maintaining the Status Quo**

Delta levee failure and island flooding effects may include loss of life, residential and commercial buildings and their contents, machinery, and crop production; interrupted highway and railway travel, emergency services, and water supply; and ecosystem effects. Economic consequences may extend well beyond the Delta, especially for curtailments in exported water. When island levees fail, island repair and recovery costs can be substantial, and, as exemplified by the Jones Tract failure in June 2004, the financial effects and repair costs can exceed the value of the land. Costs associated with 11 of the 28 islands that flooded from 1969 to 1983 were about $177 million (Prokopovitch 1985). Estimated costs to repair the 2004 Jones Tract failure totaled $90 million (Suddeth et al. 2008), but the actual levee breach repair cost was $30 to $40 million; additional costs were from property damage. URS Corporation and J. R. Benjamin & Associates, Inc. (2009) estimated the cost of repairing a levee breach to range from $20 million to $40 million. URS Corporation and Jack R. Benjamin & Associates, Inc. (2009) indicated that a seismic event during 2005–2030 is the single greatest risk to levee integrity, resulting in multiple island failures and in economic costs of over $10 billion resulting from water supply and utilities service disruption, emergency repairs, and infrastructure replacement. High water failures were also estimated to be in the billions of dollars (CDWR 2009).

Costs of repairing a levee failure depend on the size of the breach, the materials, engineering and construction costs of fixing and reinforcing breached levees, armoring and repairing the suddenly-inundated unrocked landside slopes that experience erosion damage when winds arise, pumping out the island, and the lost profits from agricultural production on the island (assuming annual crops) (Suddeth et al. 2010). These costs, which reoccur each time the island fails, should be considered in the present net value for the island and in the average cost of failure repair. The cost of not repairing an island when it fails is the sum of the cost of rebuilding or re-locating existing infrastructure (such as highways and railways). Suddeth et al. (2010) suggested that comparing the estimated cost of repair and no repair will lead to identifying the least expensive or most profitable choice. Using estimated property and asset values, levee upgrade costs, and probabilities of failure from the DRMS study, Suddeth et al. (2010) developed a list of islands to be repaired if they failed. Of the 34 subsided islands analyzed, between 11 and 25 islands economically justified repair after a levee breach.

**Water Supply Consequences**

The Delta provides a portion of the drinking water for more than 27 million Californians—nearly two-thirds of the state’s population. The primary elements of state-wide costs as a result of levee failure are agricultural losses, urban user losses from water supply disruption, and the lost use of major infrastructure (e.g., state highways that cross the Delta). However, not all islands that provide conveyance or salinity barriers, or that contain essential infrastructure are essential to ensuring water supply. The current Arcadis (2016) island water supply risk metric is based on counts of water supply functions (conveyance, salinity barrier, water user groups served, and infrastructure), relative water quantity used, and number of water user groups served.

Based on a survey of urban water suppliers, DRMS estimated urban user loss costs with a 50% reduction in supply at $1 to $1.5 billion. Over 60% of the estimated cost was for southern California suppliers. The DRMS study assumed that reductions in
agricultural water supply would result in additional groundwater use and reduced crop acreage. With implementation of the Sustainable Groundwater Management Act, additional groundwater pumping may not be a viable option in most of the San Joaquin Valley. URS Corporation and Jack R. Benjamin & Associates, Inc. (2008b) estimated the statewide costs for water supply reduction and infrastructural damage to range from tens of millions of dollars to multiple billions of dollars, for one to multiple flooded islands resulting from hydrologic events.

**Maintaining the Status Quo**

Ongoing system operation results in relatively high costs for municipal water treatment because of the presence of bromide and dissolved organic carbon (Chen et al. 2010). Dissolved organic carbon (DOC) concentrations in Delta export waters consistently exceed the 3 mg L\(^{-1}\) water-quality objective. Formation of disinfection byproducts (DBPs) such as trihalomethanes (THMs) can result from high DOC concentrations, which has required utilities to increase treatment costs.

Amy et al. (1990) estimated that agricultural drainage from Delta island organic soils contributes 20\% of the THM formation potential (THMFP) at Delta export pumps. Recent U.S. Geological Survey (USGS) analysis of historic data indicates percentages that range from 13\% to 49\% from in-Delta sources (Kraus et al. 2008). Higher percentages are associated with winter drainage. Deverel et al. (2007) demonstrated that oxidation of soil organic matter and subsequent mobilization of DOC to drainage ditches are the primary mechanisms that lead to DOC export from subsided Delta islands. Oxidation of soil organic matter in drained agriculture also results in losses of carbon dioxide that range from 15 to 20 tons carbon dioxide per ha (e.g., Knox et al. 2015), which could be stopped or greatly reduced if islands are converted to rice and permanently-flooded carbon sequestration wetlands (Knox et al. 2015) or tidal wetlands.

Documented increased wet and non-farmable and marginally-farmable acreage resulting from subsidence and consequent seepage from the 1980s to 2012 (Deverel et al. 2015) brings into question long-term sustainability of the status quo. As differences in elevation between surface water and island surfaces have increased because of subsidence, seepage rates onto islands have increased, and are predicted to increase in the future. Wet, non-farmable, and marginally-farmable acreage affects the future of farming and property values, and should be considered when levee investments are prioritized.

**Ecosystem Consequences**

Achieving the coequal goals of protecting, restoring, and enhancing the Delta ecosystem requires successfully establishing a resilient, functioning estuary and surrounding terrestrial landscape that can support viable populations of native resident and migratory species with diverse and biologically appropriate habitats, functional corridors, and ecosystem processes (DSC 2013). High-value habitat types include those that are important to recovering species (e.g., Herbold et al. 2014), those that replace historic habitat losses (e.g., Whipple et al. 2012), and those that emphasize priorities from conservation plans (DSC 2013): tidal marsh, non-tidal marsh, managed wetlands, riparian forest and scrub, seasonal floodplain, alkaline seasonal wetland, and vernal pools. Location and number of levee failures, time of year, and water conditions influence the effects on the ecosystem. When multiple islands flood, the DRMS study team estimated vegetation losses of up to 39\%. Moreover, large-scale habitat losses were predicted, with consequent displacement of birds and other species. However, if levees fail, ecosystem and water-quality benefits could also result from increases in tidal habitat.

Potential high-value, non-tidal habitat that could result from restoration or levee failure has been quantified as the maximum of known proposed restoration (tidal or leveed non-tidal) and elevation-based mapping of habitat potential (Arcadis 2016). Arcadis (2016) calculated expected flooding of high-value non-tidal habitat as the sum of the product of the annual probability of flooding with the area of existing and potential non-tidal habitat.

**Risk**

Risk analysis combines the hazards, the estimated frequency of the different magnitudes of these
hazards, and the consequences of failures in a probabilistic approach. Arcadis (2016) estimated the expected annual damage (risk) for varying flood levels on infrastructure and assets as the sum of the product of the probability of annual flooding for estimating economic losses. Through the DRMS Phase I study, the URS Corporation and Jack R. Benjamin & Associates, Inc. (2008, 2009) implemented a multi-step process to develop conditional probabilities of levee failure under seismic, high water, and sunny-day stresses. They used geotechnical modeling and professional judgement to develop probability-of-failure curves to analyze how levees responded to loading from a stressing event for different combinations of loading parameters and levee characteristics.

The DRMS study estimated mean annual number of failures for Delta islands and Suisun March that ranged from less than 0.01 to over 0.07. Most Delta island annual mean failure estimates ranged from 0.03 to 0.05. Sherman, Tyler, Venice, Sargent Bernhard, and New Hope were assigned values over 0.07. The DRMS study estimated probabilities of multiple island failures that ranged from over 70% to less than 20% for 10 to 30 simultaneous levee failures associated with high-water conditions between 2005 and 2030. Higher probabilities were estimated for a major earthquake: 40% from 2005 to 2030. The frequency of levee failure and flooding for single and multiple islands, and the generation of levee failure sequencing, were used to perform both levee emergency response and repair as well as hydrodynamic analyses, which in turn provided input into cost estimates.

The DRMS study results have been challenged because of their alleged failure to fully and accurately account for substantial recent levee upgrade investments, the extent of subsidence, analysis of uncertainty and error, and misrepresentation of failure risks from Sea level rise, flooding, and seismic hazards (e.g., Brandenberg and Stewart 2008; California Central Valley Flood Control Association 2011; Hultgren–Tillis Engineers 2015;). The way in which risk for individual levee segments has been used to evaluate levee system risk was not sufficiently robust to properly consider spatial variability and correlation of demand and capacity (Kwak et al. 2016). Moreover, independent scientific review (CALFED Science Program Independent Review Panel 2008) concluded that although DRMS study is acceptable as a tool for informing policy-makers and others about potential resource allocations and strategies to address risk, important caveats relate to prediction uncertainty and ecosystem consequences. Specifically, the CALFED panel considered future economic consequences to be over-estimated, and recommended additional sensitivity analyses.

The DLIS seeks to improve upon the DRMS study results, specifically relating to inclusion of additional levee structural information to develop non-seismic fragility curves. To reduce risks and further advance the coequal goals of water supply reliability and Delta ecosystem restoration, the DLIS objectives include development of a methodology that decision-makers can use to evaluate and recommend priorities for investments in Delta levees. Seismic levee risk is treated separately from hydrologic and hydraulic risk. Current planning is for seismic risk to be based on USGS ground motions (similar to Figure 8) and fragility curves developed by the DRMS study, despite the known problems with those fragility curves, as discussed previously. For this reason, we recommend that fragility models be redeveloped in consideration of changes in levee configuration, updated site amplification models, and improved knowledge of the mechanisms (including earthquake damage) that threaten levee stability, as described in the previous section.

Risk reduction analysis in the DLIS support tool relies on representing levees using non-seismic fragility curves. Baseline risks have been estimated using fragility curves that reflect the current data set. To update fragility curves, geotechnical data can be added as it becomes available. To develop fragility curves, reclamation district engineers (Gilbert Cosio, Chris Nuedeck, and Gil LeBrie) provide ongoing review of the process to provide the most recently available input. Fragility curves require updating in response to ongoing subsidence and settlement of the levee system, because of improvements funded with state and reclamation district investments, and would require updating because of the potential change in levee geometry resulting from a seismic event.
The DLIS risk methodology (Arcadis 2016) will estimate expected annual damages (EAD), which is an average annual monetary value of current and future losses from flooded Delta infrastructure and other assets based on the probability of flooding. Among the future hazard changes, the DLIS is considering only potential sea level changes in the risk analyses. Arcadis (2016) used revised hydrologic and hydraulic levee fragility curves generated since the completion of the DRMS study (e.g., CDWR 2012). The other future hazard changes (i.e., subsidence, earthquakes) are considered to be too unpredictable to provide a meaningful estimate of levee response to changed conditions (Arcadis 2016).

How would a single-island levee affect levee vulnerability and associated costs on other islands? For example, flooding of an island may impede access to other islands. Seepage may also increase on islands adjacent to flooded islands. Concern has been expressed about an increase in wind fetch that would result in greater water-side erosion on nearby islands. URS Corporation and Jack R. Benjamin & Associates, Inc. (2008a) concluded that this domino effect was not supported by the long-term survival of remnant levees of flooded and abandoned islands. Twenty-three years after Mildred Island flooded, the entire levee—except the breach area—is still visible. Similar observations were made for Little Franks Tract as well as Little Mandeville and Rhode islands, which flooded 23, 12, and 35 years ago, respectively. These observations indicated that the remaining levees did not erode extensively after they were breached.

What lessons from the DRMS study apply to the ongoing DLIS? The DRMS study used a coordinated multi-discipline scientific effort that included a wealth of data and analyses prepared by topical experts (who responded to rigorous and substantial technical review) in various subject areas; geomorphology; subsidence, seismology, climate change, flood hazard, wind-wave hazard, emergency response, levee vulnerability, hydrodynamics, water quality, water management and operations. The DRMS study channeled information into comprehensive estimates of failure probability, consequences, and risk, which resulted in highly useful products. The science and approach remain valid and applicable.

Within the DLIS, we recommend integration of the best available science into analysis of future risk for all potential hazards. Interdisciplinary science and levee practice in which there is skillful discussion and dialogue among scientists, engineers, and practitioners would be optimal. The flexible risk-analysis tool may serve as a technical consensus-building tool in which different options and approaches could be tested, and uncertainty assessed and discussed by experts from various related scientific disciplines. In this way, future inclusion of the risk from future subsidence and earthquakes could be assessed.

This approach could also lend itself to the following factors for prioritization of levee investments being considered: sustainability as discussed previously relative to the Deverel et al. (2015) results; accommodation space below sea level (which increases with ongoing subsidence) that can fill with flood water when levees fail (see Deverel and Leighton 2010); and effects of levee failure on habitat and water quality. Hydrodynamic modeling suggests reductions in central Delta salinity from flooding of Liberty Island and other Delta islands that results from modifications to the distribution of the tidal prism (URS Corporation and Jack R. Benjamin & Associates, Inc. 2011; 2016 phone conversation with J. DeGeorge, unreferenced, see “Notes”). Hydrodynamic modeling of island flooding in Suisun Marsh also indicated benefits to Delta salinity (U.S. Bureau of Reclamation et al. 2011).

**Delta Levees and the Ecosystem**

The Delta once supported abundant diversity and large wildlife populations. Since reclamation, agriculture and urban areas replaced the large expanse of approximately 365,000 acres of tidal freshwater emergent wetlands and over 1,600 km of associated tidal channels (Whipple et al. 2012; Brown et al. 2016, Figure 12). Interrelated changes included loss of ecological function; reduction in habitat extent and heterogeneity; loss of connectivity within and among habitat types; degradation of habitat quality; disconnection of habitats from sustaining physical processes; and invasive species (SFEI–ASC 2014). SFEI–ASC (2014) identified historical ecological functions: processes and characteristics
that supported the life histories of resident and migratory fish, marsh wildlife, water birds, riparian wildlife, and marsh–terrestrial transition zone wildlife. Modern channels are wider, straighter, deeper, and simpler, and generally lack the fine-scale structure and micro-topography that provided abundant aquatic habitat. There is also evidence for reduced organic matter to food webs that existed in the pre-reclamation freshwater Delta (Brown et al. 2016).

The California Water Code specifies inclusion of habitat restoration in levee projects and directs the CDWR to create “net long-term habitat improvement.” Moreover, levee-improvement projects authorized under Assembly Bill 360 “shall include provision for the protection of fish and wildlife as determined to be necessary by the California Department of Fish and Game have a net aquatic habitat enhancement.” The CDFG (now called the California Department of Fish and Wildlife, CDFW) is also required to determine that the proposed expenditures are consistent with a net long-term habitat improvement program and have a net benefit for aquatic species in the Delta. There is little publicly-available evidence for compliance with these provisions of the Water Code (Davenport et al. 2016).

In addition, the Delta Reform Act of 2009 required the Delta Stewardship Council to promote expansion of floodplains and riparian habitats in levee projects, and established objectives that include interconnected habitats; establishment of migratory corridors for fish, birds, and other animals; and the restoration of habitat necessary to avoid a net loss of migratory bird habitat. The Delta Plan promotes the expansion of riparian habitat on levee projects and requires evaluation of levee habitat within the Delta.

**Monitoring and Mitigation**

**Assessment of Levee Structure**

Levee subsurface investigations provide essential information about the ability of levees to withstand external stresses. Without knowledge of internal composition and structural integrity, assessment of levee response to earthquakes or high water, and prioritizations of maintenance and upgrades, are uncertain. Delta levee artificial fills contain a spatially-variable heterogeneous mixture of geologic materials of varying strength and compressibility that include silt, sand, clay and peat. Levees are founded upon peats and mineral soils that range from clays to sands that may be poorly consolidated and loose. Quantification of levee inner layering and structural integrity are required to realistically assess how levees will respond to external forces, and for maintenance planning. Traditionally, borehole logs, blow counts, and cone penetrometer testing (CPT) have been used to assess the internal structure of levees and the potential for liquefaction. The lateral extent of a liquefiable stratum is an important issue to determine earthquake vulnerability. Liquefaction of a sand layer that extends hundreds of meters along a levee (and is similarly present along other levees) is more likely to yield a multiple-island earthquake catastrophe.

Delta levees have been investigated extensively (URS Corporation and Jack R. Benjamin & Associates, Inc. 2008), and CDWR maintains an extensive borehole–CPT database that includes data collected since the DRMS study. Although substantial geotechnical information can be often gained from borehole and CPT logs and sampling, the heterogeneous inter-layering of materials can make it difficult to stratigraphically correlate from one borehole or CPT to another. Moreover, soil sampling and laboratory characterization of soil materials often involves disturbed samples that may not represent in situ conditions, and which, from a holistic, island levees-as-a system perspective, may be too widely spaced to enable engineering properties (e.g., shear strength, volume change characteristics, and seismic properties) that critically influence levee risk to be adequately assessed. Estimation of peat thicknesses can provide useful information for levee upgrades that compensate for post-earthquake peat consolidation and potentially increased settlement rates. The presence of loose, saturated sand or low-plasticity silts (as indicated by low blow counts) can indicate liquefaction potential. Recommended geotechnical practice includes drilling boreholes at regular intervals; in situ sampling (in addition to CPT) and shear strength testing; visual soil textural and depth-to-groundwater evaluation; and laboratory testing for moisture content, density, particle size, and Atterberg limits (USACE 2000). Professional judgment should be used to configure a site investigation for existing
levees, including assessment of the spatial variability of geologic strata on which the levees rest. Most levees in the area of organic soils appear to contain, or be underlain to some degree by, organic materials (Atwater 1982; URS Corporation and Jack R. Benjamin & Associates, Inc. 2008; Deverel and Leighton 2010; Kleinfelder 2011; URS Corporation and Fugro-Lettis & Associates 2011; Deverel et al. 2015). However, because the present-day artificial levees were mostly constructed on the overflow banks of the river and distributary sloughs, most rest on natural levee deposits (Figure 12) or on peat and mud deposits that interfinger in the subsurface, creating vertical interbeds of silt and sand with organic-rich material. URS Corporation and Jack R. Benjamin & Associates, Inc. (2008a) revealed that the foundation materials for Delta levees varies greatly, even between neighboring islands, because of a history of channel migration and river meandering.19

Beyond the DRMS study efforts, more recent studies to characterize geologic materials within and underneath levees and associated vulnerability have been conducted or are ongoing. For example, the CDWR’s Non-Urban Levee Evaluations (NULE) Project evaluates over 2100 km of non-urban state and federal project levees and over 660 km of appurtenant non-urban non-project levees.20

**Geophysical Methods and Remote Sensing**

Moore and Shlemon (2008) suggested that geophysical and remote sensing techniques were for developing a more comprehensive and systematic picture of levee internal structure. These techniques may provide continuous soil data along survey paths for less cost than traditional borehole logging. Surface-based geophysical techniques have since been more widely tested. Methods included electromagnetic (EM) induction, electrical resistivity, capacitively-coupled resistivity (CCR),21 ground penetrating radar (GPR), and seismic (multi-channel analysis of surface waves) methods to rapidly interrogate subsurface conditions. In general, geophysical techniques have not, thus far, proved useful for widespread use in the Delta.

![InSAR-derived map showing rate of ground movement at a site on Sherman Island that underwent relatively rapid deformation during 2009–2015.](https://uavsar.jpl.nasa.gov/, accessed 2016 December 28).
To map subsurface variations in subsurface electrical-conductivity conditions and to identify excessively permeable zones, EM and CCR surveys were conducted along levees on the Feather River, Bear River, American River, Sutter Bypass, Yolo Bypass, Sacramento River, Stanislaus River, San Joaquin River, and tributaries, and proved useful to assess foundation conditions and potential under-seepage areas (USACE 2015). Also, the CDWR has used airborne EM to survey urban levees. However, where there is extensive borehole data in the Delta, EM technology results have not compared well with borehole data (2016 in-person conversation with G. Cosio, unreferenced, see “Notes”).

The multi-channel analysis of surface waves method (Park et al. 1999; Miller et al. 2000) uses low-frequency surface waves (e.g., 4 to 100 Hz), which propagate within several tens of meters of land surface, to estimate shear-wave velocity variations with depth. Shear-wave velocity can be correlated with soil stiffness and compaction. Ferriz (2016) recently tested this technique on Sherman Island, where shear wave variations were evident with depth. Ground-penetrating radar transmits high-frequency electromagnetic pulses into the subsurface. Leclerc and McDaniel (2006) attempted to use GPR to detect peat/mineral layering in Delta peat soils, and reported a lack of radar reflections below depths of about 1.5 m, indicating rapid signal attenuation, and thus indicating that where there is saturated peat GPR cannot be used effectively to assess levee stratigraphy.

Remote Sensing

Two remote sensing techniques provide important information about levee deformation, movement, and conditions: synthetic aperture radar (SAR) and light detection and ranging (LiDAR). Previous SAR satellite studies (Cohen et al. 1998; Brooks et al. 2012) faced obstacles related to small-scale changes in land-surface elevation and levee deformation. Longer wavelengths for SAR imagery combined with regular acquisitions, high spatial resolution, and novel data-processing techniques have been effectively used to monitor subsidence and levee deformation, and to identify areas where seepage may be occurring (Jones et al. 2016).

Interferometric SAR (InSAR) measures surface deformation directly through radar remote sensing via high-altitude aircraft or space craft (Bamler et al. 1998). Measurements simultaneously cover large areas at one time in imaged swaths that are tens to hundreds of kilometers wide. It is possible to achieve measurement accuracies of approximately 5–to 10-mm surface deformation when earthen levees are imaged in a single pair of measurements, and significantly higher accuracy, as fine as 1–2 mm yr⁻¹, when using a time-series of multiple images (e.g., Sharma et al. 2016). InSAR also differs from traditional survey methods and LiDAR because it can see through clouds, smoke, haze, and image surfaces without solar illumination.

To determine the cumulative deformation that occurred between data collections, the InSAR technique involves acquiring an initial image that serves as a baseline reference, relative to which movement is quantified by subsequent imaging and processing. Earthen levee and farm field monitoring is particularly challenging because surface disturbance (temporal de-correlation) caused by grazing and agricultural (e.g., cultivation) activities corrupts the deformation signal. Nonetheless, single pairs of images, and a time-series of images collected at intervals of roughly 2 months, have successfully been used to identify deformation on or near levees (Jones et al. 2012, 2016; Sharma et al. 2016).

InSAR is particularly useful to identify areas of localized levee deformation, and will be valuable in determining the extent of movement after major floods, breaks, or seismic events. For example, Jones et al. (2012) investigated the levee on the north side of Bradford Island where a barge caused levee damage that necessitated repair between InSAR data collections in July and September 2009. The method has been used to identify rapidly-subsiding areas in the Delta, including an area on Sherman Island where movement is occurring on a levee and extending a significant distance inland from the levee (Figure 13). Regularly collected and processed airborne InSAR can, therefore, detect levee deformation remotely, and could potentially help identify areas that require attention. InSAR may also provide reasonable estimates for soil subsidence on Delta islands. InSAR-derived subsidence rates for Sherman Island during 2009–2014 averaged 1.3 ± 0.2 cm yr⁻¹ (Sharma et
al. 2016), generally consistent with rates reported in Rojstaczer and Deverel (1995) and Deverel et al. (2016).

A second LiDAR survey (the first was conducted in 2007) may improve our knowledge of the spatial distribution of oxidative subsidence rates, and would certainly provide useful information about how levees conform to the PI 84–99 and HMP standards. However, in light of the uncertainty in the LiDAR data conducted in 2007 (15 cm at the 90th percent confidence interval), for a return LiDAR flight in the Delta during the next few years, uncertainty needs to be accounted for in the delineation of Delta-wide estimates of subsidence that are occurring at rates of about 1 to 3 cm yr$^{-1}$ or less. Periodic InSAR data collection with sub-centimeter accuracy is a tool of substantial potential value, but ground truthing is needed to verify subsidence rates and to serve as reference points for the InSAR-derived rates. This could be supplied by continuous GPS stations and extensometers (Deverel and Rojstaczer 1996). Extensometer data on Sherman Island (Deverel et al. 2016) was useful to validate the Sharma et al. (2016) estimates.

Bawden et al. (2014) reported the use of terrestrial LiDAR (T-LiDAR) to topographically map 3-D surface elevation displacements with exceptionally high resolution. These were used to calculate levee-surface changes during and after a seepage test on Twitchell Island. The high density of measurement points allowed sub-centimeter-scale changes in the levee surface (cracks and consolidation) to be accurately determined. This first-of-kind assessment in the Delta provided substantial insight about how to optimally detect, measure, and characterize dynamic deformation and surface changes. T-LiDAR may prove useful if set-up and data acquisition can prove economical. The USGS Water Science Center in Sacramento recently invested in a high-resolution bathymetric and LiDAR system that may help map levee geometry for a reasonable cost (2016 phone conversation with J. Howle, un referenced, see “Notes”).

Assessment of Seismic Effects

A key data gap for improved assessment of seismic vulnerability is the characterization of levee materials and the extent of liquefiable materials. Figure 14 shows the extent of available borehole and CPT data in the Delta. A key area for future investigation is improved geotechnical characterization of these earth materials, and consideration within the risk assessment process of the full range of ground failure mechanisms that may develop from local soil conditions, especially as related to the potential for liquefaction of levee foundation and fill materials, and the deformations of levees founded on peat. This work would provide a clearer picture of overall seismic vulnerability.

Mitigation and Restoration

Mitigation of Delta Levee Vulnerabilities

During Phase II, the DRMS steering committee and consultants, and the CDWR identified improvements in the forms of building blocks or individual improvements and scenarios, ensembles, or combinations of building blocks that could mitigate levee vulnerability, with the objective of achieving multiple risk-reduction objectives (URS Corporation and Jack R. Benjamin & Associates, Inc. 2011). For example, levee upgrades were proposed to reduce the risk of levee failures, isolated and non-isolated Delta
conveyance mechanisms were intended to preserve and protect water-export capabilities if levees fail, and ecological restoration solutions were chosen to increase and diversify the habitat in the Delta and Suisun Marsh and mitigate the potential effects of levee failures.

Some scenarios and building blocks have been implemented, or progress has been made toward implementation. These include preparedness improvements such as acquisition of easements; development of capabilities to rapidly assess levee stability, repair, design, and construction; emergency planning (CDWR 2014); and acquisition of flood fight materials and supplies and training. The Middle River earthquake recovery strategy establishes a freshwater pathway from the Sacramento River to the export pumps in the south Delta, generally along Middle River (CDWR 2014), which will apply if many islands flood after an earthquake. Repair of levees and construction of channel barriers along this corridor can help restore Delta water exports during extreme events. Land-use changes to stop and reduce island subsidence include wetland construction and rice cultivation on Twitchell and Sherman islands.

To mitigate the effects of earthquakes on levees, many possible mechanisms of deformation that could result in a breach must be considered. Possible mitigation options include ground improvement (especially when liquefaction contributes to ground deformation), construction of setback levees, increasing levee height, or placement of buttress fill (e.g., Hultgren–Tillis Engineers 2009). Uncertainties in soil properties and seismic demands should be considered when mitigation measures are designed and implemented, so the reliability of the levee system—pre- and post-mitigation—can be evaluated holistically. Levee mitigation should be based on analysis of systems that comprise many reaches. Failure at any particular point along a levee system results in inundation of the protected island; target failure probabilities from cross-section analysis would need to be low (and factors of safety would need to be high) to provide an acceptable level of system reliability.

**Levee- Habitat Projects**

The DLIS intends to guide habitat enhancement that will mitigate adverse environmental effects of levee projects, help ensure compliance with relevant legislation and co-equal goals, and benefit aquatic species. Although levee-related habitat improvements will provide a small part of all ecosystem benefits needed by native species, levee habitat projects need to occur within the larger context of Delta restoration, and ensure that state levee investments contribute to achieving co-equal goals and providing net benefit for aquatic species.

The Ecosystem Restoration Program (ERP 2014) vision advocated setting back or flattening levees to enhance river meanders and restoration of erosion and deposition processes, as well as providing fish and wildlife with access to floodplains. Restored habitats will re-establish a greater land–water connection and facilitate transport and exchange of sediment, nutrients, and organic materials that contribute to ecosystem productivity. Challenges to implementation include: U.S. Army Corps of Engineers—required removal of trees and most shrubs on and around levees under their jurisdiction, lack of information about the effectiveness of existing and potential levee–habitat projects, and local resistance to allocating land for restoration. Four key structural modifications—and related habitats, setback levees, adjacent levees, extra-wide levees, and planting benches—have been assessed (Davenport et al. 2016). Levee-related habitat types include channel-margin habitat, freshwater marsh (tidal and non-tidal), shaded riverine aquatic, and riparian habitat. Extra-wide levees (Figure 2) (sometimes referred to as habitat levees) provide a gradual waterside slope on which riparian, shaded riverine aquatic, and channel margin habitat can be established. Benefits to salmonids have been a primary focus, but benefits to native and non-native species requires evaluation. These structural modifications provide overhanging riparian vegetation, i.e., shaded riverine aquatic; slower water velocities, and soft bank substrates, which may offer important protective and feeding habitat for migrating juvenile salmon (Murphy and Meehan 1991; Smokorowski and Pratt 2007). Overhanging vegetation also protects small fish from predation by birds. In contrast, levee revetment at the channel margin is generally unsuitable habitat for
juvenile salmon because predators in the gaps of the riprap can ambush smaller fish (McLain and Castillo 2009). Greenberg et al. (2012) presented evidence that Delta shaded aquatic riverine habitat provides cooler microclimate during the summer months when native fish can be temperature-impaired in open channels. Levees that are set back a substantial distance from channels can facilitate restoration of natural riverine processes and provide broad areas of floodplain habitat that benefit aquatic and terrestrial target species (Golet et al. 2013).

Davenport et al. (2016) assessed 15 Delta levee-related habitat improvement projects. Appropriate fish and wildlife monitoring data were lacking for most projects, data were inconsistent across projects, and monitoring of wildlife response was rarely implemented. A few of the projects demonstrated through subsequent wildlife monitoring that target species occupy restored areas. Determining net benefit to targeted species will require evidence of increased occupancy and of a relationship between increased habitat availability and population growth.

Recommendations include improved consideration for and monitoring of effects on all wildlife, in particular fish and birds. Block et al. (2001) advocated using measures of population dynamics. Additionally, future habitat improvement projects should be strategically located and planned in the light of conceptual models for target species and future changes in sea level, sediment input, and infrastructure, and then managed adaptively. Larger spatial scales and complex landscapes that mimic the pre-development habitats (Figure 12; Whipple et al. 2012) are preferable and will benefit a wider array of wildlife (Brown 2003; Herbold et al. 2014). The Delta Plan calls for the Delta Science Program to develop landscape-scale conceptual models in collaboration with other agencies, academic institutions, and stakeholders.

Levee policy has focused on habitat associated with levees. However, tidal wetlands in the Delta such as those on Liberty Island, Little Holland Tract, and Lower Sherman Island Lake, will likely benefit native pelagic fishes (Brown et al. 2016). In light of the pelagic organism decline; the potential water-quality benefits of flooded islands; the economics of levee failure, repair, and upgrades; and island flooding, levee investments and development of tidal wetlands in appropriate areas of the Delta merit consideration in tandem. Current policy emphasizes levee improvement. We suggest that improved recognition, evaluation, and coordination are needed between the creation of high-value tidal ecosystems and the needs of levees. The DSC (2015) advocates for future restoration opportunities, but the mechanisms to optimize future restoration are unclear.

CONCLUSIONS

Delta levee science and practice have evolved substantially during the last decade, and hundreds of millions of dollars have been spent on levee upgrades. In the light of newly-available data and information, we attempted a reappraisal of key factors and processes that affect levee vulnerability.

We appraised key sources of levee vulnerability, external forces and processes that influence future Delta levee failures are seismic shaking, high water and subsidence. We attempted to answer questions about implications of recent data for seismic ground motion and earthquake threats.

Estimated ground motions are large enough to induce failure but local applicability is uncertain. Sources of seismicity include multiple local faults and background seismicity due to larger but more distant faults. Source models are uncertain with respect to earthquake rates and magnitudes. The use of background seismicity models also indicates a lack of completeness relative to observed events. Moreover, seismic threats lack spatial certainty primarily due to data for liquefaction potential of levee and levee-foundation materials.

A key seismic threat is liquefaction-induced strength loss of levee fill and foundation materials. Liquefaction threats to levees lack spatial certainty primarily due to data for liquefaction potential of levee and levee-foundation materials. Even if levees remain standing following an earthquake, secondary failure mechanisms can threaten the long-term stability of levees and may have contributed to a large number of observed levee failures within approximately a year following the 1906 San Francisco earthquake. These secondary mechanisms include earthquake-induced levee cracks causing
accelerated and potentially destabilizing seepage, as well as accelerated rates of settlement for levees founded on peat. These are particularly important processes for Delta levees because they hold back water continuously. These failure mechanisms are not considered in current risk assessments for Delta levees and merit consideration in future work.

While fragility functions quantify the probability that any particular levee segment or reach will be damaged by shaking, the probability of survival of a levee system equals the probability that all of the segments survive the earthquake. Levees comprise a series system in which failure of any single component constitutes system failure especially since they constantly restrain water as in the Delta.

We also attempted to address questions about threats of high water. There is a strong link between climate change to coastal flooding and storms. Global circulation and hydrologic models indicate increased frequency of extreme flooding events, risk of coastal flooding and extreme water heights. An increase in the average sea level at Golden Gate will affect water levels in the Delta and thus increase hydraulic stress on Delta levees and contribute to an increase the annual likelihood of levee failure. Quantification of these effects merits further investigation.

We addressed the question about how subsidence affects levee probability of failure. The results of mechanistic modeling for varying levee geometry and marsh thicknesses indicate thinning of peat soils will increase probability of failure due to increased seepage forces. The spatial extent to which subsidence affects present-day and future levee failure probability is uncertain and requires further investigation. This is related to uncertainty in the extent of subsidence rates in organic soils. Additional uncertainty stems from tension-induced differential consolidation due to peat deformation.

Monetary consequences for historic and future levee failure and island flooding range from hundreds of millions to billions of dollars. Ongoing analysis of future risks focus on high-water failure and rely on revised hydrologic and hydraulic levee fragility curves. The DRMS study used a coordinated multidiscipline scientific effort to arrive at risk estimates. While the risk estimates have been challenged, the science and approach remain valid and applicable.

Integration of the best available science into analysis of future risk (including seismic and subsidence effects) for all potential hazards is recommended. Interdisciplinary science and levee practice in which there is skillful discussion and dialog among scientists, engineers and practitioners will be optimal.

Levee policy has focused on habitat associated with levees which will benefit from more coordinated and systematic monitoring of target species. However, levee habitat represents a small portion of potential Delta habitat restoration. In light of the pelagic organism decline and the economics of levee failure, repair and upgrades and negative and positive consequences of island flooding, prioritization of levee investments and development of tidal wetlands in appropriate areas of the Delta merits consideration and cost/benefit analysis. Better recognition and coordination is needed between the creation of high value tidal habitat and levee needs.

Improved description of levee upgrades and strength can likely result from analysis of the extensive database of borehole data and integration of these data to develop improved fragility analysis. Assessment of potential levee failure in general and seismic failure in particular, depend on a holistic and systematic assessment of island levees.

To address uncertainty in key areas, we suggest:

1. Improved description of levee upgrades and strength and improved probability of failure estimates based on recent information.
2. Improved potential estimates for seismic failure throughout the Delta.
3. Improved assessment of the effects of subsidence on levee stability and the spatial distribution of subsidence rates.
4. Improved understanding of levee habitat and the potential for integrating levee investments and prioritization with development of tidal habitat.
5. Improved assessment of effects of projected increased height and duration of Delta surface-water stages that account for water-supply scenarios and corresponding risks and benefits.
6. Additional hydrodynamic modeling to simulate a wide range of scenarios and corresponding risks and benefits to water supply and quality.

The March 2011 tsunami taught the Japanese that they must be ready for “Soteigai” (pronounced sew-ty-guy)—that which is outside our imagination.

ACKNOWLEDGMENTS

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NOTES

Cosio G. 2016. In-person conversation between Steven Deverel and Gilbert Cosio, MBK Engineers, about EM technology.


END NOTES

1. The Delta Levees Maintenance Subventions Program is a cost share program that provides technical and financial assistance to local levee-maintaining agencies in the Delta to maintain and rehabilitate non-Project and eligible project levees. Since the inception of the Subventions program in 1973, the California Department of Water Resources (CDWR) has reimbursed more than $175 million of eligible levee maintenance and rehabilitation work.

2. After 1950, incidents of levee failure from foundation or levee instability had doubled. Structural failures are often preceded by a localized partial failure involving 200 to 1,000 feet of levee. Partial failure includes settlement of the levee and the formation of cracks and sinkholes in the landward levee slope.

3. The CDWR inspected non-project levees around 52 islands and tracts, and concluded that the majority of the Delta levees did not meet Corps of Engineers maintenance standards; of the 52, 4 were rated “very poor,” 28 were rated “poor,” and 20 were rated “fair.” The CDWR identified more than 500 potential problem sites. The CDWR (1982) also examined the problems and feasibility of upgrading 537 miles of non-project levees surrounding 56 islands. The estimated cost for proposed levee-improvement plan implementation was $3.7 billion. Although the CDWR stated that the proposed levee improvement project would substantially reduce the frequency of levee failures, future levee failures are recognized to be inevitable.

4. As outlined in Section 12311 of the California Water Code.


8. These figures were based on evaluation of a range of PL 84–99 upgrade costs taken from multiple islands, including Twitchell, Sherman, Bouldin, and King, based on conversations with levee and CDWR engineers.

9. The data cutoff at 1950 was intentionally selected to remove the older historical events during which the levee configurations were dissimilar to the current levees.

10. Hopf (2011) separated out failures that occurred directly on levees designed to fail, on the “height-restricted” levees, and incidents on the smaller (less than 200 hectares) island levees, as well as those in the Suisun Marsh and those not designed to protect the agricultural, residential, and other economic activities of the main Delta agricultural islands.

11. Three standards specify minimum freeboard above the 100-year of 300-year flood frequency elevations, crown width, access, and waterside and landslide slope steepness: the short-term HMP, according to Water Code section 12984(a); the PL 84–99 of the U.S. Army Corps of Engineers (1988); and the Bulletin 192–82 standard (CDWR 1982).

12. Exit gradient = \((WLE_{sa} - WLE_{Ed}) / D\) where \(WLE_{Ed}\) and \(WLE_{sa}\) are the water level elevations in toe drain and underlying aquifer, respectively, and \(D\) is the vertical distance from the drainage ditch bottom to the bottom elevation of the organic soil.

13. On July 9, 1906: “Flood Conditions are Critical—Twitchell Island is inundated and Venice Island is filling. The levee that keeps the water off of the big Levi tract broke this morning. Unconfirmed report states that Sherman Island is filling with water due to a big levee break. Fears that the cross levee near the Levi tract on the lower division of Roberts Island may break.”
14. The moment magnitude scale denoted as $M_w$ or $M_L$ is used by seismologists to measure the size of earthquakes in terms of the energy released. The scale was developed in the 1970s to succeed the 1930s-era Richter magnitude scale ($M_R$). Even though the formulations are different, the new scale retains a similar continuum of magnitude values to that defined by the older one and is officially used by the USGS (Hanks and Kanamori 1979).

15. Generally, flood barriers are constructed above the crest stage prediction. Capping is usually accomplished with earth fill and plastic.

16. We estimated the 99.9th-percentile stage values for selected stations in the Delta and compared them with nearby levee heights, 100-year stages, and HMP and PL 84–99 elevations. The nearby levee crown elevations (based on the LiDAR data) ranged from 0.19 to 3.4 m above the 99.99th-percentile values. HMP and PL 84–99 elevations ranged from 0.4 to 2.4 m above the 99.99th-percentile values. The lowest differences were associated with the San Joaquin River at Andreas Landing and the San Joaquin River at Venice Island. Stations where there were sufficient data available from CDWR were: Sacramento River at Georgiana Slough; Sacramento River below Georgiana Slough; San Joaquin River at Andreas Landing; San Joaquin River at Venice Island; Mokelumne River at the San Joaquin River; Threemile Slough; and Old River at Bacon Island.

17. Slurry cut-off walls consist of a cement–bentonite soil mix placed in a trench about 1 foot wide from the crown of the base on the levee. The trench extends along the length of the levee.

18. Water Code sections 12311 through 12318.

19. For example, URS Corporation and Jack R. Benjamin and Associates, Inc. (2008) found that on southern Sherman Island, levee materials consisted of dredged loose to medium sand and silt. Beneath the levee is a thick layer of consolidated peat that is underlain by an approximately 25-foot-thick layer of silty clay, under which is a dense sand stratum. In contrast, on nearby western Webb Tract, levee materials consisted of peat, silty–clayey sand, and silt underlain by organic foundational materials that are in turn underlain by loose to dense silty sand and silty clay.


21. Electrical resistivity uses current in electrodes driven into the subsurface to measure subsurface resistivity, which can be correlated with lithology. Capacitively-coupled resistivity (CCR) is similar to electrical resistivity, except that resistivity measurements are made by using a set of cables or capacitive plates instead of metal stakes in the ground, as is the case with the EM technology.


23. Shear-wave velocity is directly proportional to the square root of the shear modulus and inversely proportional to the square root of the bulk density.


25. In lieu of or in combination with an extra-wide levee or an adjacent levee, a planting bench on the waterside levee slope may be installed to provide the appropriate depths and elevations for establishing channel–margin habitat. These benches may be stabilized with riprap (broken rock) covered with a mixture of soil and rocks that can support tidal marsh or riparian vegetation.
ABSTRACT

Over the past 15 years, the development and application of multi-dimensional hydrodynamic models in San Francisco Bay and the Sacramento–San Joaquin Delta has transformed our ability to analyze and understand the underlying physics of the system. Initial applications of three-dimensional models focused primarily on salt intrusion, and provided a valuable resource for investigating how sea level rise and levee failures in the Delta could influence water quality in the Delta under future conditions. However, multi-dimensional models have also provided significant insights into some of the fundamental biological relationships that have shaped our thinking about the system by exploring the relationship among X2, flow, fish abundance, and the low salinity zone. Through the coupling of multi-dimensional models with wind wave and sediment transport models, it has been possible to move beyond salinity to understand how large-scale changes to the system are likely to affect sediment dynamics, and to assess the potential effects on species that rely on turbidity for habitat. Lastly, the coupling of multi-dimensional hydrodynamic models with particle tracking models has led to advances in our thinking about residence time, the retention of food organisms in the estuary, the effect of south Delta exports on larval entrainment, and the pathways and behaviors of salmonids that travel through the Delta. This paper provides an overview of these recent advances and how they have increased our understanding of the distribution and movement of fish and food organisms. The applications presented serve as a guide to the current state of the science of Delta modeling and provide examples of how we can use multi-dimensional models to predict how future Delta conditions will affect both fish and water supply.

KEY WORDS

Hydrodynamic modeling, UnTRIM, SUNTANS, SCHISM, RMA2, Delft3D, low salinity zone, X2, fish movement, fish distribution, food organisms, water supply, future conditions.

INTRODUCTION

It is notable that when the State of Bay–Delta Science 2008 (Healey et al. 2008a) was published, newly emerging multi-dimensional models of the Sacramento–San Joaquin Delta (Delta) merited only a short discussion in the chapter dealing with water
supply (Healey et al. 2008b). As a result of significant advances in both computational power and the development of multi-dimensional models over the past decade, multi-dimensional models of the Delta have been applied much more widely, and have proved their usefulness in helping us to better understand not just issues relating to water supply, but also the complex interactions between physics and biology that drive the distribution and movement of fish and food organisms.

Before delving into the details of multi-dimensional physical modeling, it is important to recognize at the outset that the ability of any model to predict physical, chemical, and biological processes varies, and strongly depends on what is being modeled and on the time-scale of interest. Roughly speaking, the shorter the time-scale and the more physical the process, the better the model will perform (Figure 1). For example, tidal water levels and currents can generally be calculated with a high degree of precision. Conversely, modeling biological or ecological processes that evolve over longer time scales, is a significantly more challenging modeling task. On this end of the spectrum, predicting how a marsh will change over several decades as a result of the interactions among sediments, vegetation, and fauna remains a remarkably hard modeling challenge, particularly given the uncertainty with how the forcing variables will change in the future.

The challenge in modeling physical processes in the Delta is its complexity. Flows in this geometrically complex domain are influenced by tides; winds; freshwater inflows from tributaries that are influenced by rain, snowmelt, and reservoir operations; in-Delta agricultural diversions and return flows; the construction and operation of temporary barriers; permanent operable control gates, and the operation of the Central Valley Project (CVP) and State Water Project (SWP) exports. Some aspects of this system can be represented by water accounting models such as CalSIM II (Draper et al. 2004; CDWR 2013, Sections A and B), which operates on a monthly time-step to balance the requirements of reservoir storage and flow releases, operation of the SWP and CVP exports, and the maintenance of applicable water quality objectives. Other aspects of the system, such as salinity intrusion, result from complex interactions of tides, wind, and freshwater outflow, and require a three-dimensional (3-D) model operating on a short time-step to accurately represent vertical and horizontal circulation processes. Thus, the type and complexity of the model that is needed to model a system depends to a large degree on the type of processes being evaluated, and the questions that need to be answered. Table 1 provides an overview of some of the primary physical processes that are important drivers in the Delta, and an assessment of whether the processes are explicitly represented by 1-D, 2-D, and 3-D models. In Table 1, and throughout this chapter, we use 2-D to indicate depth-averaged 2-D models; however, we recognize that for some applications 2-D can also indicate

![Figure 1](Schematic view of the spectrum of modeling physical and biological processes in the Delta)
laterally-averaged models. The ability of a model to represent some processes depends both on resolution and dimensionality, so for some processes listed in Table 1, multiple colors are shown to indicate that the ability of a class of models to represent this process may also be resolution and algorithm-dependent. Although this type of assessment is somewhat subjective, Table 1 can serve as a resource for selecting the type and dimensionality of model that may be necessary to evaluate specific types of applications.

When a model cannot (because of dimensionality or resolution) or does not (because of numerical formulation) represent a process, that process can often be successfully parameterized using integrated processes, simplified closure terms or approximations, some of which involve tunable numerical coefficients. One example of this is the so-called “diffusion analogy” described by Fischer et al. (1979) for dispersion in a 1-D model such as DSM2 in order to accurately represent salinity intrusion. In this case, the primary mechanisms responsible for salinity intrusion include gravitational circulation (vertical), lateral shear, and tidal trapping (horizontal), which are 3-D and 2-D processes. Instead of modeling these directly, dispersion parameters are tuned on each channel segment to represent salt-intrusion processes that are not resolved in 1-D. In 3-D, cross-sectional variations leading to dispersion are modeled explicitly. A more modest diffusion analogy is still made in 3-D, in the form of turbulence closures of varying complexity that determine rates of vertical mixing (e.g., see Wang et al. 2011). Horizontal turbulent mixing is generally neglected because it has almost always been found to be weaker than numerical diffusion associated with the numerical scheme(s) used to represent advection (Gross et al 1999; Chua and Fringer 2011).

Although physical simplification and approximation can be effective for simulating conditions within historical bounds, they limit the predictive capability of the model if the system changes significantly. For example, a 1-D model cannot be used to determine the effects of sea level rise since the dispersion coefficients tuned for existing sea levels would not be applicable under those future conditions. To work around this limitation for the Bay Delta Conservation Plan (BDCP) project, 3-D model simulations (MacWilliams and Gross 2010) were used to simulate salinity under future conditions with sea level rise, and the dispersion coefficients in the 1-D model in the western Delta were recalibrated to match the

Table 1  Physical processes, and an assessment ability of 1-D, 2-D, and 3-D models to represent these processes. Key: green shading = yes; yellow shading = partially; red shading = no.

<table>
<thead>
<tr>
<th>Process name</th>
<th>1-D</th>
<th>2-D</th>
<th>3-D</th>
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<tbody>
<tr>
<td>Tides and water levels</td>
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<td>Mixing at junctions</td>
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<td>Mixing in open water embayments</td>
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<td>Tidal trapping</td>
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<td>Wind-driven circulation</td>
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<td>Gravitational circulation / salinity intrusion</td>
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<td>Secondary circulation</td>
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<td>Sediment routing</td>
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<td>Erosion and deposition / morphological evolution</td>
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<td>Temperature (depth-averaged)</td>
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<td>Temperature stratification</td>
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<td>Passive particle tracking (larva)</td>
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<td>Active particle tracking (fish)</td>
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salinity intrusion predicted by the 3-D model (CDWR 2013). The alterations were modest, and in this way, exploration was possible of an 80-year hydrology under scenarios that incorporated the effects of sea level rise on salinity intrusion. Within the design community, such multifidelity combinations of detailed and simplified models of physics have been formalized and exploited (see Robinson et al. 2008 and references therein); in the Bay–Delta such an approach may be a necessity in cases where insights arising from multi-dimensional modeling of secondary flows need to be scaled to the full domain and analyzed for effects over decades.

Because multi-dimensional models do not rely as heavily on tuning parameters and instead represent the inherent physics of the system, they are often more suitable for evaluating the effects of significant changes to the system. Of course, the internal details and resolution have to be correct in order for the physicality argument to hold, and often some of the specific sub-processes in a 3-D model are hard to validate or require study-specific considerations and field study. An example of this described below for Franks Tract would be the applicability of standard estuarine turbulence, which is well studied and appropriate in the stratified lower estuary but isn’t particularly tailored for a shallow, vegetation-clogged channel or open water body. One other noteworthy challenge to accurate 3-D modeling is the spatial and temporal variability in bottom drag associated with mobile beds. For example, Fong et al. (2009) found that the bottom drag coefficient in Threemile Slough for flow from the Sacramento River into the San Joaquin River was three times larger than for the flow in the opposite direction, indicating a complex interaction of tidal currents and bedform morphology that presumably varies on sub-tidal and seasonal time scales.

The increased use of multi-dimensional models reflects advances in modeling made since the 2010 Interagency Ecological Program (IEP) modeling review (Blumberg et al. 2010). Worldwide, the estuarine modeling community has trended towards the use of unstructured (sometimes adaptive) meshes and semi-implicit algorithms that allow cross-scale, multi-year, hydrostatic circulation studies at locally medium resolution over a domain the size of the full Bay–Delta. Recent work continues on higher fidelity algorithms, including higher-order algorithms for problems that involve fronts or eddying regimes and non-hydrostatic modeling of phenomena such as secondary circulation and internal waves. More explicit representations of turbulence such as Large Eddy Simulation (LES) or even Direct Numerical Simulation (DNS) are also becoming more computationally feasible in models that cover smaller domains. We expect more near-field applications of higher-fidelity modeling to be a notable contribution in the next edition of the State of Bay–Delta Science, but will focus here on the more commonly applied 1-D, 2-D, and 3-D models that represent the state-of-the-art of modeling in the estuary, where flow is assumed to be hydrostatic and Reynolds-averaged. At the other end of the spectrum, developers and users of circulation-scale 3-D models in the last 5 years have showcased coarsening strategies such as low horizontal-resolution meshes with subgrid bathymetry (Casulli 2009; MacWilliams et al. 2016) and adaptive vertical meshing (Zhang et al. 2015). The demand for these types of features underscores the limiting role computational cost still plays, particularly for simulating decadal or longer periods in 3-D. In developing a multi-dimensional numerical model, there are many important decisions that must be made relating to spatial and time accuracy, stability, energy and momentum conservation, volume and mass conservation, grid structure, and where variables are defined on the grid. Although these aspects of numerical methods are critically important in developing numerical models, they are beyond the scope of this paper. Details about the numerical methods used in each model discussed below can be found in the references cited for each model.

**BAY–DELTA MULTI-DIMENSIONAL MODELING BACKGROUND**

Hydrodynamic modeling of the Delta started in earnest in the late 1970s when under U.S. Bureau of Reclamation (USBR) contract, the late Professor Hugo B. Fischer of the University of California, Berkeley, developed what is now known as the Fischer Delta Model (FDM), a model that represented the Delta as a network of 1-D channels. Later, the California Department of Water Resources (CDWR) developed its own similar 1-D model, DSM2, to which a particle-
tracking model was added. DSM2 is still widely used in planning simulations for the Delta. The model enjoys a wide user base, couples well to the statewide planning model CalSIM II, routes water through the Delta accurately over a wide range of forcing, and because of its ubiquity and speed has perhaps the best understood performance over decades. DSM2 is still widely used for contemporary applications, including the California Water Fix Biological Assessment (CDWR 2016a), and for quantifying benefits or adverse effects that could result from water storage projects proposed for the Water Storage Investment Program (CWC 2016). In modern usage, the DSM2 model is often modified or informed by multi-dimensional models in areas such as sea level rise that involve coupling to the ocean and complex circulation.

Multi-dimensional modeling of the Delta began in the mid-1990s with applications of both RMA10 (DeGeorge 1996) and TRIM2D (Monsen 2000), although earlier 2-D models of Suisun Bay (Smith and Cheng 1987) and in particular Burau et al. (1993) should be noted. Some of the earliest multi-dimensional modeling of San Francisco Bay (bay) was done with the structured grid TRIM2D and TRIM3D models (Casulli and Cheng, 1992; Casulli and Cattani 1994). TRIM2D was used in the San Francisco Bay Physical Oceanographic Real-Time System (SFPORTS) for many years (Cheng and Smith 1998) and was applied to the Delta by Monsen (2000).

One of the earliest demonstrations of the power of multi-dimensional models to affect conceptual thinking about how the Bay–Delta physically works is the 2-D (depth-averaged) modeling of Suisun Bay discussed in Burau et al. (1993). Using the TRIM2D model (Cheng et al. 1993), strong horizontal mixing associated with channel–shoal velocity gradients was shown to rapidly mix scalars that might serve to mark a nominal estuarine turbidity maximum (ETM), suggesting that the classical picture of an ETM does not apply to Suisun Bay. This was an important factor in leading the U.S. Environmental Protection Agency (USEPA) flow workshops in the early 1990s to focus on the general aspects of how flow and salinity were related, rather than attempting to look at detailed management of the position of an ETM as had originally been proposed. Moreover, the striking visuals of the model results that Jon Burau presented (then and more recently) also played a role in shifting the conceptual understanding of Delta hydrodynamics away from one concerned only with sub-tidal, approximately riverine flows to one in which tidal time-scale variability is dominant. Similarly, 2-D finite element modeling of the salinity response to levee failure done by Resource Management Associates, Inc. (RMA) for the Delta Risk Management Study (DRMS) dramatically illustrated the risk posed by earthquakes to the use of the Delta as a water conveyance (CDWR 2009).

An important aspect of a multi-dimensional model application is the fact that it can lead to significant insights about system-level functioning. For example, TRIM2D modeling by Monsen et al. (2007) showed how the contributions of various sources of water to the Delta (e.g., the Sacramento River or the San Joaquin River) varied with position within the Delta and with different combinations of export pumping rate, gate operation, and barrier placement. Using the same model, Monsen (2000) also noted that closure of the Delta Cross Channel (DCC) did not eliminate entrainment into the pumps of particles (e.g., small fish) carried down the Sacramento River towards Suisun Bay; instead, it shifted the path of these particles to the pumps through Threemile Slough rather than through the DCC.

Notably, much of this earlier modeling was computationally constrained. For example, the 50-m Cartesian finite difference grid used by Monsen (2000) tended to lead to unrealistically slow flows in many of the smaller channels of the Delta that could not be resolved on a 50-m grid. Similarly, Gross et al. (2009) relied on a “false Delta” to represent the Sacramento–San Joaquin Delta in their model of the bay because the Delta could not be represented on the 200-m grid that was necessary for computationally feasible 3-D simulations of the San Francisco Estuary (estuary). Moreover, because of limited disk storage space for model output, biogeochemical or individually based models of organisms needed to be coupled directly to the hydrodynamic model. Now, with the easy availability of relatively inexpensive, powerful desktop computing, routine use of medium-sized clusters, and very inexpensive disk storage, much larger and more detailed grids can be run, with the results archived for use and re-use with other models, such
as the off-line coupling with particle-tracking models (e.g., Gross et al. 2010; Kimmerer et al. 2014). The result of these technological advances—as well as the introduction of unstructured grids that permit high resolution in areas of interest while allowing much coarser grids away from the area of interest—has facilitated computing flows in a domain that extends from the Gulf of the Farallones to riverine reaches of the Sacramento and San Joaquin rivers while still resolving the relatively narrow channels that make up much of the Delta (e.g., MacWilliams et al. 2015). Applications of these types, and the important lessons learned about the system from model application, are described in the following sections.

Because of the complexity and small size of many channels in the Delta, a finer grid resolution is needed to resolve these channels than is needed in other portions of the bay. The application of unstructured grids allows for the use of the finer resolution necessary to resolve small channels in the Delta, while using larger grid cells in the broader areas of the bay. Thus, the development and application of unstructured grid models such as UnTRIM, SUNTANS, SCHISM, RMA2, and Delft 3D FM led to the development and application of multi-dimensional models that span all of the bay and the entire Delta.

**UnTRIM**

UnTRIM (Unstructured nonlinear Tidal Residual Inter-tidal Mudflat) is a 3-D hydrodynamic numerical model that solves the Navier–Stokes equations on an unstructured horizontal grid and z-level vertical grid. The governing equations are discretized using a finite difference–finite volume algorithm. Although UnTRIM is a proprietary model, a complete description of the governing equations, numerical discretization, and numerical properties of UnTRIM are provided in Casulli and Zanolli (2002, 2005), Casulli (1999, 2009), and Casulli and Walters (2000). The most recent version of UnTRIM introduced “subgrid” bathymetry, which allows the bathymetry to be applied within each grid cell and on each grid face at a resolution higher than the hydrodynamic model grid itself (Casulli 2009; MacWilliams et al. 2016). The use of subgrid bathymetry allows for accurate representation of channel geometry with larger grid cells (which allow for larger time-steps) and relaxes some constraints on grid quality since grid lines no longer need to follow the coastline (since grid cells can be partially wet).

The UnTRIM hydrodynamic model has been implemented in the bay and Delta to simulate tides, inflows, and water diversions; water surface elevations; 3-D velocities; and salinity throughout the Bay–Delta system (MacWilliams et al. 2015). The UnTRIM Bay–Delta model was the first 3-D model applied to the Bay–Delta system that spanned the entire estuary from the Pacific Ocean through the Sacramento River and the San Joaquin River (MacWilliams et al. 2007, 2008, 2009, 2015). The UnTRIM Bay–Delta model has been applied as part of the Delta Risk Management Strategy (MacWilliams and Gross 2007), several studies to evaluate the mechanisms behind the Pelagic Organism Decline (e.g., MacWilliams et al. 2008), and the Bay Delta Conservation Plan (MacWilliams and Gross 2010). The UnTRIM Bay–Delta model has also been applied for a range of studies by the U.S. Army Corps of Engineers (MacWilliams and Cheng 2007; MacWilliams et al. 2009, 2012a, 2014).

The UnTRIM Bay–Delta model has also been coupled with the Simulated WAves Nearshore (SWAN) wave model (Booij et al. 1999; SWAN Team 2009) and the SediMorph sediment transport and seabed morphology model (BAW 2005; Weilbeer 2005), as a fully-coupled hydrodynamic-wave-sediment transport model that spans all of the bay and the Delta. This model has been used in studies of sediment transport that support the San Francisco Bay Regional Dredged Material Management Program (MacWilliams et al. 2012b; Bever and MacWilliams 2013, 2014; Bever et al. 2014) and for the Prospect Island Tidal Habitat Restoration Project (DMA 2014a).

**SUNTANS**

SUNTANS (Stanford Unstructured Nonhydrostatic Terrain following Adaptive Navier–Stokes Simulator) is an open-source 3-D circulation model developed at Stanford University that computes flows on an unstructured grid using a finite-volume method (Fringer et al. 2006; Wang et al. 2009; Wolfram et al. 2016). SUNTANS is highly parallelized and designed to be used on large parallel computing
clusters. At its heart, SUNTANS is similar to the 3-D unstructured-grid UnTRIM model (Casulli and Walters 2000; Casulli and Stelling 2010), and uses two key approaches developed by Casulli (1990, 1999): (1) a semi-implicit free surface; and (2) a fractional step method to efficiently calculate the effects of non-hydrostatic pressures such as might be important at channel junctions or in channel bends. It includes algorithms for sediment transport and, through coupling with the SWAN model (Booij et al. 1999), surface wave effects on bottom stresses and flows (Chou et al. 2015). Various grids have been created for parts and the entirety of the Bay–Delta in the context of applications to the South Bay Salt Pond restoration (Hsu et al. 2013) as well as for detailed flow modeling in the Delta (Wolfram et al. 2016).

**SCHISM**

SCHISM (Semi-Implicit Cross-Scale Hydroscience Integrated System Model) is an open source semi-implicit model (Zhang et al. 2015, 2016) most recently extended from the Semi-implicit Eulerian–Lagrangian finite-element (SELFIE) model (Zhang and Baptista 2008) and now an open source project led by the Virginia Institute of Marine Science (VIMS). SCHISM is semi-implicit in time, solving the Reynolds-averaged hydrostatic primitive equations using several of the same strategies as UnTRIM and SUNTANS. However, SCHISM treats a number of the steps in a novel way, by combining mass and momentum analytically (rather than algebraically) and in discretizing the resulting equation using finite elements. As a result, SCHISM is able to use a terrain-conforming vertical gridding system and more relaxed horizontal gridding constraints, which can be beneficial in channels with complex bathymetry. The SCHISM suite includes coupling to a wind-wave model, 3-D sediment, 2-D morphology, and several nutrient models. Only the nutrient model has been applied in the Bay–Delta.

The Bay–Delta SCHISM project is an application of 3-D SCHISM that allows cross-scale, multi-dimensional flow and transport in the Bay–Delta to be studied (Ateljevich et al. 2015). Although in the public version of the model the entire domain is represented in 3-D, VIMS and DWR are on the verge of releasing a version with more flexible vertical coordinates (Zhang et al. 2015) that allow regionalized tuning of vertical resolution and selective use of 2-D. Bay–Delta SCHISM has been calibrated on flow, water surface elevation, salt, and temperature over the full Bay–Delta (Ateljevich et al. 2015) and on nutrients for the Bay as part of the Salmon Ecosystem Simulation and Management Evaluation (SESAME) project. The application is currently being adapted by the National Oceanic and Atmospheric Administration (NOAA) as the standard for datum conversions as part of the VDatum (vertical data transformation software) program. CDWR has applied SCHISM to a variety of drought-related problems, including tidal effects, visualization and quantification of mixing processes for salt, circulation in Franks Tract, and near-field velocity resulting from the installation of drought barriers, which is discussed in more detail below. The base model resolution was chosen to resolve horizontal variation of primary velocity realistically along the larger channels in the bay and Delta; the model can run at speeds of 0.25 to 0.5 years per day of computation for a 2009 salinity and temperature benchmark on clusters at CDWR (typically 128 cores per simulation) and VIMS (typically 144 cores per simulation). Although the use of high performance computing can be difficult for non-institutional users, one advantage of a parallel computational approach to nutrient problems is that the transformation calculations involve little interprocess communication and scale efficiently.

**RMA2**

RMA has developed a series of finite element models for 1-D, 2-D, and 3-D simulation of flow, salinity, water quality, and sediment transport in streams and estuaries. The RMA Bay–Delta model of the bay and Delta is a coupled 1-D and 2-D model that uses RMA2 and RMA11 computational engines. The model domain extends either from Martinez through the Delta or from the Golden Gate through the Delta, with the bay, western Delta, and Franks Tract portions represented in 2-D, and the remaining channels of the Delta represented in 1-D. The RMA Bay–Delta model has been calibrated and applied in many previous studies such as the Flooded Islands Pre-Feasibility Study (RMA 2005), the Delta Risk Management Strategy project (URS Corporation
2011), and the Prospect Island Tidal Restoration study (RMA 2012). RMA has also more recently developed a 3-D model of the pre-development estuary using UnTRIM (RMA 2015; Andrews et al., submitted).

**Delft3D**

UNESCO–IHE (Institute for Water Education), Deltares, and the USGS (U.S. Geological Survey) have developed the Bay–Delta model within the Computational Assessments for Requirements for Change for the Delta Ecosystem (CASCaDE) II project, applying the Deltares Delft3D-FM (flexible mesh) software. Delft3D-FM is an unstructured version of Delft3D, a widely-used hydrodynamic modeling software suite developed by Deltares of the Netherlands. Delft3D-FM, in contrast to Delft3D, utilizes a finite-volume, unstructured grid framework, allowing for variable resolution in regions of complex topography and bathymetry, and in regions where forcing functions and responses change rapidly (Kernkamp et al. 2011). The unstructured grid framework allows for polygon-shaped grid cells of arbitrary degree in 2-D (latitude and longitude) space, and includes 1-D channel networks. Model capabilities include 3-D salinity and temperature transport and dynamics, an atmospheric heat flux model driven by spatial fields of relative humidity, air temperature, and cloudiness for water temperature dynamics, dynamic wind-wave coupling, and formulations for sediment transport and morphodynamics.

Delft3D-FM has been applied to the San Francisco Bay–Delta region for the evolution of hydrodynamics, salinity, and temperature dynamics. This model is applied as part of the USGS-led CASCaDE II project that applies a linked modeling approach to provide scientific basis for regional policy decisions about water supply and Bay–Delta ecosystem health. The Delft3D-FM model domain includes the Pacific Ocean north to Point Reyes; the south, central and north San Francisco bays, the lower Yolo floodplain up to Fremont Weir; numerous channels of the north, central and south Delta; and the flooded islands of Franks Tract and Mildred Island. Links have been created with Deltares-developed sediment (Achete et al. 2015), phytoplankton, and habitat suitability models, in which spatial and temporal maps of hydrodynamics serve as the primary driver of the other models. This linked model approach has been successfully applied in a proof-of-concept framework to generate spatial habitat maps for key Bay-Delta species, including *Corbicula fluminea* and Delta Smelt (*Hypomesus transpacificus*). Related to this work, Knowles and Lucas (2015) describe the initial development of a new phytoplankton model that provides a new tool to explore links between physical and ecological processes in the Bay–Delta. We expect that this model and the work done through the CASCaDE project will be usefully applied in the future for many of the types of applications we describe in later sections.

**Other Multi-Dimensional Models**

Additional 2-D and 3-D models have been applied to simulate parts of the estuary, including Si3D (Semi-implicit 3D), Mike–21, FVCOM (Finite Volume Coastal Ocean Model), and EFDC (Environmental Fluid Dynamics Code). The Si3D model was developed by the USGS (see Smith 1997) for application to the Delta and has been applied to look at dissolved oxygen dynamics in the Stockton Deepwater Ship Channel (DWSC) (Doyle et al. 2008; Monismith et al. 2008). Since 2014, FVCOM has been applied as part of the San Francisco Bay Operational Forecast System to provide nowcast and forecast guidance of water levels, currents, water temperature, and salinity in the bay and at its entrance (Peng et al. 2014; NOAA 2016). However, the FVCOM model extends into only a small portion of the western Delta, so it is not discussed in detail. Other multi-dimensional model applications have focused largely on salt transport in the bay (Gross et al. 2009; Chua and Fringer 2011), the effects of sea level rise in the bay (Hollieman and Stacey 2014), and sediment dynamics in the bay (Ganju and Schoellhamer 2009; van der Wegen and Jaffe 2013) without resolving the Delta. The applications discussed below will focus specifically on multi-dimensional modeling in the Delta, primarily with a view towards consequences of habitat and flows for the transport of biota in and through the Delta.
MULTI-DIMENSIONAL MODELING APPLICATIONS

As seen in Table 1, 2-D and 3-D models both represent many of the physical processes that drive flow and transport processes in the Delta. The following two sections highlight recent studies that have investigated the ability of 2-D and 3-D models to represent specific processes.

Modeling Junction Dynamics in 1-D, 2-D, and 3-D

A key aspect of transport and dispersion in the Delta appears to be the dispersive effect of flows through the many junctions of the Delta. Analyzing temperature dynamics in the Stockton DWSC, Monismith et al. (2009) found that the dispersive flux of heat from the San Joaquin side of the Delta into the Bay required effective dispersion coefficients of ca. $1000 \text{ m}^2 \text{s}^{-1}$ whereas dispersion in the DWSC itself was ca. $30 \text{ m}^2 \text{s}^{-1}$ (c.f. Schmieder et al. 2008). Monismith et al. (2009) argued that the nominal mechanism for this behavior was chaotic dispersion: dispersion associated with flow splits at the junctions. Computations made using the particle-tracking model STARWalker (Stanford Three-dimensional Augmented Random Walker) reported in Sridharan (2015) show the importance of flow behavior at junctions. STARWalker uses flows computed by DSM2. The CDWR Java Particle-Tracking Model (JPTM) used with DSM2 (e.g., see Kimmerer and Nobriga 2008) essentially assumes complete mixing at junctions. In contrast, STARWalker can be run with either complete mixing at junctions or by allowing particles to follow streamlines that are computed using a highly idealized potential flow model of each junction in DSM2.

The way junction conditions (complete mixing vs. streamline following) influence particle dispersion and paths through the Delta was evaluated by Sridharan (2015), who found significant differences in spatial distributions of particles released at various source locations, such as Rio Vista, depending on the junction model. In particular, Sridharan (2015) found that there were large differences in the timing and rates of particle arrival at the export pumps between model runs that used complete mixing and model runs that used streamline following. This is an important practical metric of model performance. A particularly striking example of this difference was found for a model of Delta Smelt salvage during the spring of 2000, with the STARWalker’s salvage prediction being significantly better ($r^2=0.48$) with streamline following than with complete mixing at junctions ($r^2=0.01$). We note that 3-D particle tracking simulations based on UnTRIM and the Flexible Integration of Staggered-grid Hydrodynamics Particle-Tracking Model (FISH-PTM) simulations reported by Gross et al. (2010) appear to agree even better with salvage observations.

Thus, proper description of flows at junctions may be important to predicting the transport pathways of any quantity, e.g., larval fish, through the Delta. Observations of flows in the Georgiana Slough/Mokelumne River junction reported in Gleichauf et al. (2014) show the complexity of these flows, which can include separation behavior that was well represented by a highly resolved 3-D SUNTANS model of the region (see Schoellhamer et al. 2016). Beyond showing fidelity to observations, these model results also revealed a reach-scale feature of the flow not readily observable by fixed instrumentation or limited transecting: the creation of interspersed patches of fluid from different sources as a result of the phasing of flows that enter and leave the junction as acted on by lateral mixing in the junction (Figure 2). This nicely illustrates that multi-dimensional modeling, beyond its utility at addressing straightforward engineering questions, can also provide important new conceptual insights about physical processes in the Bay–Delta.

There are, necessarily, trade-offs between model resolution and model scale, raising the question of what level of detail is necessary and sufficient to properly model junction effects on system level dispersion. Wolfram et al. (2016) examined this issue through a series of model studies of the Georgiana Slough/Mokelumne River junction. The most detailed computation used very high resolution (an average of 3-m horizontal and 0.8-m vertical) and included the effects of non-hydrostatic pressures—an level of detail that would be difficult to replicate at the scale of the whole Bay–Delta and for seasonal time scales if computed on a desktop workstation. Based on comparison to observations, this model run (the “base case”) was taken to represent actual conditions. Other
Figure 2  SUNTANS simulation for June 1, 2012, at: (A) 12:30; (B) 13:00; (C) 14:00; (D) 16:30; (E) 18:30; and (F) 21:30. Red signifies water that originally came from Georgiana Slough (GS) and blue represents water coming from the north Mokelumne (NMK). The insets show the mean flow in each branch of the junction during the simulation. Source: Gleichauf et al. (2014).
particles travelled farther downstream and entered the interior Delta through either Threemile Slough or through the confluence of the Sacramento and San Joaquin rivers. Moreover, by examining which paths were important for certain given conditions (e.g., Export:Inflow ratios, gate operations, and barriers), this type of analysis could be used to design operational strategies to reduce entrainment at the export pumps. Although this example draws on the use of a well-established 1-D model, it illustrates what could be done quite profitably with the flow fields and particle tracks derived from multi-dimensional models, likely with more certainty since the representation of the physical system is more accurate when all spatial variations are accounted for explicitly.

High Resolution Modeling of Secondary Circulation

Numerical simulations of circulation and residence time in Clifton Court Forebay (CCF) by MacWilliams and Gross (2013), provide an interesting example of a relatively simple problem, where the flow dynamics are inherently 3-D. Clifton Court Forebay (CCF) is a regulating reservoir in the southern Delta that is used to improve operations of the California State Water Project (SWP) Harvey O. Banks Pumping Plant and water diversions to the California Aqueduct (Clark et al. 2009). Inflows to CCF are controlled by five radial gates, and outflows from CCF include water exports from the Banks Pumping Plant to the SWP and from CCF to the Byron Bethany Irrigation District. The purpose of these simulations was to provide a better understanding of circulation patterns, flow pathways, and residence time in the CCF to support ongoing studies of pre-screen loss and fish facility efficiency for Delta Smelt at the SWP export facilities. During windy periods, the flow within the CCF is highly 3-D, with strong wind-driven surface velocities driving a counterclockwise subsurface gyre (Figure 3). During periods of high winds, this gyre in CCF resulted in significant mixing, and increased the range of estimated transit times from the radial gates to the Banks Pumping Plant. The vertical variability of wind-driven velocities also resulted in mixing that was manifested by a large range of estimated residence times for high-wind conditions. In contrast, during higher-export and low-wind conditions, residence times were much shorter, and

model runs included a 3-D run with coarser resolution (6-m horizontal) and hydrostatic pressures and a 2-D run with the same resolution as the coarse 3-D run. To assess how well the models handled mixing at the junction, for each model run Wolfram et al. (2016) computed the fraction of fluid from each of the four sources in the four receiving channels, and compared the results to the base case. The case of complete mixing—i.e., what is used in standard DSM2-PTM modeling (e.g., Kimmerer and Nobriga 2008)—can be computed analytically and so it was not necessary to make separate DSM2-PTM runs for this case. Based on computed correlation between the results of the various model runs and the base case, Wolfram et al. (2016) found that: (1) the lower-resolution, hydrostatic 3-D case provided nearly the same result as did the base case ($r^2=0.98/0.99$ with the DCC open/closed); (2) the 2-D run was nearly as good as the 3-D run ($r^2=0.94/0.95$); and (3) the complete mixing model was somewhat less accurate ($r^2=0.61/0.71$). In examining the different models, Wolfram et al. (2016) concluded that the tidal timescale details have less effect on mixing by the junctions than do the bulk features of the flow and differences in sub-tidal flow structure (in this case caused by operation of the DCC). As a consequence, simplified models like the DSM2-PTM provide some ability to model Delta-scale dispersion, especially if streamline following is used at junctions. Nonetheless, the work of Wolfram et al. (2016) suggest that substantial improvement in fidelity of the results can be obtained using 2-D models, although the extra computational effort of high resolution 3-D models may not provide much improvement beyond that of 2-D models in some applications. This demonstrates that for modeling some processes that can be adequately represented in 2-D (Table 1), a 3-D model may not be necessary. However, physical processes that are themselves inherently 3-D—e.g., stratified flows associated with salinity gradients in Suisun Bay and the western Delta, or temperature stratification in the DWSC—require 3-D modeling (Table 1).

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increased with distance from the Banks Pumping Plant. This example demonstrates that, particularly under windy conditions, a 2-D model of CCF would produce significantly different circulation patterns and residence times than a 3-D model. Even though wind effects can generally be represented by a 2-D model (Table 1), in some cases wind can have a strong influence on 3-D processes that cannot be represented using a 2-D model.

USING MODELS TO UNDERSTAND OBSERVATIONAL DATA

Because models can give us much greater spatial and temporal data resolution than we can get from monitoring at a limited number of times and stations, models can enhance our understanding of both physical and biological monitoring data. This section presents two examples that highlight how combining modeling with observational data can help to improve our understanding of the observations.

What We Have Learned from Models about the X2 Data Sets

Abundance or survival of several estuarine species have historically been positively related to freshwater flow, as indexed by the position of the daily-averaged 2-psu isohaline near the bed, or X2 (Jassby et al. 1995; Kimmerer et al. 2009, 2013). As noted in the State of Bay–Delta Science 2008, X2 is used in managing flow into the estuary, and is considered a measure of the physical response of the estuary to changes in freshwater flow (Kimmerer 2002). In most applications, X2 is estimated using either autoregressive equations (Jassby et al. 1995; Monismith et al. 2002; Gross et al. 2009; MacWilliams et al. 2015) or surface salinity from a small set of fixed observation stations that are typically near shore (CDEC 2016). As MacWilliams et al. (2015) discussed, there are limitations to both approaches, particularly for low and high values of X2, and models have been instrumental both in increasing our understanding of the relationship between X2 and flow and in improving these calculations.

One application of a 3-D circulation model to the Bay–Delta is the study of unsteady salinity intrusion in northern San Francisco Bay, i.e., X2 variability with flow, reported in MacWilliams et al. (2015). In this case, MacWilliams et al. (2015) used the model results to develop an improved auto-regressive model, like that described in Jassby et al. (1995), suitable for use in planning. The value of modeling in this case is two-fold: (1) whereas low-flow values of outflow have high uncertainty, the model calculates X2 variability for specified (hence known) flows, and thus, the 3-D model (and possibly the new autoregressive equation) provides the ability to invert measured salinities at low flow to compute outflow; and (2) computed salinity fields for very high flows provide for accurate determination of X2 at high flow for cases where the existing salinity monitoring

![Figure 3](image-url) Predicted (A) surface and (B) subsurface circulation patterns in CCF on June 25, 2008, at 00:00 PST, during a period with relatively high winds, when the radial gates were closed and the Banks Pumping Plant was not operating. Source: MacWilliams and Gross (2013).
network has only coarse resolution or during periods when X2 is downstream of all the salinity stations. Moreover, since high flows are rare, the model can be used to represent conditions that are rarely observed.

Two interesting implications of the MacWilliams et al. (2015) autoregressive model are that: (1) the time constant of adjustment to flow changes is large when flow is small and small when flow is large—behavior that Denton (1993) also incorporated into his so-called G model; and (2) the steady-state response to flow is such that $X_2 \sim Q^{-1/5}$ whereas Monismith et al. (2002) found that $X_2 \sim Q^{-1/7}$ based on surface salinity measurements described in Jassby et al. (1995) and USGS Polaris transect data. It is not clear at this time which power law is correct. Monismith (submitted) developed a time series model based on simplified dynamics—including $X_2 \sim Q^{-1/6}$—that appears to be as accurate as that of MacWilliams et al. (2015); accordingly, differences in power law behavior may not be of much practical purpose. A potentially more substantive concern with 3-D model results for high flows is that the highly stratified conditions that exist at high flows are difficult to model accurately. Numerical diffusion associated with computing vertical momentum and salt transport and also with horizontal transport of salt can weaken the net salt transport associated with gravitational circulation (Chua and Fringer 2011). This weakening of salt transport would tend to increase the sensitivity of modeled X2 to flow.

Using Models to Investigate Fish Survey Data Sets

Another example of how hydrodynamic models have been used to increase our understanding of long-term data sets is the quantitative analysis conducted by Bever et al. (2016) to combine long-term fish sampling data from the estuary with detailed 3-D hydrodynamic modeling to investigate the relationship between historic fish catch and hydrodynamic complexity. In their analysis, the Fall Midwater Trawl (FMWT) Delta Smelt catch data from 1967 to 2012 were used to rank stations based on their relative historic Delta Smelt catch (Figure 4A). Thirty-five metrics of environmental complexity were developed from the FMWT observations and a set of simulations from the UnTRIM Bay–Delta model. Bever et al. (2016) then evaluated the correlations between historic Delta Smelt catch and 35 quantitative metrics of environmental complexity at each station to determine which variables were most highly correlated to historic Delta Smelt catch. The three metrics found to be most predictive of historic Delta Smelt catch were the percent of the time salinity was less than 6 psu, the maximum depth-averaged current speed, and the Secchi depth at each FMWT station in the vicinity of Suisun Bay.

Using these three quantitative metrics of environmental complexity derived from observed data and 3-D model predictions (Bever et al. 2016), the relative ranking of stations for Delta Smelt catch in Suisun Bay across 4 decades could be predicted. It was also possible to develop a 2-D map of the historic habitat index based on these three metrics (Figure 4B), and to evaluate how these conditions varied during different years (see Bever et al. 2016). This analysis revealed that a key to historic Delta Smelt catch is the overlap of low salinity, low maximum velocity, and low Secchi depth regions. The predictions of the hydrodynamics from the 3-D numerical model were integral in further understanding spatial variability in the fish catch on scales smaller than the estuary-wide salinity gradient. These results also demonstrated that hindcasts from multi-dimensional models can be combined with long-term data sets to explore environmental conditions at different spatial and temporal scales in order to improve the understanding of observed biological data. Although Bever et al. (2016) focused only on the relationship between hydrodynamic complexity and Delta Smelt in the estuary, the methods they developed could be extended to other species and areas of interest in the estuary.

USING MODELS TO UNDERSTAND CIRCULATION AND WATER SUPPLY

Evaluating the Effect of the Emergency Drought Barriers

Bay–Delta SCHISM was introduced in 2014–2015 during an extreme drought, so many CDWR applications of SCHISM have been associated with the effects of the emergency drought barrier at West False River (CDWR, forthcoming). For salinity, SCHISM was first deployed as an expository tool, producing animations to demonstrate the tidal
pumping salinity intrusion mechanism that motivated the West False River barrier design (Figure 5).

SCHISM was recalibrated for extreme low flow in 2014, achieving agreement with data collected in 2013–2015, and was used together with the 1-D DSM2 model for some operational forecasts and retrospectives. The SCHISM model was not used to study expected barrier performance in 2015 because the screening methodology for selecting locations and benefits of the barrier focused on the water cost of compliance with State Water Board Decision D-1641 water quality objectives, an iterative optimal control problem that required thousands of rapid trial evaluations, and DSM2, with a run speed of 80,000 times real time, served this purpose better.

Over 2015, SCHISM evolved into the main tool used to answer velocity and circulation questions

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**Figure 4**  
(A) FMWT (circles) and Bay Study (triangle) stations in Suisun Bay, colored based on relative station ranking based on historic Delta Smelt catch; (B) Two-dimensional maps of the station index for Delta Smelt based on the salinity, velocity, and Secchi depth metrics. Modified from Bever et al. (2016).
associated with the emergency drought barrier and to vet sites for scour monitoring. SCHISM predicted changes to tidal range and flow patterns accurately enough to identify errant flow stations and ultimately was able to describe global circulation changes in salinity—the dominant water supply question.

A related issue concerned circulation changes in Franks Tract. One hypothesis was that the barrier would cause pockets of very long residence time to develop behind the barrier relative to ambient waters and lead to significant algal blooms—which ultimately were not observed. Initial modeling by CDWR with SCHISM indicated that a wide low velocity gyre could form in the eastern portion of Franks Tract, stimulated by amplified oblique inflow from Old River down remnant north–south pathways, and that this recurrent circulation pattern could offset the increase in residence time and induce more mixing. At the same time, it was noted that this was a fragile flow structure that could be suppressed by wind and macrophyte vegetation, and the specifics (speed of wind, prevalence of pondweed) would not be known until later. As a result of the perceived sensitivity of the result, a water quality site in Franks Tract near the Old River inlet was set up so CDWR could monitor it continuously, ultimately confirming relatively low transverse mixing between the eastern remnant channels and Franks Tract. The model, data, and associated discussion in Delta Science Program workshops contributed to an improved reconstruction of Franks Tract dynamics and how to focus future monitoring.

**Using Models to Evaluate the Effects of Sea Level Rise**

A particularly valuable aspect of models is their ability to forecast conditions that have yet to occur. Some of the first applications of 3-D models in the Bay–Delta focused primarily on salt intrusion,
and comprised valuable tools to investigate how sea level rise and levee failures in the Delta could influence water quality (MacWilliams and Gross 2007; MacWilliams and Gross 2010). MacWilliams and Gross (2010) evaluated salinity intrusion over one annual hydrologic cycle under five levels of sea level rise between 15 cm and 140 cm based on Delta outflows and operations as they were during 2002. Their model predicted that these increases in sea level would result in an increase in X2 throughout the year (Figure 6A), with predicted median increases ranging from 0.7 km for the 15-cm sea level rise scenario to more than 7 km for the 140-cm sea level rise scenario. However, the increase in X2 was highest during the transient conditions that occurred during larger outflow events, and the maximum X2 increases predicted for each rate of sea level rise over the 1-year period were between 60% and 120% higher than the predicted median increases (Figure 6B).

Chua and Xu (2014) used SUNDANS to look at the effect of sea level rise on salinity intrusion, finding that at moderate steady-state flows (300 m$^3$ s$^{-1}$), a 1.5-m sea level rise would result in a 10 km increase in X2 relative to present conditions. At higher steady-state flows (2000 m$^3$ s$^{-1}$), the effect was less pronounced, with a similar sea level rise producing only a 4 km increase in X2. This suggests that the transient and steady-state influence of sea level rise on X2 may be different. Also using SUNDANS, Holleman and Stacey (2014) explored how shoreline condition—i.e., levees at present shore locations as opposed to tidal flooding of existing low-elevation regions around the Bay such as the marshes of San Pablo and Suisun bays—affect tidal dynamics. They found that “preserving original shorelines, produces

![Figure 6](image-url)
additional tidal amplification” whereas allowing for the “flooding of adjacent low-lying areas introduces frictional, inter-tidal regions that serve as energy sinks for the incident tidal wave.” When sea level rise is combined with tidal amplification, the predicted increase in $X_2$ is higher than when tides are not amplified (Figure 6A). The overall result is that sea level rise can significantly alter tidal dynamics as well as increasing salinity intrusion. Thus, these modeling studies suggest that any evaluation of large-scale Delta restoration or modification should include sea level rise modeling (see also the NRC 2014 report on Everglades restoration).

**USING MODELS TO UNDERSTAND PHYSICAL HABITAT**

**Using Models to Evaluate Historic Habitat**

Although models are commonly used to forecast conditions that have yet to occur, a recent application of the UnTRIM model by RMA demonstrates the utility of using models to hindcast historic conditions for which very limited observation data are available (RMA 2015; Andrews et al., submitted). Anthropogenic changes to the estuary over the past 2 centuries have altered both the hydrology and the geometry of the estuary (Andrews et al., submitted). To examine how these changes have altered physical habitat in the estuary, 3-D hydrodynamic models were constructed to study the system in its “pre-development” condition (before

**Figure 7** Inundation frequency in the pre-development upper San Francisco Estuary (source: Andrews et al., submitted). Historical observations are given along with the general area where the observation was made. The corresponding modeled values are the median of monthly inundation frequencies for months with maximum net estuary outflow less than 1000 m$^3$s$^{-1}$. Areas which remain dry are shown as beige. Areas which remain wet are shown as gray.
significant modern anthropogenic influence) and in its contemporary condition. The pre-development system model was created to match the pre-development channel configuration in the upper estuary by the San Francisco Estuary Institute (Whipple et al. 2012), and calibrated by varying the marsh plain elevations to match sparse observed data points of tidal characteristics. These tidal characteristics included tidal range in channels and marsh plain inundation depth and frequency (Figure 7), as well as broader metrics, such as the extent of freshwater tidal habitat.

Through comparison of the pre-development and contemporary models, Andrews et al. (submitted) evaluated how changes to both the hydrology and the geometry of the estuary have affected salinity. Their results show that salt intrusion in the pre-development system was found to be slightly more sensitive to outflow and responded faster to changes in outflow than in the contemporary system. Changes in estuary hydrology were responsible for more of the salt intrusion differences between the two systems than were changes in estuary geometry and bathymetry. For example, for the same inflows, their results indicated that the X2 position in the pre-development and contemporary systems would be located within 5 km of each other 90% of the time (RMA 2015). However, when differences in both net outflow and bathymetry between the contemporary and pre-development systems were considered, there were larger differences in both average and seasonal predictions of low salinity zone (LSZ) area and volume. On average, the model predicted a higher variation in LSZ volume, area, and average depths in the pre-development simulation (RMA 2015; Safran et al. 2016). These insights into how changes to the hydrology and geometry of the system have affected physical habitat, provide a valuable resource to guide future restoration efforts in the estuary.

Figure 8  (A) Daily-averaged depth-averaged salinity between Carquinez Strait and the western Delta on a day when X2 is approximately 75 km; (B) percent of the same day that the depth-averaged salinity is within the low-salinity zone (between 0.5 and 6 psu). Source: MacWilliams et al. (2015).
Expanding our Perspective Beyond X2 to a Broader View of the Low Salinity Zone

Modeling has played an important role in broadening our perspective beyond a 1-D view of X2, to a better understanding of the position, area, and volume of the LSZ and how these variables relate to X2 (e.g., Kimmerer et al. 2013; MacWilliams et al. 2015). Detailed maps showing the location of the LSZ, and the tidal excursion of the LSZ over each day (Figure 8) have broadened the discussion beyond X2 to begin to focus on the range of X2 values that produce low salinity habitat in the areas of Honker Bay and Grizzly Bay (DMA 2014b; MacWilliams et al. 2015). This shift from a linear (X2) to a more geographic focus has also allowed us to broaden our focus from thinking only in terms of the LSZ, to better understanding the importance of overlapping regions of low salinity, high turbidity, and low velocity for producing conditions that have historically yielded the most consistent catch of species such as Delta Smelt (Bever et al. 2016).

Recent work by MacWilliams et al. (2016), demonstrates that multi-decadal 3-D simulations of the Bay–Delta are now computationally feasible. They used salinity predictions from a 35-year simulation to develop maps of salinity distribution over seven periods for six fish species, and combined the salinity distribution maps with historic fish-sampling data to allow for visualization of fish abundance and distribution. These maps can be used to explore how different species respond to annual differences in salinity distributions in the estuary and expand the understanding of the relationships among salinity and fish abundance, distribution, and population resiliency. Thus, models have played an important role in increasing our understanding of the mechanisms and processes that may be responsible for the correlations between X2 and fish abundance, and are suitable for future applications that inform and support a broader system-wide approach to understanding Delta ecology.

The Effect of Large-Scale Restoration in the Delta on Turbidity and Sediment Transport

The Prospect Island Tidal Habitat Restoration Project is a joint effort by the CDWR and the California Department of Fish and Wildlife (CDFW) to restore the 1,600 acres in Prospect Island to freshwater tidal wetland and open water (subtidal) habitats to benefit native fish and improve aquatic ecosystem functions (CDWR 2016b). The UnTRIM Bay–Delta model was applied with the SWAN wave model and the SediMorph morphologic model to evaluate potential effects of the Prospect Island Restoration Project on sediment transport and turbidity in the Sacramento DWSC and the Cache Slough complex (DMA 2014a). This approach allowed for a direct method to evaluate changes in sediment dynamics in the project vicinity, because sediment transport, deposition and re-suspension, the effect of wind waves, and the potential for deposition within Prospect Island to influence regional sediment dynamics were all explicitly simulated.

Baseline conditions and a range of restoration alternatives were compared to assess how different restoration design configurations affected hydrodynamics and turbidity during both summer conditions and during a “first flush” with high flow and high turbidity from the Sacramento River. These comparisons enabled specific alternatives to be identified that resulted in the smallest effects on the turbidity regime in the DWSC and the Cache Slough Complex where elevated turbidity provides an essential component of Delta Smelt habitat. In addition, model results also identified several important influences of the restoration on the turbidity regime that were not anticipated in advance. For example, under Baseline conditions during low outflow periods typical of late-summer and early-fall, turbidity tends to be relatively low in Miner Slough because of the downstream transport of low turbidity water from the Sacramento River. However, when the levee between Prospect Island and Miner Slough is breached, the model predicted that the tidal prism of Miner Slough significantly increases because of the filling and draining of Prospect Island each tidal cycle. This increased tidal prism draws turbidity from Cache Slough up Miner Slough on flood tide and results in an increase in turbidity from the mouth of Miner Slough to the breach location (Figure 9). This higher-turbidity corridor in lower Miner Slough, also leads to a small reduction in turbidity in both Cache Slough and the Sacramento DWSC. Over time, these turbidity dynamics are expected to evolve as
deposition within Prospect Island and the evolution of shallow tidal wetlands reduces the tidal prism.

This application highlights that tidal restoration in the Delta can have regional influences on turbidity, and may also affect downstream sediment supply because of deposition that occurs within breached islands. A recent independent scientific review has also raised concerns that water diversions in the north Delta could “exacerbate the downstream sediment starvation that is already occurring” (Simenstad et al. 2016). Both tidal restoration in the Delta on a large scale, and major changes to Delta operations could have a large influence on the sediment transport regime in the Delta. These changes and how they are implemented could influence the rate of sediment accumulation within restored Delta islands, have regional effects on Delta turbidity, and affect the downstream sediment supply to the bay. Modeling of these changes using available 3-D hydrodynamic, wind wave, and sediment transport models can be used to evaluate future changes to the Delta system, and to develop alternatives that will achieve the greatest possible benefits for water supply and habitat restoration while minimizing adverse effects on the sediment supply to downstream areas in the system.

Figure 9  Predicted turbidity in the vicinity of Prospect Island under Baseline conditions and under restoration Alternative A (Alt A), with a single breach (B) between Miner Slough and Prospect Island. Source: Delta Modeling Associates (2014a).

One of the most powerful applications of multi-dimensional modeling is to use models to examine links between organism behavior and flow (see Cowen et al. 2007). One classic example of these links is that of “selective tidal stream transport” (STST): the net horizontal transport of weakly swimming organisms by exploitation of vertically sheared tidal flows. Effectively, by changing vertical position at appropriate tidal phases, organisms, such as copepods or larval fish, can move large distances each tidal cycle or maintain a geographic position despite water movement. For example, using 3-D modeling, Simons et al. (2006) showed how various zooplankton could maintain particular positions in the LSZ of the St. Lawrence Estuary (see also North et al. 2008). Tracking individual “particles” with behavior moved by flow has also been used to assess connectivity of different regions in space, e.g., different coral reefs in the Caribbean (Cowen et al. 2000) and regions of the California coast (Simons et al. 2013).

More generally, beyond including the effects of hydrodynamic transport on populations made up of particles (individual organisms) moving through the model domain, so called individually based models (IBMs) can also incorporate physiology, growth, reproduction, etc. (e.g., Rose et al. 2013). However, there is the substantial challenge of knowing the behavior to be used in the model. For example, depending on which of the bioenergetically possible swimming behaviors Mysid shrimp used in the St. Lawrence Estuary, Simons et al. (2006) found that there could be either upstream migration from the LSZ to Montreal, maintenance of position in the LSZ, or downstream migration into the Atlantic Ocean. Nonetheless, as suggested by Banas et al. (2009) in the context of modeling nutrient-phytoplankton dynamics in the Washington Shelf–Salish Sea region, this form of coupled modeling may also offer researchers the ability to invert observed data on the distribution of organisms to infer behavior.
As implemented in 3-D models discussed below, particle tracking of organisms has three components: (1) transport in three dimensions by computed (or assumed) currents; (2) mixing by turbulence via a random walk formulation, something that depends on the turbulence closure used; and (3) swimming/sinking/rising prescribed by the modeler. One particular advantage of particle tracking is that it is essentially error-free for advection, in contrast with numerical representations of scalar advection that always involve artificial diffusion and other undesirable aspects (Gross et al. 1999). We note that although particle tracking is conceptually simple, its implementation can be challenging (e.g., see Edwards et al. 2000), requiring care with how particle movements are treated near boundaries (Gross et al. 2010) and with ensuring that the random number generator used in the random walk step is truly random (Hunter et al. 1993).

Using Particle-Tracking Models to Understand Larval Retention

The work of Kimmerer et al. (2014) demonstrates the value of coupled observations and modeling. In several field studies carried out in Suisun Bay, Kimmerer and colleagues (Kimmerer et al. 1998, 2002) had attempted to find evidence of STST in larval fish and zooplankton distributions. Kimmerer et al. (1998) found that 2-D Eulerian calculations based on observations showed that the sampled organisms should have been swept out to sea by the observed net, subtidal flow, whereas in reality, they were able to remain in Suisun Bay.

Using hydrodynamic model output from the UnTRIM Bay–Delta model (MacWilliams et al. 2015) with the particle-tracking code FISH-PTM (Gross et al. 2010), Kimmerer et al. (2014) explored the effects of swimming behavior on the fate of particles that were initially located in Suisun Bay and the western Delta. In their study, they were able to examine how particle fate varied with both hydrology (wet or dry years) and behavior. Echoing the results of the study of the St. Lawrence Estuary by Simons et al. (2006), both pure downwards swimming (i.e., sinking) and tidally varying upwards and downwards swimming increased retention of particles in Suisun Bay at all flows (Figure 10).

Notably one of the behavioral patterns—upwards swimming at 0.75 mm s⁻¹ and downwards swimming at 0.75 mm s⁻¹ (their “Tidal 0.5” behavior)—produced distributions of particles in different salinity classes that matched with reasonable accuracy observed distributions of the copepod *Eurytemora affinis*.

Beyond addressing the question posed by Kimmerer et al. (1998) of to what extent tidal correlations of organism position in the water column and flow are important to the retention of organisms in the estuary, additional features of the effects of combining swimming with complex 3-D transport emerged. Notably, Kimmerer et al. (2014) showed that much of the retention at low Delta outflows happened in the sloughs of Suisun Marsh. Finally, beyond showing that possible behaviors enhanced retention, the Kimmerer et al. (2014) study also showed clearly how testing hypotheses with coupled 3-D physics–behavior models often leads to new questions. For example, as Kimmerer et al. (2014) ask at the end of their paper: What cues trigger a given swimming behavior? What behaviors might zooplankton use to remain in the Bay–Delta when flows are high? In the long run, this interdisciplinary iteration among modeling, observations, and the development of new conceptual models may be key to understanding how and why various restoration and management actions succeed or fail.

Using Particle-Tracking Models to Estimate Fish Entrainment

Gross et al. (2010) developed the FISH-PTM to represent particle-transport processes. The model can be used to simulate both passive particles and particles with swimming behaviors. The FISH-PTM runs offline, using hydrodynamic model results saved from a 3-D model. This makes it possible to simulate a large number of particle-tracking scenarios and to evaluate different particle behaviors without re-running the hydrodynamic model.

Using hydrodynamic results from the UnTRIM Bay–Delta model (MacWilliams et al. 2015) and available observations, Gross et al. (2010) applied the FISH-PTM to estimate the hatching distribution of Delta Smelt in 1999 and 2007. The hatching distributions predicted for the 1999 conditions indicated hatching in areas that are consistent with current biological understanding based on recently conducted larval surveys. Gross et al. (2010) evaluated four
scenarios consisting of a passive scenario and three different upward swimming scenarios. The hatching distributions estimated for simulations with different hypothesized vertical migration behaviors were similar to the hatching distributions estimated for the passive particle-tracking scenario. The FISH-PTM simulation results indicated that, regardless of behavior, the fate of most of the larval and juvenile Delta Smelt was loss from natural mortality. The estimated entrainment losses for the different scenarios correspond to 2% to 3% of the total larval and juvenile fish that were estimated to hatch during 1999. The behaviors evaluated did not significantly influence the percent of the overall population of larval and juvenile Delta Smelt lost to export pumping. In all four scenarios, fewer than 20% of the fish hatched during 1999 were estimated to survive to the end of the simulation on July 21, 1999, corresponding to the end of the 20-mm survey period. Based on this, Gross et al. (2010) concluded that the vertical migration behaviors explored so far have limited influence on Delta Smelt distribution and fate.

The estimates of Delta Smelt distribution and, in particular, hatching distribution, are extremely relevant to ongoing policy decisions. Any project that modifies flow pathways and mixing in the Delta is likely to decrease entrainment of fish from some regions and increase entrainment of fish from other regions. Therefore, to confidently estimate the effects of such a project, it is critical to estimate the distribution of Delta Smelt and any other relevant fish species. Modeling tools and approaches such as those used by Gross et al. (2010) that coupled 3-D hydrodynamic modeling results with a properly validated PTM model—particularly if applied in a probabilistic framework—will be useful supplements to ongoing observational programs in estimating the distribution and entrainment of Delta Smelt and other
species for current conditions and different Delta operations scenarios.

**Linking Hydrodynamics with Nutrients and Fish**

As part of the ongoing SESAME project (SFSU Romberg Tiburon Center, NASA, NOAA–NMFS), the biogeochemical model Carbon, Silicate, and Nitrogen Ecosystem (CoSINE) (Chai et al. 2002, 2003; Xiu and Chai 2014) was coupled to SCHISM and the Bay–Delta application, and tailored to handle domain-specific questions. The SESAME model is a full life-cycle bioenergetic model of salmon that covers riverine, estuarine, and near-coast waters. The role of SCHISM is to model flow, food, and temperature in the estuary portion of the domain, which is applied in turn to individual salmon to evaluate the costs of migration through and rearing within the Bay–Delta. The model calibration is currently constrained to the bay and the lower Sacramento River; the application is in the process of extension to the full Bay–Delta, and, in the future, NOAA expects to combine SESAME with SCHISM's particle tracking to include the influence of detailed local velocity on migration and entrainment.

The SESAME approach to salmon modeling is individually based and uses particle modeling, but the emphasis of the ecosystem approach is on migration through a breadth of environments and life stages. Chinook Salmon spawn in rivers, and after a few months rearing in freshwater, juveniles migrate to the coastal ocean where food supplies and growth prospects are typically greater than in their natal river (Gross et al. 1988). The individual-based sub-model for juvenile salmon in SESAME includes a bioenergetics model to determine growth, and an area-restricted search algorithm to simulate foraging behavior. Domain-specific but coupled hydrodynamic and biogeochemistry models are used for the near coast (Regional Ocean Modeling System, ROMS), estuary (SCHISM) and upper Sacramento (River Assessment for Forecasting Temperature, RAFT).

**DISCUSSION AND CONCLUSIONS**

It is clear that multi-dimensional modeling of hydrodynamics in the Bay–Delta has advanced significantly in the last decade, to the point that it can now be used to accurately predict of flows both locally (e.g., in CCF) or at the scale of the whole Delta. The application of 3-D models of this type is the current state-of-the-art for modeling the Delta.

Multi-dimensional models have provided significant insights into some of the fundamental biological relationships that have shaped our thinking about the system, including exploring the relationships among X2, flow, fish abundance, and the LSZ. Through the coupling of multi-dimensional models with wind-wave and sediment-transport models, it has been possible to understand how large-scale changes to the system are likely to affect sediment dynamics, and to assess the potential effects on species that rely on turbidity for habitat. The coupling of multi-dimensional hydrodynamic models with particle-tracking models has led to advances in our thinking of the retention of food organisms in the estuary, the effect of south Delta exports on larval entrainment, and the pathways and behaviors of salmonids that travel through the Delta.

At present, we have succeeded in building models that are well-suited to engineering use. The models described above have been applied with success to answering specific questions about the Delta. These examples demonstrate that 3-D models in particular are now sufficiently well-developed that they should be the tool of choice for evaluating focused questions, particularly those involving purely hydrodynamic processes, such as the effects of sea level rise or large-scale restoration on salinity or sediment transport in the Delta. In more detailed studies, such as those involving secondary circulation or requiring resolution of vertical structure, resolved 3-D models are the only choice. In cases where 3-D models are harder to scale to longer times and broader spatial scales, the results of 3-D models may be harvested to inform coarser or simpler models (e.g., CDWR 2013, Section D8).

Lower-dimensional 1-D models will continue to be used in a variety of settings. Their simplicity and speed are valued in widespread deployment, and their empirical performance can be understood and be competitive over decades or centuries. Speed is important in algorithmic settings such as optimization and Bayesian analysis. Additionally, 3-D models may rely on forcing that is hard to predict.
or flow features that are hard to simulate precisely, so that even where a complex process is known to be physically relevant it may not be represented well in a multi-dimensional model without sufficient resolution and data. For example, the importance of good bathymetry data for any application cannot be overstated. Even the best extant hydrodynamic model will not produce accurate flows if the bathymetric data is not accurate. In selecting a model for planning studies, dimensionality (Table 1), geographic and time scales, and the processes being simulated (Figure 1) must all be considered in any assessment of a model’s predictive skill.

Through interdisciplinary work with biologists and ecologists, multi-dimensional models have enabled us to address key questions about the role of physical processes in shaping the ecology of the Bay–Delta (e.g., Lucas et al. 2002; Kimmerer et al. 2014; MacWilliams et al. 2016; Bever et al. 2016). In this realm, uncertainty in parameterizations of biological processes may be a much more significant limitation than uncertainty in computing transport and mixing. For example, 3-D models of the cyanobacterium Microcystis have been built (e.g., for the Swan River Estuary in western Australia [Robson and Hamilton 2004]), but they are seriously hampered by lack of knowledge about what factors influence Microcystis colony size, a critical determinant of the likelihood of bloom formation (2006 email conversation between S. Monismith and D. Hamilton, unreferenced, see “Notes”). In any modeling exercise it is important to recognize this fundamental challenge of understanding the critical uncertainties. At this point in time, it may be the case that improving biological and biogeochemical models is more important for improving Delta management than for improving multi-dimensional physical models. As one of us (SGM) was told in 1987 by the then-head of the IEP when he (SGM) began to get engaged in Bay–Delta science, “What is the use of a Rolls Royce understanding of the physics if all we have is a Yugo understanding of the biology?”

Overall, what is most important is that the entire modeling exercise—i.e., choice of type of model, spatial discretization, the periods of simulation—be chosen properly to match the purpose of the model and the time-frame in which model results are required. At one end of the spectrum might be long-time-scale results that are required at relatively short notice. One very successful example of this type of modeling was the time series modeling (essentially a zero-dimensional model) done by Denton, Gartrell, and Sullivan at the Contra Costa Water District to examine the water supply implications of proposed X2 regulations then under development by the EPA. At the other end of the spectrum, one could imagine carrying out large-eddy simulation (LES) (c.f., Sotiropoulos 2015) to model the very detailed structure of flows at the proposed north Delta intakes. In this case, individual-based models of fish behavior (Goodwin et al. 2014) could be combined with high-resolution LES results to examine how interactions of flow and behavior would affect entrainment of out-migrating juvenile salmon. The fact that only relatively short periods of time (a few days) would need to be modeled makes this computationally feasible. Intermediate cases might include using 3-D modeling to predict in advance the effect on Delta salinities of the drought barrier installed in False River in 2015. In this case, the modeling that was required had to be reasonably accurate and delivered in a relatively short time, but the simulation period was a few months rather than decades (or days), again necessitating a compromise between computation scale and execution time.

The Future of Multi-Dimensional Modeling in the Delta

We anticipate that we will see continued improvements in model resolution and computational speed, at least in the application of multi-dimensional models using desktop computing and small clusters. In this regard, the needs of research intended to advance scientific understanding of the system may differ from what is needed in the near-term to address particular questions of system management (e.g., temporary barriers). In the former case, modeling should push the boundaries of what is possible; however, such studies are often undertaken with supporting field data collection and require either hydrodynamics, coupled physics, or forcing that may not extrapolate easily to planning scenarios. In the latter case, what is most important is to focus on what would provide the largest improvement in the fidelity of models used to make management and
engineering decisions, and on identifying the types of multi-dimensional results that are accurate and robust for evaluating hypothetical scenarios. We believe both directions are important. Indeed, most of the examples described above would have been considered cutting-edge for environmental computations only a few years ago, whereas today, they should be considered to reflect appropriate engineering practice.

Several particular problems can and are being addressed now and in the near future with existing models:

1. **3-D structure and temporal variation of primary and secondary production as affected by in-Delta flows, gate and barrier operations, Delta outflow, and nutrient inputs.** The challenges here are to represent both the large effects of benthic grazing and zooplankton population dynamics adequately. Another possible direction for research would be to examine the competition between cyanobacteria like *Microcystis* that can form blooms when mixing is weak (Huisman et al. 2004) and diatoms for which strong mixing can offset sinking.

2. **Modeling temperature variability in time and in 3-D space.** Climate change will result in higher temperatures that will more frequently exceed lethal limits for Delta Smelt (Wagner et al. 2011). However, this conclusion is based on observations made largely with near-surface temperature sensors. Observations of temperature stratification in the DWSC reported in Monismith et al. (2008) suggest that near-bottom temperatures can be several degrees cooler than near surface temperatures during the day. Thus, deeper waters may offer Delta Smelt thermal refugia in the face of increasing surface temperatures. At present, our ability to model temperatures in general, and stratification in particular, is limited by lack of meteorological data for the Delta (2015 in-person conversation between S. Monismith and L. Herdman, unreferenced, see “Notes”) and by limits to the predictability of turbidity and hence shortwave attenuation with depth. The development and calibration of a temperature model of the Delta is in progress as part of the CASCaDE project (Knowles and Lucas 2015), and future results from this work may provide further insights into how climate change will affect Delta water temperatures.

3. **Modeling wind and sea level effects on Delta outflow and hence salinities in northern San Francisco Bay and the western Delta.** Low-frequency variability in sea level in the Delta associated with both the coastal ocean sea level variability and with winds on Suisun Bay may explain flows associated with filling and emptying of the Delta observed by Oltermann (1998) that can be substantially larger at times than the corresponding outflow estimates from Dayflow (Monismith 2016). Although the sub-tidal hydrodynamic forcing that causes this variability is a complicated mixture of rectification of tidally varying bottom stresses and momentum advection (Smith and Cheng 1987; Monsen 2000), it can be directly addressed by 3-D modeling. We note that this may be challenging given that subtidal variability represents a small signal in the presence of large (tidal) variance (c.f. Sankaranarayanan and Fringer 2013), and there can be surprising subtleties with respect to averaging the effects of tidal variations in stratification on flow dynamics (Stacey et al 2010).

4. **Applying multi-dimensional models to understand the mechanisms behind the flow abundance relationships described by Jassby et al. (1995).** Initial applications by Kimmerer et al. (2009, 2013) explored hypotheses relating to the area and volume of the LSZ. However, more recent analyses by Bever et al. (2016) and MacWilliams et al. (2016) demonstrate the role that multi-dimensional models can play in understanding more broadly the combination of habitat characteristics that have led to historic abundance and distribution. These methods can be applied to other species and other portions of the bay and Delta to increase our understanding of habitat suitability for different fishes and life cycle phases.

5. **Applying coupled 3-D hydrodynamic, wave, and sediment models to investigate large-scale changes to the sediment dynamics of the system.** Observed long-term trends such as the clearing of the Bay (Schoellhamer 2011), system-wide changes to the
Delta resulting from large-scale restoration such as EcoRestore (CNRA 2016a), or changes to water conveyance such as WaterFix (CNRA 2016b) will affect sediment dynamics in the bay and Delta and subsequently influence species that are affected by turbidity. The example shown here for the Prospect Island Tidal Habitat Restoration Project (DMA 2014a) demonstrates that the tools to address and plan for these issues are already available.

6. **Develop coupled 3-D models of lower trophic levels.** This work has been started as part of the SCHISM work described above, and recommendations that prioritize species and processes have been developed as part of work panels and modeling white papers organized by the San Francisco Estuary Institute and Central Valley Regional Water Resources Control Board (Trowbridge et al. 2016).

7. **Support interdisciplinary work on coupled “fish and flows” modeling.** This has been accomplished in several recent studies described above (e.g., Bever et al. 2016; MacWilliams et al. 2016). Other examples where such collaboration has been formalized include the SESAME and NASA-HICO projects funded by NASA, which includes participants with local expertise in biogeochemistry but also borrows extensively from out-of-state experience in remote sensing and light transfer.

8. **Develop a 3-D operational Delta model that assimilates data in real time and issues and updates forecasts for flows, temperatures, and salinities.** We note that this is currently done for the California Current System by the set of West Coast Integrated Observing Systems (CeNCOOS [Central and Northern California], SCOOS [Southern California], etc.) and for San Francisco Bay (but not extending through the Delta) as part of the the San Francisco Bay Operational Forecast System (NOAA, 2016).

**Final Comments**

Although this paper focuses on modeling, it is clear that model utility is often limited by the availability of suitable data for boundary conditions (e.g., salinities and temperatures at open boundaries), forcing (e.g., winds), as well as observations to be used for calibration and verification of model output. Conversely, observations are necessarily limited in spatial detail and extent—something at which models excel. Thus, we believe that models and observations (at least those that go beyond monitoring) are most usefully done together, perhaps using models first to design the observational program and then using the resulting observations to confirm model results and to refine the model. At the same time, we acknowledge that this leads to a dichotomy of approach, one formulated to the cutting edge and guided by ample forcing and supporting data, and the other asked to give robust results over hypothetical planning scenarios fed by estimates.

Although the use of multi-dimensional models to advance system understanding is clearly valuable, it must be acknowledged that most (but not all) of the hydrodynamic modeling done to date has been intended to address very specific questions, i.e., as engineering studies of particular issues such as the effects of sea level rise, levee breaks, or temporary barriers on Delta salinities. Thus, we feel that there is much unexploited potential for using multi-dimensional models in interdisciplinary research like that described in Kimmerer et al. (2009) and Bever et al. (2016), which advance understanding of complex coupled physical–biological dynamics. As noted by Blumberg et al. (2010):

“When not used to make specific engineering-type predictions, models can also be used to explore hypothesized linkages of forcing (e.g., flow) and responses (e.g., fish behavior, population dynamics, or water quality), suggesting relationships that can be explored through further analysis of data or by design of new data collection programs. Finally, models also can serve to link researchers from different disciplines, i.e. to provide a forum for interdisciplinary collaboration between [sic] fisheries biologists, social scientists and engineers.”

With a view towards advancing multi-dimensional modeling of the Bay–Delta system in ways that will improve its utility to address regulation, management, and policy concerns, we echo the importance of the
recommendations made to the IEP as a result of a 2009 review of Bay–Delta modeling (Blumberg et al. 2010). Few will disagree that data and model integration and interdisciplinary collaboration on modeling and field research are essential steps for the future of Delta management. Yet, we can predict that it will continue to be difficult for disparate agencies, universities, and consultants to sustain collaborations beyond short funding cycles. Despite these challenges, significant advances have been made over the past decade in the development of application of multi-dimensional models. The numerous examples presented here represent the current state-of-the-science in Bay–Delta modeling, and indicate that we are continuing to make significant progress in using multi-dimensional models to interpret observational data, evaluate changes that affect circulation and water supply, advance our understanding of physical habitat, and test our hypotheses about fish behavior and movement.

In conclusion, modeling is an art that balances process resolution, computational speed, and accuracy. It is parsimonious with the data available to validate results and inferences. Modeling requires system knowledge and conceptual models to assess the time and space dimensions of the phenomenon or the objective being studied and in turn hone questions and approaches for modeling analysis. Multi-dimensional modeling provides the tools to move beyond the reductionist approach of trying to simplify the complexity of an entire system to a single variable such as X2, and instead focus on whole-system interactions. The challenge for the multi-dimensional modeler then becomes to take the enormous amount of information generated by the model and present it in a way that can be used to increase understanding of the system, without averaging out all of the important details. In this regard, the means of distilling the large amount of information into a meaningful result becomes one of the hallmarks of modeling success.

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NOTES


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Recent Advances in Understanding Flow Dynamics and Transport of Water-Quality Constituents in the Sacramento–San Joaquin River Delta

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ABSTRACT

This paper, part of the collection of research comprising the State of Bay–Delta Science 2016, describes advances during the past decade in understanding flow dynamics and how water-quality constituents move within California’s Sacramento–San Joaquin River Delta (Delta). Water-quality constituents include salinity, heat, oxygen, nutrients, contaminants, organic particles, and inorganic particles. These constituents are affected by water diversions and other human manipulations of flow, and they greatly affect the quantity and quality of benthic, pelagic, and intertidal habitat in the Delta. The Pacific Ocean, the Central Valley watershed, human intervention, the atmosphere, and internal biogeochemical processes are all drivers of flow and transport in the Delta. These drivers provide a conceptual framework for presenting recent findings.

The tremendous expansion of acoustic and optical instruments deployed in the Delta over the past decade has greatly improved our understanding of how tidal variability affects flow and transport. Sediment is increasingly viewed as a diminishing resource needed to sustain pelagic habitat and tidal marsh, especially as sea level rises. Connections among the watershed, Delta, and San Francisco Bay that have been quantified recently highlight that a landscape view of this system is needed, rather than consideration of each region in isolation. We discuss interactions of multiple drivers and information gaps.

KEY WORDS

Hydrodynamics, transport, water quality, Sacramento–San Joaquin River Delta, ocean, watershed, anthropogenic, fluvial forcing

INTRODUCTION

Movement of water and constituents carried by water within the Sacramento–San Joaquin River Delta (Delta) depend on forcing from the Pacific Ocean, the Central Valley watershed, human intervention, the atmosphere, and internal biogeochemical processes. Semidiurnal and diurnal tides from the ocean slosh water back and forth several kilometers daily through the Delta’s complex network of channels. The ocean is a source of salinity, the spatial distribution of which can affect flow dynamics and transport. At a longer time-scale, oceanic sea
level determines sea level rise (SLR) in the Delta. The Central Valley watershed provides freshwater via river flow episodically during California’s wet season, and total runoff varies dramatically inter-annually. The watershed is a source of constituents carried by water including nutrients, contaminants, sediment, and organic particles. Water diversions and associated manipulations of inflow and flow paths alter flow dynamics and transport. Wind can alter Delta water levels and generate wind–waves, which can lift material resting on the bottom into the water column. Biogeochemical processes create sources and sinks of constituents within the Delta. Water and its constituents that exit the Delta in the seaward direction affect San Francisco Bay (the bay).

Technological advances in instrumentation in the past 2 to 3 decades have enabled rapid and continuous measurements of water flow and constituent concentrations. Mike Simpson and Rick Oltmann of the U.S. Geological Survey (USGS) were the first to use acoustic current meters to measure water flow in the Delta, doing so from moving boats (Simpson and Oltmann 1993) and continuously from a fixed station (Oltmann 1995; Ruhl and Simpson 2005). Today, an extensive flow station network in the Delta utilizes acoustic instrumentation (Bura et al. 2016). Optical sensors invented in the 1980s were first deployed in the Delta in the late 1990s to continuously measure suspended-sediment concentration (Wright and Schoellhamer 2005). Optical sensors that continuously measure carbon and nutrients were first deployed in the Delta in the 2000s (Bergamaschi et al. 2012; Downing et al. 2009; Pellerin et al. 2013). When these instruments are deployed with sufficient density to enable several channels or locations to be observed simultaneously, continuous data streams that include all tidal fluctuations can be collected, and these data have led to improved understanding of flow and constituent transport in the Delta.

This paper, part of the State of Bay–Delta Science 2016, summarizes advances in the last 10 years in flow dynamics and how water-quality constituents move within the Delta. Water-quality constituents include salinity, heat, oxygen, nutrients, contaminants, organic particles, and inorganic particles (i.e., sediment). These constituents are affected by water diversions and other human manipulations of flow, and these constituents themselves greatly affect the quantity and quality of benthic, pelagic, and inter-tidal habitat in the Delta. The geographic scope of this paper is the Delta, how the watershed affects the Delta, and how the Delta interacts with the bay and ocean. Though we attempt to cover the breadth of our topic by using a conceptual framework of flow and transport in the Delta, we do not intend to present an exhaustive list of recent publications.

This paper is organized by the primary drivers of flow and transport in the Delta: ocean, fluvial, anthropogenic, atmospheric, and biogeochemical. Our intent is to present new discoveries rather than a conceptual model of flow and transport. Conceptual models available elsewhere include flow (Monsen et al. 2007; Sridharan 2015) and sediment transport (Schoellhamer et al. 2012). Though we do not present a conceptual model, a conceptual framework for the drivers that determine flow and transport within and through the Delta is presented in Figure 1. We also discuss interactions of multiple drivers and information gaps.

**Figure 1** Conceptual framework for drivers of flow and transport in the Delta. The black arrows represent flow and transport within and through the Delta. Anthropogenic drivers depicted in gray are dams, barriers in the delta, and water exports. Atmospheric drivers depicted in yellow are storms, solar radiation, and wind. Other drivers depicted are biogeochemical processes in green, fluvial inputs in brown, and oceanic exchange in blue.
Other papers that comprise the State of Bay–Delta Science 2016 cover the geographic setting and some specific flow and transport topics in detail, and, thus we minimally include those topics. The geographic setting of the Delta is presented elsewhere in this issue (Healey et al. 2016). Numerical modeling of hydrodynamics and transport in the Delta has greatly advanced in the past 10 years, and is a key tool for testing hypotheses and planning management actions. The numerical modeling paper (MacWilliams et al. 2016) discusses these advances in detail. Movement of fish in the Delta is affected by water flows and is a key management concern. Fish movement is presented elsewhere in this volume (Moyle et al. 2016; Perry et al. 2016).

FLOW AND TRANSPORT DRIVERS

Oceanic Forcing

The Pacific Ocean, via the bay, affects the Delta by propagation of tides, salinity intrusion, and sea level.

Tides

The Bay–Delta is a classic example of a coastal plain estuary in which terrestrial freshwater mixes with salt water entering the estuary from the ocean. The primary agent for this mixing is energetic tidal motions (Fischer et al. 1979) forced by propagation of ocean tides into the Bay–Delta (Cheng et al. 1993). At the western end of the Delta, tidal currents are generally 10 to 100 times larger than tidally averaged currents (see Figure 2; Oltmann 1998). Accordingly, in much of the Delta, tides generally cause the instantaneous currents experienced by organisms, sediment erosion, and deposition (Brennan et al. 2002), and vertical turbulent mixing important for example, to phytoplankton dynamics (Jones et al. 2009).

The flow leaving the Delta past Mallard Island and entering the bay is central to regulations of flow and water quality. Direct measurement of Delta outflow has proven to be difficult, so two methods have been used to estimate outflow. The first is the hydrologic balance embodied in Dayflow1 and the second is to sum the flows measured by four key USGS flow stations (Oltman 1998). Figure 3 compares two flows for water years 2008–2014. Overall, the comparison is quite good, with a slope near 1, $r^2 = 0.92$, and an RMS error of 6.4 Kcfs. On the other hand, if one examines only low flows, here defined as Dayflow $\leq 10$ Kcfs, then there is almost no relation between the two flows ($r^2 = 0.10$; RMS error = 5.1 Kcfs). Thus, at times when the need is most critical to know what

1. See http://www.water.ca.gov/dayflow/documentation/dayflowDoc.cfm#Introduction

![Figure 2](image_url)  
Figure 2  Tidal (grey) and tidally-averaged (red) flows at Mallard Island inferred from USGS flow stations at Rio Vista, Jersey Point, Threemile Slough, and Dutch Slough
outflow is, so as to properly manage project (and gate) operations, the uncertainty is very large.

Tides also provide much of the frictional resistance to the mean tidally averaged flow in the Delta. It is this frictional resistance that ultimately determines patterns of mean flow. For example, even though total freshwater flow toward the pumps is set by pumping rates, the spatial distribution of those flows is controlled mainly by the resistance of the different routes by which freshwater can reach the pumps. Fong et al. (2009) show one way in which this behavior might affect Delta-scale transport: They found that because of bedforms, flows through Threemile Slough experience larger drag for flows from the San Joaquin to the Sacramento than vice versa. This should translate into a stronger tidally averaged flow from the Sacramento to the San Joaquin than would occur were the drag symmetric for flow direction. Consequently, the effect of asymmetric friction should be to shift a larger fraction of the overall outflow through Threemile Slough rather than through more northern channels such as Georgiana Slough.

Parameters that quantify a physical, chemical, or biological constituent of the aquatic environment may be misrepresented by low frequency data that fail to observe tidal variations (Lucas et al. 2006). To better understand biogeochemical processes, newly developed optical instruments of various types have been deployed in the Delta and upstream. The new instrumentation can make unattended measurements in situ fairly frequently (e.g., every 15 minutes). This capability is particularly important in a tidal setting where tidal advection and dispersion continuously affect constituent concentrations. For example, Ganju et al. (2005) and Downing et al. (2009) used data from Browns Island to assess the minimum sampling frequency necessary to accurately calculate advective

Figure 3  Delta outflow derived from U.S. Geological Survey (USGS) measurements at four stations (points) as a function of California Department of Water Resources (CDWR) Dayflow estimates of Delta outflow for water years 2008–2014 in thousands of cubic feet per second (Kcfs). The top panel (A) is for all flows and the bottom panel (B) is for Dayflow values less than 10 Kcfs. Least square linear regression lines are shown in red. For all Dayflow values $r^2 = 0.92$ and for Dayflow values less than 10 Kcfs $r^2 = 0.10$. USGS and CDWR data from http://cdec.water.ca.gov/.
and dispersive fluxes in a tidal wetland. That frequency was on the order of three per hour.

The shape of the tidal wave changes as it propagates upstream into the Delta, which creates tidal asymmetry and retains sediment in some parts of the Delta. Morgan–King and Schoellhamer (2013) found an estuarine turbidity maximum in the backwater Cache Slough complex created by tidal asymmetry (peak flood tide currents are greater than peak ebb tide currents), a limited tidal excursion, and wind–wave re-suspension. During the study, there was a net export of sediment, though sediment accumulates within the region from landward tidal transport during the dry season. Sediment is continually re-suspended by both wind–waves and flood tide currents. The suspended-sediment mass oscillates within the region until winter freshwater flow pulses flush it seaward. The hydrodynamic characteristics within the backwater region—such as low freshwater flow during the dry season, flood tide dominance, and a limited tidal excursion—favor sediment retention. This sediment retention is reflected in the relatively fine bed sediment found in the Cache Slough area in comparison to other regions of the Delta (Marineau and Wright, forthcoming). Relatively high turbidity makes the Cache Slough complex favorable habitat for Delta Smelt (Nobriga et al. 2005). These isolated backwater regions used to be common in the Delta (Whipple et al. 2012) but are now rare because channels in much of the Delta are now interconnected to convey water to pumps or to convey floodwaters to the ocean. Restoration of isolated dead-end channels may help ecosystem restoration.

A concept that has evolved during the last decade is that transport in the Delta is driven more by tidal flow than tidally averaged flow. A longstanding conceptual model of transport in the Delta neglects tidal motions and dispersion; i.e., transport of heat, pollutants and non-motile organisms is determined entirely by advection by subtidal flows. For example, according to this model, mean flows along the Old and Middle River corridor act (among other things) to transport small Delta Smelt to the export pumps, such that restrictions on Old and Middle River flows can be used to limit entrainment (Kimmerer 2008). The basis for this assumption is that dispersion in the tidal channels is likely to be relatively weak (Fischer et al. 1979; Ho et al. 2008), and that the effects of tidal motions average out, leaving net motions as the only mechanism for net advection. On the other hand, studies of dispersion in the Delta using numerical models show that dispersion associated with tidal flows through the many junctions of the Delta can be substantial (personal communication between E. List and SGM, un referenced, see “Notes”; Monsen 2000; Sridharan 2015). The reason for this is that splitting flows at junctions drives what Ridderinkhof and Zimmerman (1992) describe as “chaotic mixing.” Observational evidence for this behavior was provided by Monismith et al. (2009) who suggested that the large heat fluxes out of the San Joaquin system required to close the overall heat balance could be accounted for with remarkably large dispersion coefficients such as those that typify chaotic mixing.

The practical consequence of strong Delta-scale dispersion is that overall transport of constituents through the system may differ significantly from what might be expected in the absence of dispersion; in particular, patterns and rates of organism entrainment at the export pumps might not be simply related to tidally averaged flow patterns. Thus, numerical models of transport used to inform policy must correctly represent the mixing effects of junctions. The challenge of this task is demonstrated by the complex flow behavior seen at the Mokelumne River–Georgiana Slough junction described by Gleichauf et al. (2014) (see also Gleichauf 2015). Tidal flows through this junction produce fronts (regions of separation) and secondary flows, both of which can significantly affect the paths taken by materials and organisms that pass through the junction (Figure 4). Nonetheless, as seen in modeling presented in Wolfram et al. (2016), on average, tidally varying flow trajectories produced by a well-resolved, 2-D, depth-averaged circulation model agreed well with the results of a high-resolution, non-hydrostatic 3-D model, suggesting that practical 2-D models might provide sufficiently accurate representations of junction flows to calculate large-scale transport. In contrast, 1-D models in common use (e.g., Kimmerer and Nobriga 2008) that assume complete mixing at junctions can significantly err in predictions of large-scale dispersion in the Delta, as well as in the
Freshwater flow affects the position of the low-salinity zone (LSZ) in northern San Francisco Bay, which is important habitat for many fish species (Kimmerer et al. 2013). The landward extent of salt from the ocean is determined by freshwater flow ($Q$) into the estuary. To control the landward extent of salinity, regulations require that $Q$ be adjusted to achieve a desired distance a salinity of 2 is from the Golden Gate Bridge ($X_2$) (Jassby et al. 1995). Given how strongly $X_2$ regulations affect water supply, forecasting how $X_2$ depends on $Q$ is an important component of project operations. Analysis of the flow and salinity data from 1967 to 1991 (Jassby et al. 1995; Monismith et al. 2002) suggested that the time-scale of $X_2$ response to flow is approximately 2 weeks, and that at steady flow $X_2$ is proportional to $Q^{-1/7}$—behavior that Monismith et al. (2002) suggest reflects the effects of tidally varying stratification on turbulent mixing, and hence the transport of salt by gravitational circulation. Using a synthesis of numerical modeling and analysis of more recent observations, MacWilliams et al. (2015) argue that: (1) the response of $X_2$ is fast when $Q$ is large, and slow when $Q$ is small; and (2) the dependence of $X_2$ on $Q$ is $Q^{-1/4}$—behavior that would be predicted from the classical theory of gravitational circulation accounting for actual variations in width and depth of the estuary. MacWilliams et al. (2015) also point out the central challenge of predicting $X_2$ when $Q$ is small: during lowflow conditions, uncertainty in $Q$ can be larger than $Q$ itself. They advocate using a numerical model to define the low-flow $X_2$–$Q$ relation and then (if necessary) using observed $X_2$ variations to infer flow.

Note that both the Jassby et al. (1995) and MacWilliams et al. (2015) models imply symmetrical responses of $X_2$ to flow increases and decreases. In contrast, Chen (2015) showed that the response to flow increases is much faster than the response to flow decreases, in essence reflecting that response time depends on the state of the estuary ($X_2$) as well as on the flow. Monismith (in press) used this idea to construct and validate an $X_2$ model in which the rate of change of $X_2$ is proportional to the difference between the current value of $X_2$ and the equilibrium value of $X_2$ based on the current value of $Q$. Since this latter model is based on the governing equations, rather than being purely empirical, it may be viewed as preferable from a theoretical standpoint. On the other hand, in practical terms, the physics-based model has quite similar accuracy to the time-series model (Figure 5). That said, one practical consequence of the asymmetrical model is that it implies that pulsed flows require a larger volume of water to maintain a given average value of $X_2$ than does the steady flow that corresponds to that average value of $X_2$.

Observations of salinity in the main channel of northern San Francisco Bay show that depth-
averaged salinity depends primarily on the distance of a given point in the estuary relative to X2 (Monismith et al. 2002). Measurements of X2 during the fall of 2011 (Stacey et al., in prep.) show that salinities in Honker and Grizzly bays tend to be lower in general than in the main channel, and respond more slowly to changes in flow and spring–neap variations in tidal mixing that also produce variations in X2 (Monismith, forthcoming).

**Sea Level Rise**

In addition to tides and salt, future sea level rise (SLR) will propagate into the Delta. Both the salinity field and tidal motions are the result of interaction of the Bay–Delta with physical forcing that itself varies in time. Outflow of freshwater from the Delta, the key factor that suppresses upstream propagation of salt, will likely change in the future because of changes in precipitation patterns caused by anthropogenic climate change (Cloern et al. 2011). At the same time, SLR is expected to have two effects: (1) gravitational circulation will be stronger when channel depths are larger, increasing upstream dispersive salt fluxes so the outflow required to maintain a particular value of X2 will be larger than it is now (Chua and Xu 2014); and (2) because of the likely flooding of large flat areas adjacent to the current Bay–Delta, frictional damping of tides will be stronger, and so tidal propagation through the system will likely change (Holleman and Stacey 2014). We note that this latter effect has not yet been investigated for the Delta itself–Holleman and Stacey’s modeling did not include the Delta. Sea level rise will also influence

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**Figure 5** X2 for 6 water years derived from USGS–USBR–CDWR fixed station CTD data (solid grey line) and as predicted by the MacWilliams et al. (2015) time series model (dashed line) and the Monismith (forthcoming) model (solid black line).
water diversions. Wang et al. (2011) simulated future water diversions from the Delta and found that at the end of the 21st century salinity intrusion from SLR of 61 cm will reduce the amount of freshwater available for water diversions.

Swanson et al. (2015) developed a Delta marsh elevation model and found that the magnitude of SLR over the next century was the primary driver of marsh surface elevation change. They defined marsh accretion parameters to encapsulate the range of observed values over historic and modern time-scales, based on measurements from four marshes in high- and low-energy fluvial environments. In addition, they modeled possible future trends in sediment supply and mean sea level. They also conducted a sensitivity analysis of 450 simulations that encompassed a range of porosity values, initial elevations, organic and inorganic matter accumulation rates, and SLR rates. More than 84% of the scenarios resulted in sustainable marshes with a moderate 88 cm of SLR by 2100, but only 32% and 11% of the scenarios resulted in surviving marshes when SLR was increased to 133 cm and an upper bound of 179 cm, respectively. Sediment supply was the next most important controlling factor.

**FLUVIAL FORCING**

The Sacramento River is the primary source of freshwater flow and sediment to the Delta. Wright and Schoellhamer (2005) found that, for the period from 1999 to 2002, the Sacramento River supplied approximately 85% of the water and sediment to the Delta. The second-largest source, the San Joaquin River, supplied about 12%, with the remaining supply from smaller tributaries such as the Cosumnes and Mokelumne rivers. The Sacramento and San Joaquin rivers both have USGS stream gages that have been in operation since 1957 measuring daily discharge and suspended sediment load (114447650 Sacramento River at Freeport and 11303500 San Joaquin River near Vernalis). Over this time-period, the average discharge and sediment loads on the Sacramento River were about 23,000 ft$^3$/s and 5,400 tons/day (note: this does not include the Yolo Bypass); on the San Joaquin River, these averages were about 4,800 ft$^3$/s and 900 tons/day. This suggests that the period from 1999 to 2002 roughly represents at the long-term averages, and that the Sacramento River supplies about 85% of the freshwater flow and sediment to the Delta. The majority of the sediment supply occurs during the winter wet season, typically between December and April. During 1999 to 2002. Wright and Schoellhamer (2005) calculated that 82% of the sediment supply occurred during the wet season, which constituted 31% of the total time Climate change will alter future discharge to the estuary. Cloern et al. (2011) simulated two warming scenarios, one with curtailed greenhouse gas emissions and one with continually increasing emissions. The scenario with curtailed emissions featured statistically insignificant decreases in precipitation and runoff from 2010 to 2099; the increasing emission scenario had statistically significant decreases in both. Both scenarios had statistically significant decreases in percentage of runoff that was snowmelt.

**Dams**

Dams and reservoirs in the watersheds draining to the Delta have affected the timing and magnitude of freshwater flows and sediment loads. In particular, sediment trapping in reservoirs has strongly affected sediment supply to the Delta. Reservoirs in the Delta watershed capture and store rainfall and snowmelt runoff for a variety of purposes, leading to changes in the flow hydrograph downstream from the dam. These changes typically take the form of decreased peak flows and elevated flows during other times of the year (Singer 2007). However, extreme flooding events still occur in the watershed (for example, 1986, 1997) because of areas of unregulated runoff as well as spills from reservoirs during periods of very high inflow. These extreme events are flood-managed in the valley, downstream from the reservoirs, by a series of weirs and bypass channels (such as Yolo and Sutter bypasses, Singer 2007; Singer and Aalto 2008).

The dams affect sediment supply more than flow, because the larger reservoirs capture approximately 100% of the incoming sediment load, releasing clear water and thus decreasing the downstream sediment supply. The two largest reservoirs in the Sacramento River watershed, Shasta Lake on the upper Sacramento River and Lake Oroville on the Feather River, capture runoff and sediment from large
mountainous areas that were likely large sources of sediment to the Delta pre-dam (some portion was also likely deposited in the Valley). Reservoir sedimentation is one factor that contributes to the approximately 50% decline in Sacramento River sediment loads since stream gaging began in 1957 (Wright and Schoellhamer 2004). In the present condition, the primary sources of sediment to the Valley floor and Delta are the largely unregulated Sacramento River tributaries in the northern Valley, that enter the Sacramento River downstream from Shasta Dam (such as Cottonwood Creek, Cow Creek, and Elder Creek). Singer and Dunne (2001), as well as analyses of unpublished data from recent turbidity monitoring along the mainstem Sacramento, indicate that sediment runoff is generated in these areas during intense rainfall events, and transported down the Valley by the Sacramento River to the Delta (with some deposition along the way, Singer and Dunne 2001; Singer and Aalto 2008). The Feather–Yuba–Bear watersheds contribute sediment as well; however, it is likely reduced from pre-dam sediment supply. The current-day supply of sediment from the Feather–Yuba–Bear watersheds is a data gap that hinders definitive analysis of sediment sources. That said, the primary source of sediment to the Delta today is likely the Sacramento River watershed between Shasta Dam and the Butte City area.

First Flush

The first major runoff event of the wet season, which typically occurs in the late fall or early winter, is commonly referred to as the “first flush,” and has particular significance in the Delta. For sediment transport, this event typically has the highest suspended-sediment concentrations, and results in a rapid change in turbidity throughout the Delta (Figure 6, Wright and Morgan 2015). The underlying mechanism responsible for this is sediment supply limitation. During dry periods, landscape disturbances increase the supply of sediment available for transport during runoff events. As the wet season progresses, sediment supply is gradually depleted, leading to decreased peak sediment concentrations (for the same discharge).

Notably, first flush sediments are typically high in associated contaminants and organic carbon (Bergamaschi et al. 2001; Hladik et al. 2009). Hladik et al. (2009) noted that suspended contaminant concentration was highest proximate to agricultural areas, but tended to be lower in the mainstem of the San Joaquin River. Likewise, toxicity was found to be higher in agricultural areas (Weston et al. 2008). Weston et al. (2014) studied pesticides in Cache Slough and found no pyrethroids during the dry season, but pyrethroids were toxic to a test organism after storms. Urban and some agricultural runoff into Ulatis Creek, a tributary of Cache Slough, was responsible.

There is also concern about the ability of sediments to transport mercury to the Delta during first flush. Elevated concentrations of mercury in suspended sediments have been reported (Domagalski et al. 2004; Roth et al. 2001). The concern is that these sediments will be deposited in reducing environments such as wetlands (Bergamaschi et al. 2012) where the mercury will be transformed into its organic and more toxic form: methylmercury (Bergamaschi et al. 2011).

Increases in Delta turbidity during the first flush event have been linked to Delta Smelt habitat, migration, and entrainment at export pumps (Moyle et al. 2016). During first flush in the lower Sacramento River, Delta Smelt migrate landward using lateral differences in tidal currents and turbidity (Bennett and Burau 2015).

During storms, when order-of-magnitude changes in concentration can occur in a matter of minutes, high-frequency sampling is necessary. Importantly, these variations are not necessarily in concert with variation in other parameters such as total suspended sediment. Saraceno et al. (2009), working in Willow Slough, a tributary of the Sacramento River, found that the peak in dissolved organic carbon associated with the storm did not align with either the peak in the hydrograph or the peak in TSS, consistent with their different sources (Florsheim et al. 2011; Oh et al. 2013).

Even for steady flows in rivers, sampling at high frequency may be necessary for chemical constituents with multiple sources or ones that are reactive in the environment. Pellerin et al. (2009) showed that nitrate loads calculated in the San Joaquin River using traditional methods can under- or over-estimate the true load by more than 30% because of
Figure 6  First flush along the mainstem of the Sacramento River in 2012. Turbidity (A), tidally-averaged discharge (B), and velocity (C). Turbidity varies with discharge at Freeport upstream from the Delta. Dispersion decreases the peak turbidity in the downstream direction. Travel time and bidirectional flow in the Delta delay the arrival of turbidity at Rio Vista and Mallard Island. Source: Wright and Morgan (2015).
variation in the nitrate concentration. A sampling frequency of approximately four per hour was necessary to accurately calculate loads and resolve sources.

**Sedimentation**

By definition, the Delta is a depositional environment for sediment. Deposition of sediment and organic matter contributes to building tidal wetlands in the Delta since the most recent sea level lowstand. The modern sediment budget of the Delta was evaluated by Wright and Schoellhamer (2005) and Morgan–King and Wright (2015), based on high-frequency monitoring of suspended sediment loads at a network of stations in the Delta. For the period from 1999 to 2002, Wright and Schoellhamer (2005) computed the total incoming suspended sediment load to the Delta to be 6.6 ±0.9 million metric tons (Mt) and the total outgoing sediment load to be 2.2 ±0.7 Mt, resulting in 4.4 ±1.1 Mt of sediment deposition over the 4-year period (about 67% of the incoming load). Bed load is approximately two orders of magnitude less than suspended load (Schoellhamer et al. 2012). The magnitude of deposition in the Delta can also be roughly evaluated over a longer time-period. McKee et al. (2013) computed sediment loads at Mallard Island (the downstream boundary of the Delta) for the period from 1995 to 2010, and incoming sediment loads are also available for this time-period for the Sacramento River (USGS Freeport gage) and San Joaquin River (USGS Vernalis gage). The missing piece of information for this time-period is the sediment load coming from the Yolo Bypass, which was found by Wright and Schoellhamer (2005) by using data from 1957 to 1961 and 1980 to be about 28% of the load at Freeport for 1999 to 2002. If the Yolo Bypass inputs are ignored, the average magnitude of deposition from 1995 to 2010 was 60%; if Yolo Bypass inputs are assumed to be 28% of Freeport loads, this magnitude increases to 67%. This suggests that the modern, decadal-scale sediment budget of the Delta is such that about 60% to 70% of the incoming suspended sediment is deposited within the Delta. This percentage varied from year to year, with a low of 39% in water year 1995 (a wet year, 1.63 Mt of deposition) and a high of 88% in Water Year 2007 (a dry year, 0.91 Mt of deposition). From 1995 to 2010, Water Year 1996 had the greatest depositional mass (4.53 Mt, 82%) and Water Year 2008 the least (0.27 Mt, 56%).

Morgan–King and Wright (2015) used a more extensive monitoring program to evaluate sediment budgets within the Delta for 2011 to 2013 for specific regions: the north Delta, central Delta, and south Delta. In general, the south Delta tends to have the highest trap efficiency (fraction of incoming sediment that is deposited), because the south Delta tends to be dominated by tidal forcing. In contrast, the north Delta tends to have the lowest trap efficiency, even though the incoming sediment load to the north Delta is by far the highest. This is because the Sacramento River can be dominated by fluvial forcing during high flows such that large amounts of sediment are passed through this region under these conditions. The central Delta tends to experience the most variability in deposition, primarily depending on upstream conditions in the Sacramento and San Joaquin rivers. During high flows, sediment loads into the central Delta through Georgiana Slough and from the San Joaquin River can be large, resulting in high deposition rates. In 2011, which was a relatively wet year, about 50% of the sediment that entered the central Delta was deposited (the highest among the three regions), whereas in 2012, which was a relatively dry year, only about 23% of the incoming sediment was deposited. This illustrates that the central Delta sediment budget depends on the upstream sediment supply.

Particles are not limited to inorganic sediment, and include plankton. Turbulence affects whether particles will remain suspended and susceptible to transport. Thus, turbulence affects the time plankton remain productive in the photic zone and whether this material will be intercepted by benthic grazers such as clams (Lehman et al. 2009; Lucas and Thompson 2012).

Freshwater flow from the Delta enters and affects San Francisco Bay. Larger Delta outflow can freshen central San Francisco Bay, and the resulting density difference flushes south San Francisco Bay, exporting sediment (Figure 7, Shellenbarger et al. 2013). When Delta outflow is small, central San Francisco Bay remains salty, there is little density difference between the central and south bays, and tidally averaged sediment transport is landward. Sediment
supply for the South Bay Salt Pond Restoration Project, the largest tidal wetland restoration project on the west coast of the United States, depends on salinity conditions created by Delta outflow. This demonstrates the dependence of south bay flushing and water quality on Delta outflow, first recognized by McCulloch et al. (1970).

Improved estimates of the historical supply of freshwater and sediment from the Central Valley watershed to the Delta and bay have been developed. Moftakhari et al. (2013) used tidal water level data recorded at San Francisco and tidal theory on the interaction of tides and river discharge to estimate discharge from 1858 to 1929. They found that the annual flow is now 30% less than before 1900, and confirmed that the flood of January 1862 was the largest since 1858. Moftakhari et al. (2015) used measurements of Sacramento River water level to extend this discharge record back to 1849. They also used historical sedimentation data to estimate a time-series of sediment supply for 1849 to 1929. About 55% of the sediment delivered to the estuary between 1849 and 2011 was the result of anthropogenic alteration in the watershed that increased sediment supply. Hydraulic mining in the Central Valley watershed created an initial pulse of sediment and, subsequently, urbanization in the San Francisco Bay Area increased agricultural land use, and forestry practices may have increased sediment supply. Sediment supply decreased about 50% since the 19th century, and the fraction of sediment delivered during winter has increased while the fraction delivered during spring has decreased, mimicking discharge patterns.

Since about 1900, the Central Valley watershed and Delta appear to be geomorphically adjusting as sediment supply from the watershed has decreased (Schoellhamer et al. 2013). Hydraulic mining from the mid-1850s to the mid-1880s created a pulse of sediment and aggradation, which was followed by a period of decreasing sediment supply and degradation. The period of adjustment from decreasing sediment supply may have ended about 1999, resulting in a stable regime. Hestir et al. (2013) analyzed monthly total suspended solids concentration data from the delta from 1975 to 2010 and found that large floods in 1983 and 1998 caused step decreases, likely from removal of erodible sediment (Figure 8). From 1999 to 2010 there was a significant decreasing trend in total suspended solids in the Delta, possibly resulting from supply limitation or trapping by aquatic vegetation. Hestir
et al. (2016) found that Delta sites with the greatest cover of submerged aquatic vegetation (SAV) had the greatest decreasing trend of total suspended solids. Schoellhamer et al. (2013) hypothesize that it is likely that the estuary and watershed can still adjust, but further adjustment will be as steps that occur only during greater floods than previously experienced during the adjustment period.

If more precipitation falls as rain rather than snow, creating greater runoff in winter and less in spring, as is the trend (Moftakhari et al. 2015); if storm intensity continues to increase (Russo et al. 2013); if reservoir trapping efficiency does not change; and if there is no more post-hydraulic mining geomorphic adjustment (i.e., erodibility of the watershed does not change, Schoellhamer et al. 2013), then sediment supply to the Delta may increase during the 21st century.

**Water Temperature**

Water temperatures in the Delta are generally determined by surface heat fluxes (Monismith et al. 2009; Gleichauf 2015) and thus can be well correlated with atmospheric temperatures (Wagner et al. 2011). Inflow temperatures are generally lower than temperatures in the interior of the Delta (Figure 9), so inflows can affect the spatial gradient of water temperature. Wagner et al. argued that any effect of inflows on the overall temperature was only discernible on short time-scales (approximately 1 month or less). On the other hand, large horizontal tidally averaged heat fluxes (of the same scale as diurnally averaged heating in summer) attributed to dispersion were found to be necessary to close the heat balances in the Sacramento Deep Water Ship Channel (SDWSC) at the Port of Stockton (Monismith et al. 2009) and the Delta as a whole (Gleichauf 2015), suggesting that flow-based heat transport may in fact be an important determinant of Delta temperatures.

An interesting aspect of the temperature distribution in 2011 was the seeming importance of cold water entering the Delta from the Merced, Tuolumne, and Stanislaus rivers, i.e., inflows much closer to the Delta than the more southerly sources of the San Joaquin (e.g., Friant Dam). From a management perspective, this suggests that cold water pools in reservoirs closer to the Delta might be useful to

![Figure 9: Delta wide average summer temperatures for 2011 and 2014. Source: Gleichauf (2015).](http://dx.doi.org/10.15447/sfews.2016v14is4art1)
help manage temperatures in the Delta, although this would also require managing the temperature structure in those reservoirs through the judicious use of selective withdrawal (Anohin et al. 2006; Fischer et al. 1979).

ANTHROPGENIC FORCING

Anthropogenic forcing of the Delta takes a variety of forms, such as (1) changes in volume and timing of Delta inflows (Moftakhari et al. 2013); (2) reductions in sediment fluxes into the Delta (Wright and Schoellhamer 2004); and (3) modification of mean and tidal flows through the operation of pumps, gates, and barriers. We presented the first two topics earlier in this paper; we discuss the later topic below.

Near the upstream end of tidal influence (around Freeport), the Sacramento River divides into several distributary channels, to the east and west of the mainstem. The east-side distributary channel, Georgiana Slough, connects with the lower Mokelumne River and, ultimately, the San Joaquin River in the central Delta. Along with the Delta Cross Channel (DCC), which connects the Sacramento River to the Mokelumne system when its gates are open, Georgiana Slough is the primary pathway for water and sediment to move from the Sacramento River to the central Delta. As shown by Wright and Morgan (2015), this pathway can result in elevated turbidity and sediment concentration in the south Delta, because of the landward net flows in Old and Middle rivers driven by pumping facilities in the south Delta. Once sediment enters the central Delta through the Georgiana Slough pathway, it is gradually advected southward though Old and Middle rivers into the south Delta; Wright and Morgan (2015) calculated a travel time of about 2 weeks from Freeport to the south Delta for a first flush sediment pulse in December 2012. This slow advection of sediment southward, which increases turbidity in the south Delta, has important implications because elevated turbidity in the region of the pumping facilities has been linked to high entrainment of Delta Smelt at the facilities (Grimaldo et al. 2009), which can lead to curtailments of federal and state water deliveries.

Both observations (Gleichauf et al. 2014) and numerical modeling of the Delta (Monsen et al. 2007) show that gate operations significantly affect hydrodynamics and transport patterns in the Delta. When the DCC gate is closed, Sacramento River water reaches the pumps through Threemile Slough and through the confluence region (Monsen 2000; Monsen et al. 2007). Because these transport paths are closer to the higher-salinity waters of the bay, salinities in the western Delta tend to be higher when the DCC is closed than when it is open, so meeting water quality standards in the Delta may require greater outflows into the bay. In a similar fashion, Monsen et al. (2007) show that operation of the Head of Old River Barrier (HORB) may significantly affect residence time in the SDWSC: residence time for particles in the SDWSC were five times smaller when the HORB was in place than when it was absent, a difference that may be important to the formation of hypoxic waters in that reach of the San Joaquin. Finally, it seems likely that tidally operated gates, notably the one in Montezuma Slough, can rectify tidal forcing and thus have significant effects on transport and salinity as well. For example, C. Enright (unpublished manuscript) suggests that when the Montezuma Slough gate is being operated to tidally pump freshwater into Suisun Marsh, it effectively diverts approximately 2500 cfs, possibly a large fraction of the dry-weather Delta outflow (approximately 3,000 to 8,000 cfs) that might otherwise be passing through Suisun Bay.

ATMOSPHERIC FORCING

The water surface is the interface between the atmosphere and Delta waters. Wind can alter Delta water levels and generate wind-generated waves, which can lift material resting on the bottom into the water column. The Delta breeze is the dry season afternoon and evening wind that blows from cooler Suisun Bay through the Delta and into the warmer Central Valley. Heat exchange at the water surface warms or cools Delta waters.

Wind-generated waves enhance sediment re-suspension in some shallow parts of the Delta. Ganju et al. (2005) found that wind wave re-suspension adjacent to Browns Island contributed suspended sediment to the main tidal channel on the island. Morgan–King and Schoellhamer (2013) observed increased suspended-sediment
concentrations (SSC) in Cache Slough and the SDWSC (which has a substantial fraction of shallow water) during spring and summer. These increases in SSC were well-correlated with wind speed. Landward sediment flux was greatest during these windy seasons, indicating that sediment retention was enhanced.

Along with wind wave re-suspension of sediments, wind waves also can greatly enhance mixing from wave breaking (Jones and Monismith 2008), an effect in the Delta that is likely to be especially pronounced in the shallow, open-water regions of Franks Tract (Jones et al. 2008) and, presumably, Mildred Island. The enhancement of near-surface turbulence by wave breaking is important in that strong turbulent shears and energetic vertical mixing, both effects of wave breaking, can act to break up colonies of cyanobacteria such as Microcystis (O’Brien et al. 2004) and, ultimately, suppress blooms (c.f. Huisman et al. 1999). More generally, wind mixing is important to phytoplankton dynamics in that near-surface turbulence plays a big role in determining the light climate experienced by individual phytoplankton cells (MacIntyre 1993).

Intensity of rainfall increased from 1890 to 2010 (Russo et al. 2013). Data from over 600 precipitation stations in the greater San Francisco Bay area, including the Delta and part of the Central Valley, were analyzed. The intensity of the largest (less frequent) storms increased the most, and a greater fraction of rain fell during large events. More intense storms increase the steepness of runoff hydrographs and increase sediment transport. They may also increase the release of metals from inter-tidal sediments (Moskalski et al. 2013).

Climate change will warm Delta waters. Delta water temperatures are strongly related to air temperature, and not as much to Delta inflow rates (Monismith et al. 2009), so increased air temperatures will warm Delta waters. For scenarios of curtailed and increasing greenhouse gas emissions, Cloern et al. (2011) and Wagner et al. (2011) estimate there will be statistically significant increases in Sacramento River and Delta water temperatures. Delta water temperatures will increases up to 5°C in summer, a change that may prove lethal for Delta Smelt. Given that growth rates for cyanobacteria increase exponentially with temperature (Robarts and Zohary 1987), this temperature rise may also enhance the likelihood of cyanobacterial blooms (Paerl and Huisman 2009). Whether this warming may increase thermal stratification was not evaluated.

Measurements made in the summers of 2004 and 2005 in the SDWSC of vertically varying temperatures show the formation and destruction of diurnal stratification that can be as strong as 2°C from the top to the bottom of the water column (Figure 10, Monismith et al. 2008). This temperature stratification does appear to be dynamically significant, and is the result of turbidity-dependent solar heating and mixing provided by the wind and surface cooling, as well as by turbulence production near the bottom. The practical significance of this stratification is that, given that existing descriptions of the temperature field of the Delta are based on the existing network of sensors, which are largely near-surface, it may be necessary to reconsider the nature of the effects of long-term temperature variability on fish habitat (Wagner et al. 2011; Gleichauf 2015). In particular, it may be possible that there are daytime refugia of colder water that temperature-sensitive fish such as Delta Smelt can use to avoid warmer surface waters. It would be most useful if the existing temperature sensor network were expanded to include near-bottom as well as near-surface temperatures.

**BIOGEOCHEMICAL FORCING**

Biogeochemical processes within the Delta can alter dissolved substances and suspended particulates. For example, Alpers et al. (2014) studied methyl mercury production in the Yolo Bypass and found methylation produced the highest concentrations in drainage from wild rice fields during harvest, and in white rice fields with decomposing rice straw during flooding. An example of biota affecting transport is invasive SAV in Delta channels. SAV began increasing significantly in the 1970s, and sites that now have greater SAV coverage had a greater turbidity decline in the later 20th century (Hestir et al. 2016). SAV can slow water movement and reduce bed shear stress, promoting sedimentation and reducing SSC. Turbidity trends were corrected for the declining sediment
supply (Wright and Schoellhamer 2004) using SSC data immediately upstream from the Delta. Hestir et al. (2016) estimate that 21% to 70% of the total declining turbidity trend results from SAV expansion.

DISCUSSION

The State of Bay–Delta Science 2008 (Healey et al. 2008) states:

... it is now recognized that the twice-daily tides are also extremely influential, causing powerful flow reversals through much of the Delta that amplifies dispersive mixing in both directions. Geometric features of Delta waterways, such as bends, junctions, shallow water areas and levees, all influence water transport and residence times.

Appreciation and consideration of tidal transport and effects were once limited to physical studies, but it has become more common in water quality and ecological studies, and has helped to improve our understanding of these fields (e.g., Bergamaschi et al. 2011; Bennett and Burau 2015).

Interactions of Multiple Drivers

We have summarized the state of the science for flow and transport by the primary drivers: ocean, fluvial, anthropogenic, atmospheric, and biogeochemical (Figure 1). Flow and transport in the Delta are determined by the interactions of these drivers, and management decisions must consider all these drivers.

Drought reduces the effect of fluvial forcing in the Delta and increases the effect of ocean forcing. The drought that began in 2012 decreases sediment supply from the watershed, increases residence times in the Delta, can increase salinity depending on regulation by water managers, and reduces hydrodynamic and salinity variability normally caused by freshwater pulses. Monitoring data sets should be analyzed to quantify the drought’s effect on the Delta. The Guadalquivir Estuary in Spain has a similar Mediterranean climate, and drought
there increased salinity and displaced communities upstream (González–Ortegón et al. 2015).

Salinity management must consider fluvial, anthropogenic, and ocean drivers. Freshwater flow to the Delta depends primarily on precipitation, snowmelt, and runoff from the Central Valley watershed. Dam operations modify the outflow hydrograph, especially for low flows. In the Delta, gates, barriers, and exports modify flow paths and the salinity field. As sea level rises, ocean forcing in the form of potential salinity intrusion increases. This will require increased anthropogenic forcing to maintain the existing salinity field in the estuary. In other words, ocean forcing in the form of SLR will affect dam operations in the watershed and fluvial forcing on the Delta. Thus, the ocean, estuary, and watershed are linked with one another, and humans have created a feedback mechanism from the ocean to the watershed.

Fluvial and anthropogenic forcing have combined to greatly alter sediment supply to the Delta, and the magnitude of the change is similar to other estuaries. The pattern of human disturbance, increased sediment supply, dam construction, and decreased sediment supply is not uncommon for the Delta. Sediment supply to the Delta increased about an order of magnitude because of hydraulic mining in the late 1800s (Gilbert 1917). Sedimentation rates increased 2- to 10-fold in other California estuaries in the 19th and 20th centuries (Warrick and Farnsworth 2009). These increases are typical of the 5- to 10-fold increase found in lake and marine sediment records downstream from disturbed watersheds (Dearing and Jones 2003). Sediment supply in the Sacramento River decreased about 50% from 1957 to 2001 (Wright and Schoellhamer 2004). This magnitude of decrease is not uncommon; river sediment discharge to the coastal zone has decreased 45% in southern California from trapping behind dams (Warrick and Farnsworth 2009), 50% to 70% from the Mississippi River (Kesel 2003), 75% from the Trinity River in Texas (Ravens et al. 2009), and, globally, riverine sediment discharge to oceans has decreased 10 ± 2% (Syvitski et al. 2005). Reforestation and dams have reduced the sediment discharge in the Changjiang (Yangtze River) 68% from the 1950s to 2000s, and the decrease is expected to reach 82% as dams become fully operational and the system adjusts to them (Hu et al. 2009).

Several mechanisms are responsible for the increasing clarity of Delta waters (Hestir et al. 2013) and clearer waters affect the Delta ecosystem. Decreasing sediment supply from the watershed, floods that export sediment, and sedimentation in SAV beds increase water clarity. Increased water clarity, in turn, may contribute to increased harmful algal blooms (Jassby et al. 2003), SAV expansion (Santos et al. 2009, 2012), and decreased fish abundances (Sommer et al. 2007).

In addition to turbidity, endangered Delta Smelt are affected by water movement, temperature, and salinity. Recent physical advances of significance to understanding and managing Delta Smelt include the ability of backwater sloughs (isolated dead-end channels) to trap sediment and increase turbidity (Morgan–King and Schoellhamer 2013), the possible role of SAV in decreasing turbidity (Hestir et al. 2016), and the role of first flush and tidal currents on their landward migration (Bennett and Burau 2015). Though restoration efforts focus on marsh habitats, backwater sloughs may be an important and scarce habitat for Delta Smelt worthy of restoration.

Marsh restoration is an anthropogenic driver, and its success depends on physical factors such as fluvial sediment supply, a nondestructive wind wave environment, and optimal tidal inundation for plant colonization and growth. Tides, channel geometry, and marsh vegetation may interact to retain sediment in restoration sites. Increasing sediment retention in the Delta would decrease sediment supply downstream to the bay.

**Information Gaps**

Here, we present what we believe are the most significant information gaps on flow and transport in the Delta, in roughly downstream order.

Though there is general understanding that sediment loads to the Delta have decreased over the past half century, likely from sediment trapping in reservoirs and other factors, the specific drivers of this trend are not understood. In particular, the current and historical watershed sources of sediment to the Delta, and how these sources have changed through time.
from anthropogenic influences in the watershed, are not understood. Because deposited sediment in the Delta essentially records the history of sediment sources, sediment “fingerprinting,” whereby the geochemical signatures of source and sink sediments are examined and linked, could fill this information gap. Changes in watershed sources, such as the elimination of sources upstream from dams, should be reflected in the sediments deposited in the Delta over time, assuming the geochemical signatures of sediments from different regions of the watershed are significantly different. In addition, the current-day supply of sediment from the Feather–Yuba–Bear watersheds is a data gap that hinders definitive analysis of sediment sources.

Water residence time is a key ecological variable that varies tremendously within the Delta. Some channels efficiently transport water and its constituents to the bay or pumps, and some dead-end sloughs have much smaller tidal excursions and longer residence times. Residence time may be particularly significant for the fate and adverse effects of toxic substances. The pristine Delta contained not only much more marsh but many more small dead-end channels, and undoubtedly had regions of longer overall residence time than today’s channelized Delta. Additional studies that examine residence time at the regional scale are needed to identify opportunities where increasing residence time improves habitat quality, and pelagic primary production.

To restore pelagic productivity in the Delta, the links between increasing water clarity, nutrient availability, residence time, and phytoplankton production need to be better understood (Cloern 2007; Lopez et al. 2006). Similarly, the relationship between flow characteristics such as velocity and turbulence and the production and sedimentation of organic-rich particles (phytoplankton, phytodetritus, and detritus) has not been well characterized for Delta environments; a mismatch between production and transport may lead to significant attenuation of this carbon supply to lower Delta food webs (Lucas and Thompson 2012).

SAV has increased dramatically in the Delta, particularly in the south Delta, which appears to trap sediment and increase water clarity. SAV, floating vegetation, and decreased sediment supply from the watershed may explain observed decreases in turbidity (Hestir et al. 2016). Given the potential strong feedback between increased clarity and increased SAV, these changes may be an important aspect of the future Delta. Thus, studies are needed to understand the effects of SAV on Delta sediment transport and turbidity.

A question that may be difficult to answer definitively is whether the Delta is now more or less turbid than it was before the Gold Rush when Delta Smelt were presumably more abundant. Dams, deforestation, mining, urbanization, agriculturalization, and river channelization have all changed radically since the Gold Rush, and affect sediment supply to the Delta. Reshaping of the Delta landscape likely altered sediment transport. With no data from before the Gold Rush, numerical models of the watershed and Delta in the early 1800s are likely the best albeit speculative approach to answer this question.

Expanding the existing temperature sensor network to include near-bottom as well as near-surface temperatures would enable monitoring of temperature stratification and near-bottom cooler water that could serve as daytime refugia for fish.

Prevention of salinity intrusion during drought is currently a topic of great interest that also merits additional study to determine how to efficiently prevent intrusion while minimally disrupting the ecosystem. Each barrier placement should be considered an experiment from which analysis of data collected before and after placement and removal would provide lessons for building a better barrier next time.

Uncertainty in low flow values of Delta discharge likely plays a major role in defining the uncertainty in forecasting low flow values of X2. A significant effort to reduce the uncertainty in low-flow values of Delta discharge is needed.

CONCLUSIONS

The tremendous expansion of acoustic and optical instruments deployed in the Delta over the past decade has greatly improved our understanding of how tidal variability affects flow and transport. Low Delta outflows are poorly estimated by a tidally
averaged water balance. Water quality sampling that fails to consider tidal variability can bias results. Transport in the Delta is driven more by tidal flow than tidally averaged flow. Mixing at junctions disperses water-quality constituents. Gates and barriers can significantly alter transport pathways. In addition, storms with similar time-scales of hours can supply and transport significant quantities of sediment and water-quality constituents.

Sediment is increasingly viewed as a diminishing resource needed to sustain the Delta ecosystem. Since 2000, the turbidity of Delta waters has decreased at an alarming rate. Native fish, including endangered Delta Smelt, favor turbid waters. Sediment supply is a key variable in determining the sustainability of tidal marsh as sea level rises.

Connections among the watershed, Delta, and the bay that have been quantified recently highlight that a landscape view of this system—rather than consideration of each region in isolation—is often needed. Reservoirs release water to maintain a salinity field in the Delta and bay that allows water diversions, and the volume of water needed to accomplish this will increase as sea level rises. The first flush of the watershed at the beginning of the wet season delivers a pulse of sediment and other constituents that affect the water quality and ecology of the Delta. The timing and magnitude of freshwater discharge from the Delta to the bay determines flushing and transport in south San Francisco Bay.

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NOTES

ABSTRACT

Anthropogenic climate change amounts to a rapidly approaching, “new” stressor in the Sacramento–San Joaquin Delta system. In response to California’s extreme natural hydroclimatic variability, complex water-management systems have been developed, even as the Delta’s natural ecosystems have been largely devastated. Climate change is projected to challenge these management and ecological systems in different ways that are characterized by different levels of uncertainty. For example, there is high certainty that climate will warm by about 2°C more (than late-20th-century averages) by mid-century and about 4°C by end of century, if greenhouse-gas emissions continue their current rates of acceleration. Future precipitation changes are much less certain, with as many climate models projecting wetter conditions as drier. However, the same projections agree that precipitation will be more intense when storms do arrive, even as more dry days will separate storms. Warmer temperatures will likely enhance evaporative demands and raise water temperatures. Consequently, climate change is projected to yield both more extreme flood risks and greater drought risks. Sea level rise (SLR) during the 20th century was about 22 cm, and is projected to increase by at least 3-fold this century. SLR together with land subsidence threatens the Delta with greater vulnerabilities to inundation and salinity intrusion. Effects on the Delta ecosystem that are traceable to warming include SLR, reduced snowpack, earlier snowmelt and larger storm-driven streamflows, warmer and longer summers, warmer summer water temperatures, and water-quality changes. These changes and their uncertainties will challenge the operations of water projects and uses throughout the Delta’s watershed and delivery areas. Although the effects of of climate change on Delta ecosystems may be profound, the end results are difficult to predict, except that native species will fare worse than invaders. Successful preparation for the coming changes will require greater integration of monitoring, modeling, and decision making across time, variables, and space than has been historically normal.

KEY WORDS

Climate change, climate variability, sea level rise, water resources, ecosystems, Sacramento–San Joaquin Delta
INTRODUCTION

The Sacramento–San Joaquin Delta (the Delta) is a hub where many flows, natural and artificial (water, nutrients, sediments, energy, and economics), converge and interact in California. And although the Delta has been in this same pivotal position throughout California’s history and prehistory, climate change is one stressor among the many that ensure that the Delta of the future will not be the same as the Delta we know today. Nonetheless, the Delta is at the foot of one of the largest, most complex water-management systems in the world, with hundreds of reservoir operations, canals, and diversions; a predictable if imperfect water-rights system; and vast swaths of managed lands above and contributing to it. That massive upstream machinery can be a source of some optimism in the face of climate change, as can the system’s long history of mostly-successful management of the wildest hydroclimatic regime in the country (Dettinger et al. 2011). If we work to understand the challenges and specifics of what climate change will bring, if we begin incorporating this understanding into decisions made today and tomorrow, and if we work to find the most effective adaptations and responses using our many natural and man-made assets, the Delta should be better off overall than many landscapes that will be facing climate-change challenges from much less robust starting points.

That is, the Delta is not a system that needs to wait passively for whatever challenges climate change brings. Looking forward, three particularly pressing scientific questions are:

- To what extent does the Delta system have built-in resiliency to future climate changes?
- Will (or when will) climate change push the system beyond its built-in resiliencies, whether physical, biological, or socio-economic?
- How will we know, and can we anticipate, when that resiliency has been exhausted?

To answer these questions usefully will require a deeper understanding of the changes to come, and of the natural variations that the Delta has experienced historically and that have been managed by society.

This review summarizes the current state of climate-change science as it applies to the restoration and sustainability of the Delta environment, facilities, and ecosystems, as a part of the 2016 State of Bay–Delta Science collection and report. These issues have been near the forefront of much intellectual activity concerning California’s water supplies and ecosystems, and often specifically the Delta’s ecosystems and water resources, with some major and recent studies of the potential effects of, and adaptations to, climate change in the Delta are listed in Table 1.

The challenges that climate change will pose to the Delta and Delta management can only be understood in the context of California’s already challenging natural climate and hydrologic variations. Thus, we begin this review with a brief synopsis of the state’s hydroclimatic variability in its natural state, and follow that with an overview of recent projections of 21st century climate change. We will then discuss sea level rise, droughts and floods, followed by climate-change challenges to the co-equal goals of water-resources reliability and ecosystems restoration and sustainability. We conclude with a discussion of key gaps in knowledge regarding climate change and its likely effects, and future science and monitoring directions to close these gaps.

HISTORICAL CLIMATE VARIABILITY

The climate of the Delta and its watershed is characterized by mildly cool, wet winters under prevailing westerly winds, followed by hot, dry summers. This seasonal pattern is shared by the Mediterranean region as well as parts of Chile, South Africa, and southern Australia. This climate regime yields strong seasonal variations in freshwater inflows to the Delta, which in turn are the source of much of the Delta’s physical and biological character. In addition to winter floods, spring snowmelts, and summer low flows, the Delta is also influenced, at its seaward end, by tidal inflows and outflows governed by natural daily, monthly, and seasonal processes. The coastal ocean also affects the San Francisco Estuary (the estuary) ecosystem and climate with its regular seasonal pattern of strong spring and early summer upwelling of cool, nutrient-rich waters.
On time-scales ranging from seasons to decades, the Delta’s natural (air) temperature variability is buffered somewhat (relative to much of North America) by California’s proximity to the vast Pacific Ocean heat sink (Dettinger et al. 1995). The catchment’s seasonal range of temperatures is generally less than seasonal swings in the continental interior, and its year-to-year temperature fluctuations are also less pronounced (in absolute terms) than other parts of the country. Nonetheless the catchment does experience brutal heat waves that can result in warm surface waters, dangerous increases in fire risks in the Delta’s upland watersheds, and significant swings in water demand by natural and, especially, human water users.

In contrast to the Delta’s comparatively buffered temperature regime, its precipitation and storm regimes are more variable and extreme than almost any other region in the country on storm-by-storm (Ralph and Dettinger 2012) and annual or longer scales (Figure 1; Dettinger et al. 2011). California’s most extreme storms have been a focus of much...
recent research, which has shown that these storms have historically been the result of landfalling atmospheric rivers (ARs). ARs are naturally occurring, transitory, long (>2,000 km), narrow (~500 km) streams of intense water-vapor transport through the lower atmosphere (<2 km above sea level). ARs gather and transport moisture over the North Pacific Ocean, connecting moisture sources from the tropics and extratropics to the West Coast (Ralph and Dettinger 2011). When these ARs encounter California’s mountain ranges, they are uplifted and cooled, and produce heavy rain and snow (Guan et al. 2010). The most intense ARs drop massive amounts of precipitation on the state. Among the largest storms in California’s history—storms that dropped more than 400 mm of precipitation within 3 days—92% have been ARs (Ralph and Dettinger 2012).

ARs are the dominant cause of the largest historical floods that have flowed through the Delta: over 80% of major floods (and levee breaks) since 1950 have been driven by ARs (Florsheim and Dettinger 2015). The Delta has experienced extremely large floods, including the New Year’s 1997 floods of recent memory and the winter 1862 flood (Figure 2), which may have exceeded the “record breaking” 1997 outflows by as much as 25% (Moftakhari et al. 2013). The 1997 flood and, very likely, the 1862 flood were caused by periods with more-or-less continual arrivals of warm AR storms on the central California coast and Sierra Nevada of warm AR storms (e.g., Dettinger and Ingram 2013). A notable characteristic of the Delta’s historical flood regime is that, although in most years high flows occur during the spring snowmelt season, the largest floods have nearly always occurred during winter months as a result of heavy and warm winter storms that yield rapid runoff and flooding of river channels and the Delta (e.g., Florsheim and Dettinger 2015).

At seasonal to multi-year time-scales, these large storms are also a key determinant of the Delta’s average flows and, especially, its large hydroclimatic variability. ARs bring the Sierra Nevada about 40% of its average precipitation and resulting streamflows (Guan et al. 2010; Dettinger et al. 2011). The arrivals, or not, of large storms—including, prominently, ARs—explain about 92% of the year-to-year and decade-to-decade variance of water-year precipitation (Dettinger and Cayan 2014; Dettinger 2016), including all the catchment’s major droughts during the historical period. Large AR storms also play an important role in ending sustained droughts in the historical period, ending about 40% of Delta droughts since 1950 (Dettinger 2013a). Although these large storms are increasingly being forecasted as much as a week or slightly more in advance (Wick et al. 2013; Lavers et al. 2016), their year-to-year variations remain poorly understood and forecasted. Taken together, the central roles that ARs play in California’s floods and its droughts strongly suggest their importance to understanding and managing hydrologic variability in the Delta on time scales from days to decades. ARs were first recognized only in 1998 (Zhu and Newell 1998) and so our scientific understanding of these features is quite new and still emerging. Their central roles in California’s hydroclimate have motivated wide ranging research to improve our ability to track, model and forecast ARs (Ralph and Dettinger 2011), including a major new storm-centered monitoring network led by the California Department of Water Resources (CDWR) and the National Oceanic and Atmospheric Administration (NOAA) (White et al. 2013); AR-focused modeling and forecasting efforts (Wick et al. 2013; Hughes et al. 2014); and, in recent winters, reconnaissance flights to visit and better
characterize ARs several days before their arrival in California (Ralph et al. 2016).

On these longer time-scales, some limited ability to forecast California’s temperature and precipitation derives from observations and forecasts of the state of the climate over the Pacific Ocean. Most attention in the past 2 decades has focused on the state of the El Niño–Southern Oscillation (ENSO) process in the tropical Pacific (Allan et al. 1996), which is the primary source of climate forecast “skill” (accuracy) almost anywhere in the world. El Niño events reorganize atmospheric circulations in the tropics in ways that divert and change the normal transports of heat and momentum (and, to an extent, moisture) out of the tropics towards extra-tropical regions, including the North Pacific and, ultimately, western North America. Thus, each time an El Niño (a period with anomalously warm sea-surface temperatures across much of the central to eastern equatorial Pacific) begins to form, there is much speculation about how it will affect winter precipitation over California. Unfortunately, across central to northern California, El Niño years have not yielded consistent precipitation outcomes at seasonal scales (e.g., Redmond and Koch 1991) and in terms of extreme precipitation or streamflow events (Cayan and Webb 1992; Cayan et al. 1999). That is, about as many past El Niño years have yielded dry weather as have yielded wet weather, although there is some evidence that the warmest El Niño years tilt the odds more decidedly towards wet conditions all along the West Coast, including in the Delta’s catchment (e.g., Hoell

Figure 2 Projected annual changes in air temperature, relative to 1961–1990 averages, in 10 selected global climate models (bright curves, 5-year moving averaged) and in 31 models (grey, unsmoothed), under low (A) and high (B) future greenhouse-gas emissions. (Source: CDWR Climate Change Technical Advisory Group 2015).
et al 2015). ENSO variability is mostly active in time-scales from 3 to 7 years, but interacts with the Pacific Basin beyond the tropics on longer time-scales, most notably in the form of the Pacific Decadal Oscillation (PDO; Mantua et al. 1997), which has historically influenced North American precipitation patterns for periods lasting for 25 years and more. The PDO, like ENSO, has historically led to stronger-than-normal contrasts in the amounts of precipitation falling in the southwestern U.S. compared to the northwestern U.S. but, also as with ENSO, the PDO's precipitation patterns tend to leave the Delta’s catchment with little precipitation certainty from year to year. Nonetheless, although these important global climate modes do not offer much predictability for Delta hydroclimate, they are almost certainly major contributors to the large range of precipitation amounts that the catchment receives from year to year. Arguably, an important but understudied part of the multi-year variation of precipitation over the Delta’s catchment occurs on time scales that are between the 3- to 7-year ENSO characteristic and the 25- to 70-year PDO scales; however, this decadal (14- to 15-year) variation is not well understood and, although significant during most of the 20th century, has come and gone in longer term tree-ring records (Meko et al. 2014; St. George and Ault 2011).

In the Delta’s widely varying precipitation regime, drought is a fact of life. The catchment has experienced severe short droughts (such as 1976–77) and less severe but more sustained droughts (such as the 1920s and 1930, or 1987–92) in the historical period. Tree-ring reconstructions of droughts in northern California have documented numerous droughts during the past 2000 years, including strong evidence of much longer and more severe droughts in the past (e.g., Meko et al. 2014; Ault et al. 2014). Precipitation deficits in the current drought (2012–present) have been extreme, although not record-breaking in water-year precipitation aggregates. On longer time scales, though, precipitation deficits during this current drought have been record breaking (e.g., in 14-month, 3-year, and 4-year totals) and have been characterized by very wet episodes bracketing the persistent dryness. For example, January 2013 through February 2014 was the driest such “season” since 1895, comprising a string of extremely dry months beginning immediately after strong AR storms in December 2012, and closing with the arrival of major AR storms in March 2014. This scenario is of special concern because it mimics, to an extent, the way that climate-change projections for the Delta are characterized by occasional very wet conditions separated by longer, drier droughts (see Dettinger 2016, and the next section, “Climate Change”).

Even more concerning has been that current drought conditions have been much aggravated by the record-breaking warm conditions that prevailed in 2014 and 2015 (Dettinger and Cayan 2014; Griffin and Anchukaitis 2014). Warmer conditions during droughts exacerbate precipitation deficits with drier soils yielding less runoff, as well as and longer periods with much reduced freshwater inflows, more wildfire risk, and warmer streams. Increasingly, warm droughts are also a consensus projection for our future climate (see “Climate Change”).

As a consequence of the large storms and long droughts that California has experienced naturally, the Delta has historically faced great floods and great droughts. These extremes have shaped the land and California’s infrastructure, politics, economy, and society (e.g., Kelley 1988) in ways that we will need to mobilize and exploit in order to address the new challenges of climate change.

**CLIMATE CHANGE**

In the next several sections, we summarize the current state of science for several aspects of climate change as it will influence the Delta. Most work to date has begun with consideration of long-term projected changes in temperatures and precipitation, and this section focuses on projected trends in these variables. Confidence is high in the continuation of warming trends, if greenhouse-gas concentrations continue to increase, and so long as global warming continues, sea levels are likewise expected to rise. Thus, we consider sea level rise in the next section. Recent climate change research around the Delta has increasingly focused on the projected future of hydroclimatic extremes, such as major storms, floods, and droughts. The state of science for hydroclimatic extremes in the Delta will comprise the third section that follows (“Droughts and Floods: Climate Extremes”), before we discuss in subsequent sections
the water management ("Water Resources Effects") and ecological implications ("Fisheries, Habitats, and Ecosystem Effects") of findings to date.

California has warmed by over 1°C since the late 19th century (Hoerling et al. 2013), and all modern climate models indicate that Earth’s climate will continue to warm as greenhouse gases accumulate in the atmosphere as a result of fossil fuel combustion and other anthropogenic effects. By 2025, the California Delta and its watershed is projected to warm above late 20th century levels by another 1°C; by 2055, between 2°C and 2.5°C; and by 2085, between 3.5°C and 4°C (Figure 3, depending on how much global greenhouse-gas emissions continue to increase; Cayan et al. [2008b]). This warming scales nearly linearly with cumulative carbon emissions into the atmosphere, so if a lower emissions pathway were achieved globally, through aggressive and rapid transitions to economies less reliant on fossil fuels, the warming would be significantly less (Maurer 2007; Tebaldi and Arblaster 2014).

Within the Delta’s catchment, local differences are certain to arise. For example, warming is likely to be amplified the farther from the coast one moves, and higher altitudes may warm faster than lower altitudes (Wang et al. 2014). The resulting amplification of warming inland across the Delta’s watershed may cause enhanced sea breezes with cooler coastal air that penetrates further inland, an effect that has already been detected in California (Lebassi et al. 2009). This effect may also be affected by (and affect)

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**Figure 3** Projected annual changes in precipitation, relative to 1961–1990 averages, in 10 selected global climate models (bright curves, 5-year moving averaged) and in 31 models (grey, unsmoothed), under low (A) and high (B) future greenhouse-gas emissions. (Source: CDWR Climate Change Technical Advisory Group 2015).
changes in coastal upwelling of deep sea waters (Snyder et al. 2003).

Future changes in precipitation are much less certain than warming and some other changes like sea level rise and surface air humidities (Cayan et al. 2008b). Among global climate models, about half project increasing annual precipitation for the Delta’s catchment and half project decreasing precipitation (Figure 4). Within this uncertainty about annual totals, more than half of the models project precipitation increases in winter months and declines in the spring and fall seasons (Pierce et al. 2013b). Also, most projections indicate that by the middle of the 21st century there will be fewer days with precipitation, but increases in the intensity of the largest storms (Pierce et al. 2013a; Polade et al. 2014; Dettinger 2016). To date, no strong consensus has emerged among modern projections about to the future prevalence of El Niño or PDO events (Vecchi and Wittenberg 2010), although the range of future ENSO fluctuations may increase (Cai et al. 2015). Thus, even the meager guidance about northern California precipitation that knowledge of future El Niño and PDO behavior would provide is not yet available to inform plans for future precipitation variations over the Delta watershed.

Winter snowfall and spring snow accumulation in the western United States have declined in recent decades, largely in response to warmer temperatures (Knowles et al. 2006; Mote et al. 2006; Kapnick and Hall 2012). Attendant changes in the timing of snow-fed streamflow have already been detected (Fritze et al. 2011). Springtime snowpack will decline significantly in the Sierra Nevada as climate warms, quite likely by at least half of present-day water contents by 2100 (Knowles and Cayan 2002; Maurer et al. 2007; Cayan et al 2008b; Pierce and Cayan 2013). As a result, by 2100, arrivals of snowmelt-fed inflows to the Delta will be delayed by a month or more. As snow retreats in a warming climate, the exposed land surface absorbs greater solar radiation, which produces a positive feedback that can accelerate local warming and snow retreat, an effect not well represented in most current projections (Pavelsky et al. 2011). The effect implies that the rate of snow loss and melt may be even more rapid than has been projected so far.

The details of these influences of warming (and precipitation change) on snowpack and snow-fed streamflows in the Delta watershed are strongly modulated by the complex topography of the state’s mountain ranges. Because global climate models (GCMs) yield climate projections on coarse spatial grids, with resolutions ranging from about 100 to 200 km, a process called “downscaling” is applied to re-introduce spatial details of climate differences.

Figure 4 Freshwater outflows from the San Francisco Estuary, as tidal-discharge estimates (TDE) based on tidal gages in San Francisco Bay at the Presidio, as a function of years in the past and time of year, illustrating the high flood flows in winter 1862 and many subsequent occasions. (Modified from Moftakhari et al. 2013.)
and variability that drive most of the watersheds, rivers, and systems of California water. The spatial resolutions of GCMs are improving, but the level of spatial detail they will provide is likely to be 50 kilometers or coarser through the next decade.

Two methods have been used in most downscaling efforts to date (CCTAG 2015): Dynamical downscaling simulates local-to-regional weather responses to coarse GCM outputs. These full-physics (or dynamic) models represent the physics of weather and climate as best we understand them at high resolutions and thus provide a full suite of climate variables (beyond “simply” temperatures and precipitation). But they also have limitations, including their own biases, uncertainties about observations to which the models are calibrated, and high computational storage requirements. The primary alternative has been statistical downscaling whereby historical weather patterns in response to various large-scale climatic conditions are interpolated into the GCM outputs by various statistical means. Statistical downscaling has the advantage that downscaled products are less computationally burdensome to develop and thus can be produced from large numbers of climate-change projections. That said, all statistical downscaling hinges on some assumption of “stationarity”—that relationships of historical large-scale to finer-scale variations will apply in the future. The statistical methods inevitably depend on the quality of historical observation data used to develop the statistical relationships.

At present, statistical-downscaled products are most widely used and are probably acceptable to meet immediate needs, as well as being consistent with several iterations of climate assessments in California in the past dozen years. Nonetheless, in years to come, either new statistical methods, new hybrids that apply combinations of both dynamic and statistical tools, or, eventually, dynamical downscaling will be needed to address the full range of issues that may threaten the Delta.

Returning to the issue of how warming will likely affect riverine inflows to the Delta, as winter storms warm and become rainier (less snow), and snowpacks melt earlier, a greater fraction of runoff generated will pass through the Delta earlier in the year. As a result, summer salinity in the upper San Francisco Bay and Delta is projected to increase (Knowles and Cayan 2004; Cloern et al. 2011). The combination of changes in temperature and precipitation, resulting in a much reduced snow regime and occasional more intense storms, is also projected to increase the frequency and magnitude of floods in the river systems that feed the Delta. By the end of the 21st century, this was found to produce robust increases in floods with return periods from 2 to 50 years for both the northern and southern Sierra Nevada, regardless of whether the climate projections considered were for overall wetter or drier conditions (Das et al. 2013).

Changes have been detected in other aspects of surface climate, including a reduction in wind speed (Vautard et al. 2010), though the driving cause is not primarily large-scale warming. Projections of large-scale wind changes over the Delta have not been much explored and remain quite uncertain, even among projections by a single climate model (Dettinger 2013b), although, as noted previously, Delta breezes may intensify. Though total atmospheric moisture content is projected to increase, warmer surface-air temperatures offset that effect to produce declines in relative humidity by as much as 14% for California (Pierce et al. 2013c). This decline would result in greater potential for evapotranspiration from soil and vegetation, intensifying hydrologic droughts. However, as CO2 concentrations in the atmosphere increase, plants tend to use water more efficiently (called a “direct CO2 fertilization effect”), which could offset some of the greater atmospheric evapotranspiration potential; but as temperatures rise, growing seasons will also tend to lengthen, which in turn will contribute to increases in total evapotranspiration (Lee et al. 2011). The net effect of these several countervailing influences on overall evapotranspiration and vegetation water demands remains a topic that needs more research, but the U.S. Bureau of Reclamation has concluded that overall agricultural-water demands in the Central Valley will increase (USBR 2015).

On the whole, uncertainties about many of these projections are smaller than they were 2 decades ago. But, perhaps as importantly, projections today do not differ markedly from projections in the past several Intergovernmental Panel on Climate
Change assessment cycles. That is, modern climate projections seem to have largely converged toward the values that we currently report. Nonetheless, our ability to predict the future climate over the Bay–Delta’s catchment is limited by several sources of uncertainty (Hawkins and Sutton 2009, 2011): (1) uncertainties concerning the rates at which greenhouse gases will be emitted into the atmosphere in the future; (2) uncertainties concerning climate-system responses to the changing greenhouse gas concentrations (essentially climate-model uncertainties and differences); and (3) the limits of long-lead predictability of natural variations of the climate system; for example, the fluctuations of ENSO and the PDO. Natural variability (#3) plays a declining role in terms of projected temperature (and temperature-driven) changes on time-scales beyond about 2 decades. The second source of uncertainty dominates uncertainties by mid-century, and by the end of the 21st century (and beyond) the first uncertainty dominates. Precipitation projections for California, by contrast, vary largely from natural variability throughout the 21st century, but with gradually increasing uncertainty deriving from the second source later in the century.

Delta systems, both natural and human-developed, are susceptible to the effects of climate change to varying extents and on differing time-scales. Effects are likely to include altered water supplies, increased flood and levee-stability risks, and important challenges to the sustainability of species and the Delta ecosystem as we know it (Cloern et al. 2011). Decisions about adaptation should accept and, indeed, expect uncertainties in projections (Mastrandrea and Luers 2012). The first source of uncertainty can be partially accommodated by considering both ends of the emissions-pathways spectrum, although as a practical matter, it is worth noting that projected climate changes early in the 21st century tend to be similar regardless of the emissions pathway assumed, but then the changes associated with different emissions pathways differ increasingly after mid-century. Because we cannot determine which of the climate models provides the most accurate projections of the future, standard practice is to consider the statistics (and especially the extent of consensus) of projections from collections or ensembles of different models, in hopes that the outcomes upon which the models agree most are the outcomes least subject to the second type of uncertainty. Attempting to characterize likely climate change effects using too few model projections runs the risk of accidentally over-emphasizing specific natural wetter or drier fluctuations in the various (few) projections, under-representing the full range and consistencies among plausible futures. In the past decade, the numbers of climate models and climate change projections available for these ensemble analyses has increased and, with them, confidence has improved in many aspects and statistics regarding likely climate changes and effects. Furthermore, detailed outputs from historical simulations by the 30 or more climate models now in use are more readily available than they were a decade ago, so that the models that perform worst in historical simulations (and their projections) can be culled from the ensembles before they contaminate assessments of likely climate change effects (CCTAG 2015). Because climate models are not synchronized (for example, as to when El Niño events occur), using an ensemble of century-long projections also reflects the evolving role of natural climate variability more clearly (e.g., Dettinger et al. 2004).

The greater confidence regarding projections of warming and the larger uncertainties concerning how precipitation will change suggest that adaptations which accommodate warming (and its consequences) might be acted on more confidently (deterministically) than adaptations directed at future precipitation changes. The greater uncertainties around precipitation change do not argue for less attention to—nor for less urgency about—adaptations to possible precipitation changes. Rather, they imply that adaptations to changing precipitation and water supplies should focus on increasing the range of possible water futures that the Delta systems—engineered and natural—can accommodate sustainably.

SEA LEVEL RISE

Water levels in the Delta are not much higher than coastal sea level, and thus will be affected by sea level rise (SLR). Astronomical tides are attenuated as they propagate landward through the north bay and into the Delta, but are still readily detectable.
The Delta and its surrounding borders are low lying, making Delta landscapes and hydrodynamics vulnerable to water level increases and extremes. During the 20th century, sea levels along the California coast rose about 20 cm (Cayan et al. 2008a; NRC 2012). Because of global warming, SLR is projected to continue, and very likely will accelerate during the 21st century (NRC 2012). Satellite altimetry has indicated that global SLR rates increased during the last 2 decades—from about 2 mm yr\(^{-1}\) to about 3 mm yr\(^{-1}\) (Hay et al. 2015). The rate of SLR along the California coast followed global rates closely during the 20th century. However, there is considerable variability on shorter time-scales. For example, the West Coast has experienced little SLR during the last few decades, while the eastern Pacific has exhibited SLR at three or more times the global rate (Bromirski et al. 2011) because of wind and pressure differences across the Pacific Ocean. Projections of the amplitude of 21st century SLR remain fairly uncertain, largely reflecting uncertainties about temperature changes and ice-cap loss rates, but most end-of-century estimates are between 0.2 m and 1.7 m of additional rise from the end of the 20th century, with outliers mostly projecting potentially even more rise (Pfeffer et al. 2008; NRC 2012; Hansen et al. 2016; DeConto and Pollard 2016).

Within the Delta, subsidence of Delta islands increases risks from SLR (Mount and Twiss 2005; Brooks et al. 2012). Increased water levels in the Bay/Delta are projected to change the tidal regime in the estuary (Holleman and Stacey 2014). Depending on how the estuary’s shorelines change in coming decades—e.g., with hardened seawalls and levees vs. restored wetlands—the tidal regime could become more amplified or more dissipated, yielding wider tidal ranges, with even local shoreline changes affecting tidal ranges in parts of the estuary both near and far. Many problems associated with SLR will be amplified or hastened when large storms coincide with high astronomical tides (Cayan et al. 2008a). Strong storm winds and wind waves compound the effects of flooding along the Delta’s land-water boundaries. Storm-generated freshwater flood flows may dwarf the high sea levels; flood stages in the Delta’s upper reaches stand several feet above normal levels. The resulting high waters increase the risk that Delta lands and surroundings will be inundated and levees breached.

Although short-term water-level extremes are of early and pressing concern, even the most gradual expressions of SLR will eventually transport more ocean salinity into the Bay–Delta (Knowles and Cayan 2004; Cloern et al 2011). Increased salinities will affect brackish and freshwater habitats and, unless managed very skillfully, threaten water supplies (more in “Water Resources Effects”).

**DROUGHTS AND FLOODS (CLIMATE EXTREMES)**

As temperatures rise, the character of California’s climatic and hydroclimatic extremes is almost unanimously projected to change. Some events are extreme because of their size relative to historical climate distributions while other events are extreme because they comprise never before seen combinations of events. Both types of extremes will likely increase in frequency and magnitude, ultimately crossing thresholds that require reassessment and adaptation of management and restoration strategies. Understanding the underlying processes is key to understanding how to adapt to these “new” events. The current drought (2012–present) highlights these considerations: Over the past 4 years, temperatures have reached new highs, and snowpack has declined to record lows while precipitation deficits have been challenging but not record-breaking. Thus, this drought has provided both record-breaking extremes (in isolation) and a historically new set of hydrologic challenges for water management. In the Delta, new water-quality challenges and greater vulnerability to salinity intrusion have resulted. Outcomes such as these are expected to become more frequent in the coming decades.

At the other extreme, central California’s largest floods have historically been driven by winter storms with heavy rains that reach higher up into the mountain watersheds than most. When these storms and floods have coincided with extreme winter tides, storm surges and high wind waves, they have formed a dual threat (high river flows and water levels) for Delta levee failures and flooding within the Delta. Warmer storms yield higher flood
flows because more of the watershed receives rainfall, and contributing runoff that immediately runs off, rather than snow, which accumulates in snowpacks. Warmer temperatures also can support greater atmospheric moisture influxes that may lead to higher precipitation rates and, thus, higher flows. At the same time, a large majority of climate models project that the numbers and (less so) intensities of ARs making landfall in California will increase significantly in the 21st century if greenhouse-gas emissions continue to increase (Dettinger 2011; Warner et al. 2015; Gao et al. 2016). Together these changes are projected to result in larger peak flows and flood risks in the warming future (Figure 5).

In current climate-change projections, both droughts and floods increase as the climate warms, with storms becoming more intense, and intervening periods drier, longer, and warmer. Although changes in these extremes have not been detected with any confidence to date, these projections offer a vision of the future in which more severe droughts tempt us to store more (increasingly, cool-season) runoff even as more severe floods motivate us to release more water in pursuit of greater flood-mitigation capacity behind our primary dams. Unique new management balances between flood-control and water-supply management imperatives will likely be needed. Water year 1997 might provide an inkling of the problems involved. Following the record-breaking floods of New Year’s 1997, the late winter and spring of 1997 was one of the driest on record, so that water released in coping with the winter floods was sorely missed later in the year. Although these conditions are disruptive to the human built system, flood and drought are natural conditions that the Delta’s ecosystems have evolved to accommodate and, in some cases, even benefit from (e.g., Opperman et al. 2009; Moyle et al. 2010; Opperman 2012).

Two important “climate change” problems that Delta science will need to resolve (or see resolved) are better understanding and prediction of future extreme events and their implications for ecosystem conservation and water supply, and identifying and anticipating thresholds beyond which these extreme events will result in substantially new adverse effects on management and adaptation.

**WATER RESOURCES EFFECTS**

Water management in and for the Delta is an ever-evolving process of addressing competing needs for a reliable supply of high-quality water, protecting and restoring ecosystems, controlling floods, and satisfying legal and regulatory requirements in the
face of highly variable climatic and hydrologic conditions (CDWR 2008; Lund 2016). Climate change will almost certainly exacerbate the challenges inherent in that process.

The many effects of climate change on the Delta outlined earlier will very likely affect operation of all water projects and uses that rely on freshwater transports through the Delta. Along with the climate uncertainties, changes in land cover and use in response to climate-change and other stresses will exacerbate the challenges to water-resources and flood-risk management even more, and should be an important focus of future assessments.

Trends toward declining late-winter and spring flows are already evident on both the Sacramento and San Joaquin rivers (Figure 6). Since the upper reaches of the Sacramento watershed are at lower elevation than those of the San Joaquin watershed, the Sacramento watershed is more sensitive to the modest temperature increases—and the attendant shifts of

Figure 6 Full-natural (reconstructed natural) April–July streamflows in the (A) Sacramento and (B) San Joaquin Rivers, as fractions of water year totals, since early 20th century (from CDWR); red line is a least-squares trend and pink curve is a 3-year moving average. The variance captured ($r^2$) by the fitted trends are (a) 9.5% and (b) 6.1%, respectively, with $p < 0.05$ in both cases.
precipitation from snow to rain and earlier melting of snowpacks—experienced thus far. In the second half of the 21st century, however, warming will have long since driven precipitation-form changes and earlier snowmelt to their practical limits in the Sacramento catchments but will continue to cause ever-larger increases in peak flows and more dramatic shifts in seasonal timing from the San Joaquin Basin (Das et al. 2013; Maurer et al. 2007). Since the Sacramento River provides nearly 80% of the freshwater inflow into the Delta (CDWR 2014b), losing the natural reservoir of snowpack in that basin will be a major challenge to the state’s water resources management (Dettinger and Anderson 2015). On the other hand, the snowfields of the San Joaquin Basin have more capacity to change in the face of continuing warming trends, so that by the end of this century some of the largest proportional challenges will likely arise from this tributary.

Water managers have recently been confronted with present-day examples of what these future changes might look like. During the current drought, each year’s average April 1 snowpack water content has been among the bottom 10 values in the record dating back to 1950. Before 2015, the previous low snow pack was 25% in water years 1977 (from lack of precipitation) and 2014 (from the combination of a moderate lack of precipitation and record-breaking warm winter–spring temperatures). Then, in 2015, the April 1 snow pack was an unprecedented 5% of historical average, reflecting moderate lack of precipitation again and even higher winter temperatures. Recent climate-change projections do not yield snowpacks this low more than 10% of the time until after about 2070 (Figure 7). But, as climate change proceeds, such low snowpacks will become progressively more common, so that 2015 can be viewed as an early warning of challenges to come.

These changes in temperature, snowpack, and runoff timing result in a greater fraction of annual flow volumes passing through the Delta during the time of year historically managed (by mandate) for flood control, that is, before April 1. This timing shift is expected to cause a cascade of changes in the watershed and Delta systems. For example, it has been estimated that, by the end of the 21st century, one or more of the major reservoirs that feed the Delta will be unable to release water during critical warm-season months because of low reservoir levels as often as once every 3 to 8 years (CDWR 2009); reservoir levels this low have not yet been experienced. Future declines in the amounts of water in storage at the end of the water year in upstream reservoirs (CDWR 2009) are analogous to a shrinking saving account, which reduces the ability to draw from those savings later, in times of need and shortfall. Reductions in upstream reservoir releases can be expected to result in increased groundwater pumping downstream (CDWR 2009, 2014a; Hanak and Lund 2012).

Projected SLR will increase pressure on over 1000 km of levees that surround Delta islands and protect the river channels that constitute a water supply conveyance corridor (CDWR 2014b). Many of these levees were not designed or built to modern engineering standards (Deverel et al. 2016). Salinity intrusion from SLR will require increased releases of freshwater from upstream reservoirs to repel that salinity (CDWR 2009). Careful evaluations of California’s water operations have indicated that Delta inflows can be managed to maintain the position of the X2 (position with a bottom-water salinity concentration of 2 ppt) under many such futures (CDWR 2009). However, maintenance of salinity levels at other locations (e.g., Vernalis on the San Joaquin River) poses its own challenges.

![Figure 7](image-url) - Odds that a year yields less than 5% (red) or 10% (orange) of 1961–1990 average April 1 snow-water equivalent across the mountains of California, in an ensemble of simulations and projections by the VIC macrohydrologic model (Liang et al. 1994) as forced by BCSD-downscaled (Wood et al. 2004) outputs from 16 global-climate models under high-(A2) and low-(B1) emissions scenarios. (Updates to results in Cayan et al. 2008b.)
Reservoir releases to repel salinity reduce the amount of water available for other purposes (CDWR 2009). With current operating rules, this tradeoff has been projected to reduce the amount of water available for export from the Delta by about 10% under mid-century climate projections, and by about 25% by end of century (CDWR 2009). Current operations are governed by complex water rights, contracts, water quality standards, biological opinions, flood control rules, agricultural and economic forces and demands, and human health and safety requirements. However, the actual effects of climate change will depend on future operating rules and future decisions, including responses to climate change itself, and the California Water Plan states that “The water management community has invested in, and depends on, a system based on historical hydrology, but managing to historical trends will no longer work because historical hydrology no longer provides an accurate picture of future conditions” (CDWR 2014a).

In addition to these salinity-management challenges, projected changes in the amount and timing of fresh water inflows combined with SLR could change water quality in other ways. For example, Ficklin et al. (2013) simulated water quality in the Sacramento and San Joaquin watershed and found that water-temperature increases of 2 to 2.5 °C could result in 10% declines in dissolved oxygen (DO) in the rivers, with high potential for detrimental effects on water quality and aquatic species. Rising sea levels and more frequent flooding of the Yolo Bypass may inundate previously dry areas, and, if conditions are right these could become new areas for the occurrence of mercury methylation (Fong et al., submitted). Increased bromide concentrations from seawater intrusion might threaten drinking-water uses (Fong et al., submitted). Much additional research is needed if we are to understand and predict the effects of climate change on water-supply quality.

Another complication in evaluating the effects of climate change is that the geometry of the Delta will likely change as a result of planned structural modifications, natural forces, and combinations of the two (Lund et al. 2008). The currently proposed Water Fix and Eco Restore programs (formerly known as the Bay Delta Conservation Plan) include plans to add water-conveyance tunnels under the Delta to move high-quality water from the Sacramento River safely to the export pumps in the south Delta (CNRA 2015), resulting in a hydrodynamically very different Delta. Delta islands could become flooded by levee failures (e.g., the 2004 Jones Tract levee failure) from an earthquake or major flood, or by planned breaching of levees to flood islands (Lund et al. 2008; Florsheim and Dettinger 2015). How changes in the geometry of the Delta might exacerbate or mitigate challenges from climate change is another area that needs more study.

More than 200 federal, state, regional, and local agencies are responsible for managing various components of the Delta system (CDWR 2014b), and have a long history of coping with the region’s highly variable climate and hydrology. As noted earlier, this is a cause for some limited optimism. However, although many future conditions will fall within the range of historically observed conditions, even more extreme events are expected to occur in the future. For example, increases in heavy precipitation are projected with high confidence and are already being observed (Kunkel et al. 2013; Pierce et al. 2013a; Dettinger 2016). At the other extreme, future droughts are projected to become more frequent, with, under the influence of warmer temperatures, higher evaporative demands and increased numbers of dry days overall (Cayan et al. 2010; Polades et al. 2014; Cook et al. 2015). Notably, the persistent high pressure over the northeastern Pacific that has steered storms away from California, causing most of the precipitation shortfall in the ongoing 2013–2015 California drought, has been projected to be three to four times more likely in today’s changed climate than under pre-industrial conditions (Swain et al. 2014). The combined effects of precipitation deficits and record-breaking warm temperatures have resulted in the current drought being even more intense than the 1977 drought, with an estimated 200-year recurrence interval (Aghakouchek et al. 2014). In fact, extremely dry soil-moisture conditions during 2014 and 2015 may be without precedent in a 1,200-year tree-ring record for the region (Griffin and Anchukaitis 2014).

The current drought offers numerous examples of what climate change responses may look like. In February 2014, for the first time, the state and federal agencies...
water projects set water allocations to zero because of low water supplies (CDWR 2014a). In 2015, drought measures for the first time included curtailments of pre-1914 water rights (SWRCB 2015a). In 2015, the State of California and the U.S. Bureau of Reclamation jointly petitioned the State Water Quality Control Board to temporarily modify Delta water quality standards (SWRCB 2015b). The U.S. Bureau of Reclamation drafted a Shasta Temperature Management Plan to guide use of the limited cold-water pool available in Shasta Reservoir to protect temperature-sensitive Chinook Salmon eggs during late summer (https://www.usbr.gov/mp/drought/docs/shasta-temp-mgmt-plan-key-components-06-18-15.pdf). The California Department of Water Resources built a $28 million emergency temporary barrier in West False River to try to protect the interior Delta from encroaching ocean salinity caused by low freshwater outflows (http://www.water.ca.gov/news/newsreleases/2015/052915.pdf). These are all examples of how the Delta’s operational and infrastructure frameworks may be modified with increasing frequency and increasing desperation in response to conditions caused—or exacerbated—by climate change. It is imperative that plans for protecting the Delta evaluate all trade-offs and opportunities, with the aim of being sufficient to meet the coming challenges and robust enough to accommodate large uncertainties that will not disappear.

**FISHERIES, HABITATS AND ECOSYSTEM EFFECTS**

Though the effects of climate change on the Delta ecosystem are expected to be profound, their exact nature is difficult to predict. This is partly because ecosystems comprise many species, each of which will respond to changes in the physical environment in its own way, affecting food web cycles (Brown et al. 2016a) and other ecological processes. The unpredictability also exists because ecosystem responses to climate change will depend on decisions about restoration and management that are being made now and in the future. That is, climate change will have very different effects on a future Delta with massive tunnels to protect export water qualities vs. a future Delta with freshwater throughflows aggressively managed to repel salt. Both futures would have winners and losers, but not the same winners and losers.

Generally, however, gradual changes in average environmental conditions are unlikely to be the largest challenges to the Delta’s organisms until those averages exceed organismic tolerances. It is much more likely that the extreme events attending those gradually deteriorating baselines will be the most challenging for biological systems for a long time to come. For example, a heat wave associated with a drought occurred in 2014, and 95% of naturally spawned winter-run Chinook Salmon eggs and fry died because the temperatures of releases from Shasta Reservoir releases exceeded their tolerance (NMFS 2015). A similar event also occurred in 2015 (http://www.sacbee.com/news/state/california/water-and-drought/article41684160.html).

Next we consider several of the expected effects of climate change on Delta species from a factor-by-factor perspective. We also consider some upstream ecological effects. Just as upstream processes affect what occurs in the Delta hydrologically, upstream processes also have important effects on in-Delta species.

In a natural system, the most obvious effects of SLR on ecosystems would be at the land–water interface, particularly in tidal marshes. As sea level rises, tidal marshes can respond in two ways. A tidal marsh might respond to SLR with increased sediment trapping and accumulation of organic material (peat building), allowing the elevation of the marsh plain to follow along with SLR, and thus maintaining a marsh–open water elevations differential similar to the historical difference. However, importantly, tidal marshes also might encroach on terrestrial habitat as the water level rises. Essentially, the marsh might extend landward with the deeper portions “drowning” and converting to other habitat types, such as a mudflat or subtidal habitat. Effects on the aquatic organisms could be minimal since they would be able to find suitable habitat conditions by moving short distances landward. However, in today’s landscape, few tidal wetlands remain, and many of those that remain cannot move landward because of the presence of levees and other hard infrastructures. Under these circumstances, SLR must be accommodated primarily by accumulation of sediment and organic material that raises the
marsh levels in place. A number of models have been applied to this problem with results that depend on the models and assumptions used (Stralberg et al. 2011; Swanson et al. 2013). Some modeled marshes keep pace with SLR, though others cannot, depending on assumed rates of SLR, amounts of sediments in the water column, and rates of organic detritus accumulation. More research is needed.

A spatially and temporally varying salinity gradient is a defining feature of the estuary’s waters. Estuarine organisms are adapted to geographically variable salinity fields that change on tidal, seasonal, annual, and longer time-scales. The most mobile organisms can simply move to remain within their preferred salinity ranges. Less mobile organisms, such as benthic invertebrates (e.g., clams), can adapt to fluctuating salinity through dispersal of eggs and larvae that can colonize new areas of appropriate habitat. In the estuary, Feyrer et al. (2015) identified five salinity guilds of fishes, ranging from freshwater to saltwater guilds. Salinity intrusion can also affect terrestrial, emergent, submerged, and floating vegetation, and other organisms.

Under natural conditions, these various species communities might respond to changing salinity conditions by simply moving (via colonization of appropriate habitats within a new salinity regime). However, salinity changes affect spatial extents, locations, and abundances of species. Moyle and Bennett (2008) have argued that management-induced reductions of variability in the Delta’s salinity fields have contributed to declines in native species, changing the Delta from a naturally variable estuarine system that supports native fishes to a reservoir-like freshwater system that favors invasive submerged aquatic plants (i.e., *Egeria densa*) and fishes such as largemouth bass and other centrarchids. Freshwater releases to prevent saltwater intrusion in the summer and fall now result in salinity gradients that historically would have been typical of extreme drought in all but the wettest years. Climate change-induced reductions in late-season water availability will make such salinity conditions even more common (Brown et al. 2013, 2014; Feyrer et al. 2010).

Overall, many of the invasive species present in the Delta are better adapted to warm temperatures and low inflows than are native species (Kiernan et al. 2012; Moyle et al. 2013, 2016). Rising water temperature will be one of the most significant climate-change stressors in the Delta. Ficklin et al. (2013) examined the effects of climate change on Sierra Nevada streams and found that spring and summer water temperatures are likely to increase from 1 °C to 5.5 °C, depending on location. Biota in sub-basins with the greatest warming are more likely to be adversely affected. Within the Delta, statistical modeling of water temperatures by Wagner et al. (2011) has projected that water temperature will likely become stressful for Delta Smelt through much of their range during the summer, and will likely change the timing of important events in their life history, such as spawning time (Brown et al. 2013). Warmer temperatures in the fall combined with earlier spawning would severely limit the time available for adult Delta Smelt to mature, with unknown consequences for the reproductive success (Brown et al. 2016b) of this bellwether species that is already on the verge of extinction (Moyle et al. 2016).

Water management actions taken to support upstream fisheries will also alter conditions in the Delta. Warmer inflows and enhanced floods and droughts are likely to adversely affect the cold-water pools of large reservoirs that support downstream Chinook Salmon, Steelhead, and Sturgeon fisheries. Several modeling studies have indicated that management of salmonids below dams and diversions will become more difficult as climate change proceeds (Yates et al. 2008; Cloern et al. 2011; Thompson et al. 2012; Null et al. 2013). These challenges are real and serious, as demonstrated by the recent mortality of federally listed winter-run Chinook Salmon below Shasta Dam (described earlier).

Although potentially disastrous in many ways, future levee failures might ultimately be of some benefit for some aquatic organisms because more aquatic habitat would be created. Many Delta “islands” are completely surrounded by levees that hold Delta waters away from their interiors, wherein land surfaces are well below the water levels outside the levees (Deverel et al. 2016). Once levees are breached and the interiors flooded, the flooding of these low-lying islands is often permanent. The benefits or damages from this flooding will vary with the species
being considered, the location and specifics of the levee failure, and the type and physical attributes of the habitat created. For example, Liberty Island, flooded in 1998, provides habitat for Delta Smelt because it has not been extensively invaded by *Egeria densa* or *Corbicula* to date, the water remains turbid, and the habitat is accessible to native species (Lehman 2010, 2015). In contrast, the flooded Mildred Island of the southern Delta has been extensively invaded by *Egeria densa* around its perimeter, supporting mainly invasive fish species (Grimaldo et al. 2012). The interior of the flooded island is too deep for *Egeria densa*, and pelagic production is relatively high; however, dense *Corbicula* in the outflow channels rapidly deplete exported chlorophyll–a, greatly reducing the benefit of primary production there to adjacent habitats (Lucas et al. 2002; Lopez et al. 2006). Flooded islands in warmer areas might well be ideal habitat for harmful algal blooms (see Fong et al., submitted). Depending on the size and location of newly flooded areas, there may be largely unexpected effects on the hydrodynamics of the entire Delta with unknown effects on the ecosystem.

Flooding in the late winter and early spring tends to benefit native fishes, particularly Splittail and Chinook Salmon (Perry et al. 2016), if floodplains remain inundated for a sufficient time (Sommer et al. 2004; Moyle et al. 2007; Jeffres et al. 2008; Moyle et al. 2016). This early flooding is important because native species tend to reproduce at cooler temperatures than many invasive species (Moyle et al. 2013). If inundations recede before water temperature increases much, reproduction of exotic species will be less successful. Conversely, droughts tend to favor exotic species because they yield fewer floodplain inundations and thus less opportunity for natives to reproduce in isolation from exotic species.

All of the above factors will be changing at the same time, and all of the communities and species will be responding with their respective individual strengths and vulnerabilities as best as each can throughout their respective life cycles. Given all the moving parts, our ability to predict in advance how climate change will affect Delta ecosystems and interact with human efforts to maintain desired ecosystem services is extremely limited. Most assuredly, there will be many surprises that require flexibility in our management systems. However, some changes we can expect. Success of habitat protection and restoration projects will require them to be designed to accommodate SLR, or to evolve gracefully into other desired habitat types as SLR proceeds. The entire life cycles of organisms of interest will need to be considered if we are to anticipate ecological effects of climate changes and attendant salinity and water temperature responses. Specifically, management that increases salinity and hydrodynamic variability in the Delta is likely to be an important tool for improving conditions for native fishes, but we need to understand far better than we do now which variations are beneficial, and how all the moving parts will interact if we are to use this tool successfully.

**THE WAY FORWARD**

We have called out earlier many specific knowledge gaps. More generally, global climate change is a “new” stressor that will influence many different climate, hydrologic, and ecosystem variables in the Delta system. Climate change will influence variables everywhere in the Delta’s catchment, but not in the same way everywhere. Initially, this century, effects will arise mostly through enhanced extreme events. In response to this inter-meshed complex of challenges, making use of the assets we have to avoid dire outcomes will require integrated monitoring systems, integrated modeling approaches, and integrated assessments of vulnerabilities and options, as well as adaptive and adaptable decision-making processes. Models of the many complex and interacting subsystems that comprise the Delta will need to be better developed to provide more realistic and reliable guidance for planning and management of the overall Delta system. The long-standing Delta Science Program-funded Computational Assessments of Scenarios of Change in the Delta Ecosystem (CASCaDE) program is one example of how such a modeling integration across scientific fields might look (e.g., Cloern et al. 2011). Greater life-cycle and end-to-end understanding of processes and responses, whether biological or technological, is needed. That is, such integrations and attention to the extremes have not always been the norm in the past.
We posed three questions concerning “how important is this event or change” in the “Introduction.” Answering these questions in the Delta, and anticipating cascading and potentially unexpected consequences of climatic events and of our responses to those events, will require a new generation of models and observations that cut across the scientific disciplines that connect as many of the parts of the Delta system, from mountain ridges to coastal ocean with all the varied landscapes in between. Meeting this requirement will depend on sustained research and observations (Dettinger and Culberson 2008), as well as considerable investment in developing the best reconstructions (through all means available) of past climates and climate effects as a baseline for the challenges and changes to come. These actions can reduce many uncertainties and help to avoid some unintended and unanticipated consequences of managing the Delta in a time of climate change. However, the uncertainties associated with climate change in the Delta will not disappear in time to allow precise outcomes to be predicted or planned for. Instead, we will know most precisely what the climate changes and effects will be as they emerge (or afterwards), and management of the Delta needs to accommodate this limitation with an urgency commensurate with what we do know or expect, and with a flexibility borne of the humble recognition of what we won’t know until later.

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ABSTRACT

Many contaminants in the California Bay–Delta (Bay–Delta) exceed regulatory standards, affect aquatic species, and potentially affect human health. Recent studies provide multiple lines of evidence that contaminants affect species of concern in the Bay–Delta (e.g., the decline of several important fish species referred to as the “Pelagic Organism Decline” or POD). Contaminants occur as dynamic complex mixtures and exert effects at multiple levels of biological organization. Multiple chemicals impair processes at cellular and physiological levels (measured as growth, development, and behavior abnormalities), and when viability and reproductive output are affected, populations are affected. As an important example, the population decline of the endangered Delta Smelt (Hypomesus transpacificus) is significantly associated with multiple stressors, including insecticide use. New analyses presented in this paper show significant correlations between pyrethroid use and declining abundance of POD fish species. Water sampled from the Bay–Delta causes multiple deleterious effects in fish, and Delta Smelt collected from the Bay–Delta exhibit contaminant effects. Fish prey items are also affected by contaminants; this may have an indirect effect on their populations. Co-occurrence with thermal changes or disease can exacerbate contaminant effects. Contaminants also pose threats to human health via consumption of fish and shellfish, drinking water, and contact recreation, in particular, mercury, cyanobacteria toxins, disinfection byproducts, pathogens, pesticides, and pharmaceuticals and personal care products. The role of contaminants in the decline of Bay–Delta species is difficult to accurately assess in a complex, dynamic system. However, tools and approaches are available to evaluate contaminant effects on Bay–Delta species, and separate the effects of multiple stressors. Integrated monitoring and focused mechanistic studies are instrumental for addressing management needs. Effect and risk assessments should be conducted for different species across multiple life stages, with emphasis on early life stages of high-priority Bay–Delta species.
The term contaminant refers broadly to a large number of substances from distributed and diverse sources that include urban and agricultural runoff, treated industrial and municipal wastewater, atmospheric deposition, and chemicals applied directly to surface waters for invasive plant and pest control.

The San Francisco Bay and Sacramento–San Joaquin Delta (Bay–Delta) has been identified as impaired for aquatic life by several specific contaminants on the Environmental Protection Agency 2010 List of Impaired Water Bodies (SWRCB 2010). The 2010 list of contaminants includes metals (copper, cadmium, mercury, and zinc), pesticides (chlordane, chlorpyrifos, DDE, DDT, diazinon, dieldrin, organophosphate insecticides, and toxaphene), and chlorinated compounds (dioxins, furans, and polychlorinated biphenyls [PCBs]). The Delta is also listed for sediment toxicity and unknown toxicity. An unknown toxicity listing results from toxicity being detected in lab or field studies, but not yet being linked to a specific chemical. Since the 2010 list was adopted, additional contaminants of concern have been identified including additional pesticides, flame retardants, nutrients, naturally occurring toxins, micro-plastics (e.g., from synthetic clothing), and pharmaceuticals and personal care products (PPCPs). Essential elements (e.g., selenium) and nutrients, when outside the beneficial ranges, may negatively affect organism or community health. A legacy of contaminants in the Bay–Delta, such as persistent organic chemicals and mercury, can, in addition to affecting aquatic life, accumulate through the food web, leading to health risks for humans and wildlife.

Although many contaminants have been identified as impairing aquatic life, it is unknown how many other contaminants may exert toxic effects. Compared to other biotic and abiotic factors that cause aquatic ecosystem degradation worldwide, the role of contaminants is often under-estimated because of a lack of comprehensive, quantitative, and effect-based analyses (Stehle and Schulz 2015). Contaminants affect populations and communities at concentrations detected in the Delta (Hasenbein et al. 2015c), but first exert their effects at the organism level by altering gene expression, physiological processes, and behavior. Historically, contaminant assessments focused predominantly on acute effects, but sublethal toxic effects can occur at exposure levels far below the concentrations that cause lethality. This does not imply that acute effects are no longer observed, but simply that there is greater awareness of the consequences of sublethal effects. Over the past decade, multiple lines of evidence demonstrate that contaminants, either singly or as mixtures, directly affect the health of Bay–Delta species (Table 1). These studies have provided much information about the risk of exposure to contaminants, and have also highlighted important knowledge gaps, including the significance of combined effects of chemical and other biotic and abiotic stressors.

Water quality standards are generally designed to be protective of 95% of aquatic life (i.e., of species for which we have toxicity data). Contaminant monitoring, coupled with toxicity testing that uses standard test species and methods, are also used as reliable indicators of “instream” threats to aquatic organisms (Grothe 1996; De Vlaming and Norberg–King 2000). In the past, contaminants were identified as impairing aquatic life primarily when chemical concentrations detected in Bay–Delta waters, sediment, or biota exceeded known water quality standards or caused toxicity. Although these tools are highly predictive of instream effects, they need to be paired with additional contaminant effect studies of resident or migratory species, whenever the abundance of these key species is linked to multiple stressors. It is noteworthy that when these standard regulatory tools were applied to the Bay–Delta and its tributaries, and the identified contaminants were examined in studies of resident species, effects were consistently confirmed.

The topic of Bay–Delta contaminants is broad, and by necessity a synthesis must focus on a subset of available information. In this paper we summarize new information which has become available since the State of Bay–Delta Science 2008 was published (Luoma et al. 2008). This synthesis emphasizes four topics:

1. The application of a weight-of-evidence approach to improve our ability to assess contaminant effects on Bay–Delta fish species that are experiencing significant population declines.

2. A synthesis of knowledge on known and emerging contaminants (not addressed in the weight-of-evidence examples).

3. An update on human health risks through fish consumption and drinking water from the Delta.

4. Recommendations on science priorities to address the current challenges in precisely defining the role of contaminants as one of many stressors.

APPLICATION OF A WEIGHT OF EVIDENCE APPROACH — CONTAMINANT EFFECTS ON DECLINING BAY–DELTA FISH SPECIES

One outstanding management challenge is that the role of contaminants in the decline of several Bay–Delta native and migratory species is not well understood. Even though the ecological effects of contaminants have been incorporated into Bay–Delta conceptual models, they have not been effectively quantified in terms of population declines. In recent years, a number of expert panels were convened to evaluate multiple stressors involved in the decline of Bay–Delta species. These expert panels identified contaminants as a stressor that warrants extensive investigation (Johnson et al. 2010; Mount et al. 2012; NRC 2013; IEP MAST 2015; Luoma et al. 2015), yet specific contaminants were not linked to the decline. Although several contaminants occur above their regulatory threshold concentration, we do not clearly understand the effect of these known contaminants on Bay–Delta species declines. Unknown contaminants could also be having an effect. Unequivocal identification of a specific contaminant as a cause of a species decline will continue to be a challenge, but recent research has generated multiple lines of evidence, which, when considered together, can be used to generate a weight of evidence that is more conclusive in identifying contaminants as an influential factor.

Quantifying the role of contaminants in observed Bay–Delta fish declines requires multiple approaches. For each species in decline, individual studies, or lines of evidence, can be synthesized to see if the weight of evidence supports that a contaminant is influencing species abundance. Some individual lines of evidence may determine that there is potential risk, but are clearly not definitive. For example, contaminants may be detected in the Bay–Delta at concentrations that exceed water quality standards, but these may not be adversely affecting the abundance of a particular species. Other lines of evidence may show species effects, but this is not enough to demonstrate that a contaminant is affecting the population as a whole. However, as we obtain multiple lines of evidence, it becomes more likely that effects are occurring, and that management action—or at least intensified studies—should be initiated. Potential lines of evidence include:

• major risk factors, such as:
  - declining species abundance not fully explained by other stressors
  - a statistical relationship between fish abundance and contaminant use
  - contaminants detected in fish habitats at levels of concern

• significant organism effects, such as:
  - effects detected in Bay–Delta waters or sediment on surrogate species
  - effects detected in Bay–Delta waters or sediment on the species of concern
  - effects detected in field-collected organisms
  - effects detected in laboratory and mesocosm studies conducted at contaminant levels detected in the Bay–Delta

• indirect effects of a contaminant, for example, on the food supply of a species known to be food limited

This synthesis focuses on the research conducted since 2005 that used field, mesocosm, and laboratory studies to evaluate the effects of contaminants on declining fish species including Delta Smelt (Hypomesus transpacificus), Longfin Smelt (Spirinchus thaleichthys), Threadfin Shad (Dorosoma petenense), juvenile Striped Bass (Morone saxatilis) and salmonids. We present available lines
Table 1  Sublethal effect-based assessments of contaminant impacts of contaminants present in the Bay-Delta

<table>
<thead>
<tr>
<th>DELTA SPECIFIC SUBLETHAL EFFECTS STUDIES FROM 2005 TO PRESENT</th>
<th>General Stress</th>
<th>Immune System</th>
<th>Osmoregulation</th>
<th>Nervous system</th>
<th>Muscular system</th>
<th>Endocrine system (reproductive)</th>
<th>Tissue effects / Histopathology</th>
<th>Low Nutritional Status</th>
<th>Development</th>
<th>Defoliates</th>
<th>Behavioral</th>
<th>Reproductive Output</th>
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<tr>
<td>Pyrethroid insecticides</td>
<td>Ch1-3</td>
<td>Sb4, Sp5</td>
<td>Ds9, Is9</td>
<td>Ds3, R1, Fm10,11</td>
<td>Ds7, Fm10,11</td>
<td>Fm11, Is12,15, R16, Jm17, Sp5, Sp5, Ds5,7</td>
<td>Sp5</td>
<td>Sp5</td>
<td>Ds4, R16,</td>
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<td>Sp5, Is8,12,14</td>
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<td>Is16,17, Ch22, R122, Sp5, Ds6,7, Is8</td>
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<td>Persistent Organic Pollutants</td>
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<td>Ammonia and Ammonium</td>
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<td>Microcystin</td>
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<td>Delta Water Samples (Laboratory)</td>
<td>Sb33, T13</td>
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<td></td>
</tr>
</tbody>
</table>

Key: Fish species: Ch = Chinook Salmon; Sb = Striped Bass; Sp = Sacramento Splittail; Ds = Delta Smelt; Is = Inland Silversides; Rt = Rainbow Trout; Fm = Fathead Minnow; Jm = Japanese Medaka; Lmb = Largemouth Bass; Ts = Threadfin Shad; Invertebrate species: Ha = Hyalella azteca; Cd = Chironomus dilutus; Ce = Ceriodaphia dubia; Pf = Pseudodiaptomus forbesii; Eg = Eudiaptomus gracilis.


See reference information for sources listed beginning on page 30.
of evidence, which, when taken together, provide a weight of evidence that contaminants, in combination with other stressors, have negatively affected Delta Smelt. There is also evidence of contaminant effects on salmon. Less is known about the precise role of contaminants on other Bay–Delta fish declines, but a synthesis of recent studies suggests the potential for contaminant effects, and identifies critical gaps in our knowledge (Table 1).

**Evidence of Declining Species Abundance Not Fully Explained by Other Stressors**

A step-decline in abundance of Delta Smelt, Longfin Smelt, Threadfin Shad, and Striped Bass (young-of-year) was observed in the early 2000s (Thomson et al. 2010). Declining numbers of estuary-dependent fish species such as Chinook Salmon (*Oncorhynchus tshawytscha*), Steelhead Trout (*O. mykiss*), and Green Sturgeon (*A公交车er medirostris*), as well as invertebrates—in particular calanoid copepods and mysids—and desirable primary producers have also been documented (Winder and Jassby 2011). Contaminants have been concomitantly detected at concentrations that can elicit acute and chronic effects (e.g., reduced growth, reduced reproduction) in related species. Studies of sublethal effects on Bay–Delta species began more recently. Although these investigations noted decreases in growth rates and fecundity, many did not fully assess the role of contaminants. The general conclusion of the recent investigations is that multiple stressors were involved (FLaSH; Brown et al. 2014).

Monitoring the abundance and distribution of aquatic species (e.g., algae, macroinvertebrates, fish) can identify changes in populations, but analyses of multiple stressors are required in order to understand why species abundance and composition fluctuates. Such analyses must include contaminants and their effects on organisms and the ecosystem if their role is to be defined.

**Example of Correlations Between Declining Fish Abundance and Increasing Insecticide Use**

Several investigations have conceptualized but not quantified the role of contaminants in Bay–Delta fish declines (Brooks et al. 2012; Scholz et al. 2012). A new analysis, presented here as a representative example of changing pesticide use, indicates that pyrethroid insecticide use in the Delta is strongly correlated with fish abundances (Figure 1).

**Why Pyrethroids as an Example?** Although pyrethroids are classified as neurotoxicants, they have a number of additional effects that can be detrimental to fish, including endocrine disruption and growth and development alteration (Table 1). Pyrethroids are the fourth most-used group of insecticides worldwide (Hénault–Ethier 2015; Brander et al. 2016a), and their use has increased steadily since 1979, while the use of organophosphate insecticides (OPs) (e.g., diazinon chlorpyrifos) has declined since their peak in the early 1990s (Figure 2). Concentrations of pyrethroids were predicted to increase markedly in waters tributary to the Bay–Delta starting in 2000 (Jorgenson et al. 2013). The California Department of Pesticide Regulation's (CDPR) Surface Water Protection Program has ranked pyrethroids as high priority for monitoring because they have high potentials to cause surface water toxicity from urban and agricultural uses (Luo et al. 2014). The Central Valley Regional Water Quality Control Board (CVRWQCB)2 is currently establishing a control program for pyrethroid insecticides to protect Bay–Delta watershed aquatic life. The hydrophobic nature and strong binding affinity of pyrethroids to particulate matter were thought to reduce or prevent their runoff into surface waters; however, studies have shown that runoff from areas treated with pyrethroids was more toxic to fish than runoff from areas treated with OPs (Werner et al. 2002; Jiang et al. 2016).

Associations between pyrethroid use (agricultural and professional urban application) in the six counties of the Delta (http://www.cdpr.ca.gov/docs/pur/purmain.htm) and fish abundance indices for the Pelagic Organism Decline (POD) species from Interagency Ecological Program (IEP) Fall Midwater Trawl (FMWT; 1978 to 2014) suggest that pyrethroids may be a contributor to fish population declines in the Delta (Table 2). The use of six pyrethroids in the Delta region (permethrin, esfenvalerate/fenvalerate, bifenthrin, cyfluthrin, cypermethrin, and lambda-
Figure 1  Ordinary least squares regression fitted line plots and 95% confidence intervals of IEP FMWT species abundance indices (autocorrelation corrected) as a function of annual pyrethroid pesticide use in the counties of the Delta from 1978–2014.
cyhalothrin) explains 24% to 73% of the interannual variability in FMWT species abundance indices (Figure 1). See Appendix A for further details of the statistical analyses.

Understanding interactions between contaminant exposure and environmental factors, and their effect on populations, will likely require comprehensive analyses of disparate datasets. For example, Bailey et al. (1994) used flow-adjusted Striped Bass abundance indices and pesticide concentrations to correlate rice agricultural discharges to their decline. Conceptually, flow variability is also hypothesized to be the most important decadal-scale driver in reducing the ecological resilience of the Bay–Delta which contributed to the POD (Baxter et al. 2010). More recently a synthesis report established quantitative relationships between Delta Smelt abundance indices and habitat location (X2; location at salinity of 2 PSU) and recruitment variables (IEP MAST 2015). Herein, models were developed to compare the relative influence of flow and pyrethroid use on species abundance (Table 3). The model that explained the greatest amount of variability in an abundance index was for Longfin

Table 2 Pearson’s r, Spearman’s rho, and Kendall’s tau correlation coefficients measuring associations between IEP FMWT abundance indices and pyrethroid use in the Delta.

<table>
<thead>
<tr>
<th>Transformed IEP FMWT abundance indices</th>
<th>Lambda transformation</th>
<th>Pearson’s r</th>
<th>Spearman’s rho</th>
<th>Kendall’s tau-a</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Lambda transformation</td>
<td>Correlation coefficient</td>
<td>p-value</td>
<td>Correlation coefficient</td>
</tr>
<tr>
<td>Longfin Smelt</td>
<td>0</td>
<td>−0.67</td>
<td>&lt;0.0001</td>
<td>−0.65</td>
</tr>
<tr>
<td>Delta Smelt</td>
<td>0</td>
<td>−0.67</td>
<td>&lt;0.0001</td>
<td>−0.58</td>
</tr>
<tr>
<td>Sacramento Splittail</td>
<td>0</td>
<td>−0.61</td>
<td>&lt;0.0001</td>
<td>−0.64</td>
</tr>
<tr>
<td>American Shad</td>
<td>−0.5</td>
<td>0.62</td>
<td>&lt;0.0001</td>
<td>0.51</td>
</tr>
<tr>
<td>Threadfin Shad</td>
<td>0</td>
<td>−0.48</td>
<td>&lt;0.01</td>
<td>−0.39</td>
</tr>
<tr>
<td>Striped Bass</td>
<td>0</td>
<td>−0.86</td>
<td>&lt;0.0001</td>
<td>−0.89</td>
</tr>
</tbody>
</table>
Smelt ($R^2$-adj = 0.79, $p < 0.001$, F-test), with average Delta inflow explaining slightly more variability than pyrethroid use. This is consistent with Baxter et al. (2010), where Longfin Smelt abundance trends were best explained by seasonal Delta outflow. For the other species, pyrethroid use was a more important determinant of abundance variability than flow. Overall, this analysis suggests that pyrethroid use may have played a comparable role to other stressors in the POD. The strong correlation observed suggests an urgent need for further analyses to assess whether this is a causal relationship, especially because research has highlighted deleterious effects at multiple levels of biological organization, in multiple species (Table 1).

These analyses are only a cursory evaluation of multiple-factor effects on fish species abundances; mechanistic studies are needed to evaluate the biological significance of the correlation. Recovery efforts for Bay–Delta aquatic species need to include analyses of contaminant effects in conjunction with other pertinent factors. Additional data and analyses are needed to better understand the interactions between habitat and biology. For example, Baxter et al. (2010) hypothesized that salinity, landscape, temperature, turbidity, nutrients, and harvest, in addition to flow and contaminants, were long-term drivers of the POD. In addition, Thomson et al. (Thomson et al. 2010) (2010) indicated that water clarity and Delta flow characteristics were drivers of Bay–Delta fish species abundances; however, the researchers did not evaluate possible contaminant effects. The ability to explain changes in fish abundance by pyrethroid use in the Delta may be altered by the inclusion of other important factors. For example, Bailey et al. (1994) showed that the discharge of rice agriculture pesticides, primarily carbamates, likely contributed greatly to earlier Striped Bass declines. However, carbamate insecticides have been replaced primarily with pyrethroids, and young-of-year Striped Bass abundances have not recovered. Monitoring, special studies, and models are needed to link the processes that occur from contaminant sources to a resulting species decline (e.g., pesticide runoff, bioenergetic costs of exposure to contaminants, decreases in food availability, or reductions in fecundity or fish survival).

It must be noted however, that pyrethroids are only one class of a multitude of pesticides detected in Bay–Delta waters, and are presented here as an example. Even though regulation of some pesticides has decreased their effect, replacement products can be similarly problematic. Effective attenuation measures are needed in order to reduce the entry of contaminants into California waterways, which will increase the probability of species recovery.

### Table 3  Predictor variables and summary statistics for multiple linear regression models explaining IEP FMWT species abundance variability from 1978–2014.

<table>
<thead>
<tr>
<th>Species</th>
<th>$R^2$-adj</th>
<th>Predictor variables</th>
<th>Partial coefficients</th>
<th>95% CI</th>
<th>$p$-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Longfin Smelt</td>
<td>0.79</td>
<td>Pyrethroid use</td>
<td>-1.097</td>
<td>(-1.422, -0.771)</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Delta Inflow</td>
<td>-1.228</td>
<td>(-1.533, -0.902)</td>
<td></td>
</tr>
<tr>
<td>Striped Bass</td>
<td>0.77</td>
<td>Pyrethroid use</td>
<td>-1.235</td>
<td>(-1.494, -0.977)</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Delta Inflow</td>
<td>-0.336</td>
<td>(-0.622, -0.050)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Delta Exports</td>
<td>-0.322</td>
<td>(-0.602, -0.042)</td>
<td></td>
</tr>
<tr>
<td>Delta Smelt</td>
<td>0.37</td>
<td>Pyrethroid use</td>
<td>-1.01</td>
<td>(-1.567, -0.452)</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Delta Exports</td>
<td>0.422</td>
<td>(0.107, 0.737)</td>
<td></td>
</tr>
<tr>
<td>American Shad</td>
<td>0.41</td>
<td>Pyrethroid use</td>
<td>0.00973</td>
<td>(0.00472, 0.01473)</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Delta Exports</td>
<td>-0.00464</td>
<td>(-0.00778, -0.00150)</td>
<td></td>
</tr>
<tr>
<td>Sacramento Splittail</td>
<td>0.54</td>
<td>Pyrethroid use</td>
<td>-0.744</td>
<td>(-1.089, -0.400)</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Delta Inflow</td>
<td>-0.651</td>
<td>(-0.995, -0.307)</td>
<td></td>
</tr>
</tbody>
</table>
Evidence of Contaminants Detected in Fish Habitats at Levels of Concern

Monitoring entities and research studies have detected multiple contaminants occurring simultaneously in Delta water samples (Ensminger et al. 2013; Orlando et al. 2013, 2014). Multiple pesticides are continuously detected in the two primary tributaries to the Delta. For example, 27 pesticides or degradation products were detected in Sacramento River samples, and the average number of pesticides per sample was six. In San Joaquin River samples, 26 pesticides or degradation products were detected, and the average number detected per sample was 9. Water quality objectives do not exist for most of these compounds. However, these were targeted chemical analyses, and hundreds of compounds have been detected in individual Delta water samples using other non-targeted techniques (2016 in-person conversation with T. Young, J. Orlando and R. Connon, unreferenced, see “Notes”).

Organisms are exposed to a dynamic mixture of contaminants (e.g., introduction of new chemicals, varied use patterns). Although pesticides generally have a seasonal pattern, PPCPs are continuously introduced into the environment (Deanovic et al. 2014, unreferenced, see “Notes”; Biales et al. 2015). This raises concern because exposure to chemical mixtures has shown adverse effects on aquatic organisms at concentrations at which no observable adverse effects occur for single constituents (Carvalho et al. 2014; Cedergreen 2014), and little is known about potential synergistic, antagonistic, or additive effects of exposure to contaminant mixtures.

Pyrethroids have been found in sediments of agricultural and urban waters upstream of the Delta at concentrations that are acutely toxic to numerous benthic and epibenthic macroinvertebrates (Amweg et al. 2005; Holmes et al. 2008; Weston et al. 2008; Weston et al. 2015b).

Copper is present throughout the Delta at concentrations known to cause adverse effects. The copper threshold established for enclosed bays and estuaries is 4.8 μg L⁻¹. Dissolved copper concentrations up to 4.64 and 4.93 μg L⁻¹ were detected in freshwater water samples from Cache–Lindsey Slough and Rough and Ready Island, respectively, and elevated dissolved copper concentrations of 37.2 and 58.9 μg L⁻¹ have also been detected at Suisun Bay and Carquinez Strait, respectively (Werner et al. 2010a). A recent study (Sommers et al. 2016) indicated that although the effect of copper on salmon olfaction is reduced in brackish and saline waters, copper can still cause avoidance behavior at environmentally relevant concentrations.

In a study conducted to evaluate the effects of pesticides, trace metals, and PPCPs present in Sacramento River samples, the most frequently detected substances were pharmaceuticals. PPCPs comprised 51% of the detected analytes: trace metals and pesticides comprised 28% and 21% of the analytes, respectively (Deanovic et al. 2014, unreferenced, see “Notes”). Other studies also found a high incidence of pharmaceuticals in Sacramento River water (Biales et al. 2015). Guo et al. (2010) completed a source, fate, and transport study that included 11 sampling sites in the Delta associated with the State Water Project (SWP). Forty-nine chemicals were detected, many at concentrations above those that elicit adverse effects.

Evidence of Effects on Surrogate Species Detected in Delta Waters or Sediment

Multiple studies have found sublethal, lethal, chronic, and acute toxicity of Bay–Delta waters to model test species of phytoplankton, invertebrates, and vertebrates (Jassby et al. 2003; Johnson et al. 2010; Blaser et al. 2011; Brooks et al. 2012; Scholz et al. 2012). Multiple-species studies that evaluated Bay–Delta ambient water samples, or conducted in situ exposures (referenced in Table 1), have repeatedly identified a broad set of mechanistic, systemic (immune, neurological, endocrine), histopathological (tissue damage), and whole-organism effects (e.g., growth, development, deformities). Endocrine-disruptive effects have been measured in samples from Sacramento River tributaries and in the Bay–Delta (Schlenk et al. 2012; Brander et al. 2013; Cole et al. 2016).

Although identification of chemical classes responsible for these effects is limited, because of the complexities resulting from mixtures of multiple chemicals, several studies (see Table 1) have highlighted pyrethroid pesticides as responsible
for toxicity, endocrine disruption, and neurological impairments in both fish and their prey (Brander et al. 2013, 2016b; Hasenbein et al. 2015c; Jeffries et al. 2015b; Weston et al. 2015a).

**Evidence of Effects on Delta Species of Concern**

**Delta Smelt:** Several recent studies on Delta Smelt support that contaminants are a significant stressor. Ambient water samples collected in the Delta contained sufficient bioavailable compounds that in laboratory exposures Delta Smelt exhibited decreases in growth and altered development compared to fish exposed to control water. Specific effects were detected in gene expression associated with their immune system and muscular system (Hasenbein et al. 2014). Apoptosis and necrosis were also observed (Connon et al. 2012). Delta Smelt collected from the estuary were observed to have tissue-level effects, demonstrated through histopathology, which was associated with decreased growth (Hammock et al. 2015). These lines of evidence support that contaminants are bioavailable in Bay–Delta waters at concentrations that are affecting Delta Smelt.

In addition to studies that rely on ambient water or field-collected fish, studies have examined the effect of specific contaminants on cultured smelt. Pyrethroids have been documented to cause general stress responses and effects on the immune, nervous, muscular and osmoregulatory systems (Jeffries et al. 2015b). Decreased growth, abnormal development, and altered behavior have been detected from exposure to pyrethroids at levels detected in the Delta (Connon et al. 2009; Jeffries et al. 2015b). However, pyrethroids are not the only contaminant thought to be negatively affecting Delta Smelt. Smelt exposed to copper in the lab at levels detected in the Delta exhibited effects on their immune, nervous, and muscular systems (Connon et al. 2011a). Growth and development, as well as behavior, were also negatively affected, the latter raising concern with about the effects on homing ability (Connon et al. 2011a). Ammonium induced effects similar to those observed after exposure to pyrethroids and metals (Connon et al. 2011b), affecting immune- and muscular-system functioning, as well as development and behavior (Connon et al. 2011b; Hasenbein et al. 2014). The weight of evidence supports the hypothesis that multiple contaminants are having a direct effect on Delta Smelt by affecting multiple levels of biological organization.

Contaminants can cause indirect effects by causing trophic cascades that affect population dynamics, food webs, community structure, and the Bay–Delta ecosystem as a whole (Fleeger et al. 2003; Johnston and Roberts 2009). Indirect effects of contaminants can also result in changes in nutrient and oxygen dynamics, altering phytoplankton and zooplankton communities (Brown et al. 2016; Moyle et al. 2016). Delta Smelt are known to be food limited (Hammock et al. 2015), and trend data shows reductions in prey availability over time (Hennessy 2011). Ambient water samples (Werner et al. 2008; Werner et al. 2010b), mesocosm studies (Hasenbein et al. 2015c), and laboratory studies (Hasenbein et al. 2015a; Hasenbein et al. 2015b) suggest that smelt prey organisms exhibit effects when exposed to ambient Delta waters or control waters amended with Delta-relevant concentrations of contaminants. Not only do pesticides reduce prey abundance, but pyrethroids have been shown to transfer to fish through prey (Muggelberg et al. 2016).

Taken together, these lines of evidence support the hypothesis that Delta Smelt are exposed to toxic levels of multiple contaminants found in the Delta. Delta Smelt populations are in decline, and the decline is significantly associated with multiple stressors. Delta water samples caused deleterious effects in Delta Smelt, and Delta Smelt collected from the Delta exhibit contaminant effects. Laboratory studies show that multiple levels of biological organization are affected by multiple chemicals, and negative organism-level responses (measured as growth, development, and behavior abnormalities) have been observed. In addition to these direct effects, food availability for Delta Smelt may be reduced by contaminants, and this may indirectly affect the population. A huge data gap is that limited studies examine the relationships between contaminants and specific responses by Delta Smelt, particularly during the early embryo-to-larval life stages.

**Other POD Species.** Much less is known about the role of contaminants in the decline of the other POD species. Although Longfin Smelt, and Threadfin Shad
declines were significantly correlated with pyrethroid use, a large data gap exists since no contaminant exposure studies have been conducted on these species. The Striped Bass decline is also significantly correlated with pyrethroid use, but exposure studies with Striped Bass have focused on polycyclic aromatic hydrocarbons (PAHs) and polyhalogenated aromatic hydrocarbons (PHAHs), including PCBs and dioxins. Exposure and bioaccumulation of polybrominated diphenyl ethers (PBDEs), PCBs, and legacy pesticides can result in these contaminants being transferred maternally from females to eggs, which affects egg size, fecundity, brain and liver development, impaired growth, and survival (Ostrach et al. 2008). These compounds are widespread in the Bay–Delta, and studies have demonstrated their effect on fish health and development (Spearow et al. 2011; Durieux et al. 2012).

Salmon abundance is declining, and several important stressors have been identified. Both pesticides and copper exposure can affect fish migration and orientation. The most commonly observed links with these behavioral disruptions include cholinesterase (ChE) inhibition, altered brain neurotransmitter levels, and sensory deprivation (Scott and Sloman 2004). Scholz et al. (2000) also concluded that exposures to low concentrations of diazinon likely increased the straying of the adult hatchery Chinook salmon over the control group. Furthermore, juvenile salmonids exposed to pesticides during development may fail to imprint to their natal waters, which can lead to increased adult straying (NMFS 2009). Chlorpyrifos exposure directly affects the nervous system (Baldwin et al. 2009) and the olfactory system (Maryoung et al. 2015). There is evidence that behavioral effects of pesticides affect salmon populations in other ecosystems. For example, cypermethrin prevented male Atlantic salmon from detecting and responding to the reproduction-priming pheromone prostaglandin, which is released by ovulating females (Moore and Waring 2001). Copper concentrations of 2 μg L⁻¹ significantly affect the olfactory system in juvenile salmonids (see video³, Sandahl et al. 2007; Grossman 2016), increasing predation risk and impairing osmotic homeostasis (Grosell et al. 2002). This is of concern because dissolved copper concentrations detected in water samples from Cache–Lindsey Slough and Rough and Ready Island were above threshold. Also, copper causes cholinesterase (ChE) inhibition, so its effects may be additive when present with OPs. In addition to behavioral effects, OPs have been shown to affect the immune system in Chinook Salmon, increasing their susceptibility to disease (Eder et al. 2008). Histopathological abnormalities and reduced growth have been reported for both invertebrate and fish species (Baldwin et al. 2009; Hasenbein et al. 2015b). Impaired gonadal or thyroid hormone levels in salmon have also been observed (Scott and Sloman 2004). Perhaps the most important point provided by existing studies is that the behavioral effects of contaminants on salmon should be investigated further. Contaminants could be the proximate cause of salmon mortality that is currently attributed to disease and predation (Grossman 2016).

In summary, this section illustrates how using a weight-of-evidence approach can facilitate a better understanding of the potential for contaminants to be influencing factors in the declining abundance of Bay–Delta fish species. Multiple studies support the potential importance of contaminants affecting Delta Smelt. Salmon studies are sufficient to prompt more study on behavioral effects. The paucity of research on the other POD species illustrates that this approach can identify critical data gaps.

Evidence that Contaminant Exposure Leads to Population Effects

Numerous contaminants detected throughout the Bay–Delta can affect the overall health of individuals, leading to behavior and reproductive impairment that translate to alterations in population dynamics. Global decreases in aquatic biodiversity have been associated with increases in pesticide contamination; for example, macroinvertebrate family richness is reduced by ~30% even when pesticide concentrations were within regulatory thresholds (Stehle and Schulz 2015). Models predict that a 6% reduction in length and 16% in mass would result in a >50% reduction in spawner abundance over 20 years in Chinook salmon (Baldwin et al. 2009). Population growth rates of Delta species are affected by exposure to contaminants (Brooks et al. 2012). Recent research

conducted in the Delta determined that contaminants sourced via water treatment plants disrupt endocrine system function in Inland Silversides (*Menidia beryllina*), resulting in alterations to gonado-somatic indices (GSI), testicular necrosis, and biased sex ratios (Brander et al. 2013). Adult Inland Silversides exposed to low, Delta-relevant concentrations (0.5 ng L\(^{-1}\)) of the pyrethroid bifenthrin had a significant reduction in reproductive output and biased sex ratios (fertilized eggs per female; Brander et al. 2013, 2016b). Risks associated with exposure to endocrine-disrupting compounds (EDCs), have been extensively reviewed (Bortone and Davis 1994; Tyler et al. 1998; Brander et al. 2013, 2016a). Risk of extinction in isolated populations has recently been associated with biased sex ratios (Grayson et al. 2014). Studies have demonstrated that exposures to neurotoxic insecticides (Baldwin et al. 2009) affect populations of multiple species and their community structures (Hasenbein et al. 2015c; Orlinskiy et al. 2015). More globally, Feist et al. (2011) describe how urban runoff contaminants in the U.S. Pacific Northwest caused up to 90% mortality of pre-spawning Coho Salmon (*Oncorhynchus kisutch*), thus, severely affecting population numbers through reductions in recruitment.

**A SYNTHESIS OF SPECIFIC BAY–DELT A CONTAMINANTS**

**Metals and Metalloids**

Contaminants that biomagnify pose major risks to aquatic species at higher trophic levels in the Bay–Delta; including fish, birds, and mammals. Two bioaccumulative contaminants, selenium and mercury, were among the high-priority water quality issues described by Luoma et al. (2008).

Linares-Casenave et al. (2015) reported high selenium concentrations in tissues of older, reproductively mature female White Sturgeon in the Bay–Delta. This is concerning because selenium-enriched yolk in sturgeon eggs can cause developmental defects as well as mortality of embryos and yolk-sac larvae, affecting recruitment. Similarly, kidney lesions, reduced growth and deformities have been observed in Sacramento Splittail fed a selenium-based diet (Deng et al. 2007). White Sturgeon and Splittail populations are exposed to high levels of selenium through their diet, notably from *Corbula amurensis*, the invasive overbite clam (Feyrer et al. 2003; Stewart et al. 2013). In recent years, the average selenium concentrations in White Sturgeon from the bay have been below the threshold (11.3 μg–g\(^{-1}\) dry weight in muscle) established to prevent effects on Sturgeon reproduction as part of the North Bay Total Maximum Daily Load (TMDL; Baginska 2015). Extensive research has been conducted to support development of the TMDL and revised criteria for the Bay, including an ecosystem-scale selenium model, a model of transport, fate, and uptake into the food web, and additional monitoring and review (Chen et al. 2012; Presser and Luoma 2013). Long-term trend monitoring by the Regional Monitoring Program for Water Quality in San Francisco Bay (Bay RMP) and the USGS also continues (SFEI 2013; Stewart et al. 2013). The Bay RMP is performing pilot studies to evaluate non-lethal methods of monitoring selenium in sturgeon muscle that would increase the number of samples available to track long-term trends. A more precise understanding of the concentrations that elicit deleterious effects would be valuable.

Mercury, in the highly toxic form of methylmercury, can pose major risks to both aquatic and terrestrial species at higher trophic levels in the Bay–Delta, including fish, birds, and mammals. Methylmercury exposure is a significant concern for special-status bird species, including the federally endangered Ridgway’s Rail (*Rallus obsoletus*) and California Least Tern (*Sternula antillarum browni*). Forster’s Tern (*Sterna fosteri*) is the species at greatest risk: Ackerman et al. (2014) found that 79% of eggs from this species were above a high-risk threshold of 1 μg g\(^{-1}\) fresh wet weight. The control plans for mercury in both the Bay and the Delta (SFBRWQCB 2006; Wood et al. 2010) include a concentration target for prey fish to protect piscivorous birds. Average concentrations of methylmercury in species of concern are also commonly in the range known to affect biochemical processes, damage cells and tissue, and reduce reproduction in fish; particularly in peripheral areas of the Delta (Sandheinrich and Wiener 2011). For example, Gehriger et al. (2012) presented histopathological evidence of immunosuppression in juvenile largemouth bass from methylmercury contaminated areas in the Delta.
Copper exposure was shown to elicit general stress responses, affect the immune, nervous and muscular systems, and impair growth, development, and behavior in Bay–Delta fishes (Geist et al. 2007; Connon et al. 2011a; Gehringer et al. 2012) and invertebrates (Callinan–Hoffmann et al. 2012). Copper toxicity and accumulation in fishes differs between species, between freshwater and saltwater environments, as well as among the specific organs that are affected (Blanchard and Grosell 2006). While increased salinity is generally considered as protective against loss of olfactory function from dissolved copper, the presence of sub-lethal levels of dissolved copper altered the behavior of juvenile Chinook Salmon by inducing an avoidance response in both freshwater and seawater (Sommers et al. 2016). Further, species-specific evaluations are needed at higher salinity sites in order to determine potential effects on species of concern.

**Persistent Organic Pollutants**

PAHs and PHAHs including PCBs and dioxins are widespread in the Bay–Delta. Numerous studies have demonstrated their effect on fish health and development (Spearow et al. 2011; Durieux et al. 2012). Two contaminants of concern have received significant attention in the last few years: polybrominated diphenyl ethers (PBDEs) and perfluorooctane sulfonate (PFOS), however, effect-based assessments are lacking for Bay–Delta species.

PBDE flame retardants have been detected in Bay fish and wildlife since the 1990s. High detections spurred voluntary reductions and a California ban that took effect in 2006, which resulted in reduced concentrations in bivalves, fish, and bird eggs (Sutton et al. 2015). On the other hand, concern has increased regarding PFOS. PFOS is widely used as a stain repellent for textiles, furniture, and carpets; as a surfactant in fire-fighting foams and metal finishing processes; as an ingredient in the production of fluoropolymers; and as an insecticide. PFOS has been detected globally, including in San Francisco Bay birds and seals (2016 in-person conversation between M. Sedlak and J. Davis, unreferenced, see “Notes”). Bird eggs collected in the southern portion of the bay in 2006 and 2009 contained levels of PFOS above a threshold (1 μg g⁻¹ wet weight) that affects offspring survival in birds. Fortunately, more recent results (2012) are 70% lower than prior levels and well below the threshold. However, PFOS concentrations in seals do not show similar declines. PBDEs and PFOS have not been monitored in the Delta.

**Ammonia and Ammonium**

Toxicity to aquatic organisms is primarily attributable to the un-ionized form, ammonia. Ammonium is increasingly converted into ammonia as pH rises. Ammonium can enhance cell membrane permeability increasing its toxicity to species, and their susceptibility to the synergistic effects of multi-contaminant exposures (Connon et al. 2011b; Hasenbein et al. 2014). Freshwater mussels, for example, are highly sensitive to increased ammonia concentrations (USEPA 2013) and total ammonia nitrogen concentrations detected in the Sacramento River, downstream of Hood, are at levels potentially toxic to *Pseudodiaptomus forbesi* (Teh et al. 2011).

**Pesticides**

Multiple insecticides, fungicides, herbicides, and antibacterials are commonly detected throughout the Bay–Delta. The CDPR reports that 13,084 pesticide formulations are registered in the state, including 1,040 registered active ingredients, and >60% of those pesticide products are applied in the Central Valley (Pesticide Use Report; http://www.cdpr.ca.gov). OPs, pyrethroids and phenylpyrazoles (e.g., fipronil) are of greatest concern with regard to fish and zooplankton health. However, there is also concern over the use of herbicides, and their potential effect on the food web. In 2014, over 12,000 L of herbicide (formulation Fluridone) were applied to over 2,600 acres to control water hyacinth in the Delta, but this does not include the herbicides applied for other aquatic weeds such as *Egeria densa*, spongeplant (*Limnobium laevigatum*), and curly leaf pondweed (*Potamogeton crispus*).

Organophosphate insecticide registrants agreed to phase out urban sales in 2001. Many researchers have called attention to the decreased use of OPs (Oros and Werner 2005; Kuivila and Hladik 2008; Johnson et al. 2010) with a move toward increased use of pyrethroids, phenypyrazoles, and neonicotinoids (e.g.,
imidacloprid), yet studies continue to detect OPs in Bay–Delta waters (Ensminger et al. 2013; Weston et al. 2015a).

Pyrethroid insecticide exposure has negative effects on hormonal and neurological development or reproductive output. At low concentrations (ng L\(^{-1}\)) they act as EDCs through blocking, mimicking, or synergizing endogenous hormones (Brander et al. 2016a). Pyrethroid metabolites are reported to have even greater estrogenic activity than parent compounds (DeGroot and Brander 2014) as well as a significant occurrence (>20%) of deformities in offspring of exposed adults (2016 in-person conversation with B. Decourten, see “Notes”). Pyrethroids such as bifenthrin and permethrin are present in the Bay–Delta at concentrations that alter numerous metabolic processes, which result in protein degradation (Werner and Moran 2008; Vandenberg et al., 2012). They also alter osmoregulation capacity (Riar et al. 2013; Jeffries et al. 2015b), nervous- and muscular-system functions (Connon et al. 2009), and behavior (Beggel et al. 2010), as well as result in reduced growth and development (Geist et al. 2007; Forsgren et al. 2013; Riar et al. 2013). Larval deformities and histopathological abnormalities have also been reported for Sacramento Splittail exposed to pyrethroids (Teh et al. 2005).

Fipronil and its degradation products have been detected in urban creeks and tributaries to the Bay–Delta. It is a broad-spectrum insecticide used on pets and for structural pest control. Its occurrence in the Bay–Delta poses new challenges because degradation products have been shown to be more toxic than their parent compounds (Weston and Lydy 2014). Delta-relevant studies on Fipronil are limited to acute toxicity evaluations on invertebrates. However, effects to nervous and muscular systems, decreased swimming performance, and potential endocrine disrupting properties have been demonstrated with Fathead Minnow that require further investigation (Beggel et al. 2010, 2012). Fipronil concentrations in bay sediment have raised concern for potential effects on benthic invertebrates (SFEI 2013).

Pharmaceuticals and Personal Care Products

Pharmaceuticals and personal care products (PPCPs), such as pain and blood pressure medications, antidepressants, antibiotics, antibacterials, and micro-plastics are used in large quantities and can enter waterways from indoor drains (e.g., excretion, improper disposal, bathing) through wastewater treatment plants. Over the past 2 decades, the U.S. Food and Drug Administration (FDA) center for Drug Evaluation and Research listed 2,817 drugs (medications containing an active substance; excluding personal care products), with 45 new drugs approved in 2015 alone.\(^4\) A large number of PPCPs have been detected in the Bay–Delta (SFEI 2013; Deanovic et al. 2014, unreferenced, see “Notes”; Biales et al. 2015), but their relative contributions to the contaminant load in the Bay–Delta have not been comprehensively assessed. PPCPs are not regulated in surface and drinking water, and the risks to aquatic life are largely unknown.

Ibuprofen is one of the most commonly detected pharmaceuticals in wastewater effluent worldwide (Fent et al. 2006). Exposure to ibuprofen was shown to affect the expression of genes involved in oxidative stress, aerobic respiration, immune function, and osmoregulation, as well as skeletal development in Inland Silversides (Jeffries et al. 2015a). Although concentrations of pharmaceuticals detected in water samples are relatively low compared to those that elicit responses in fish, the concentration detected in water samples may be misleading because un-metabolized ibuprofen levels in wild fish plasma and bile for example, can be 100 to 1000 times higher, respectively, than those found in surrounding water samples (Brozinski et al. 2013).

Triclosan is an antibacterial widely used in consumer products (e.g., toothpaste, hand soaps), and is also found in wastewater effluent. Triclosan has been shown to negatively affect swimming behavior in fish by disrupting the excitation–contraction processes of skeletal muscle (Fritsch et al. 2013). Triclosan can readily accumulate in fish muscle and brain, thus posing a risk to Bay–Delta fishes.

Cyanotoxins

Naturally occurring cyanobacteria (blue–green algae) are common in ecosystems worldwide, and can produce toxins that negatively affect the ecosystem.

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much like chemical contaminants. At least 46 species of cyanobacteria have been shown to produce toxins that pose health risks to humans and wildlife (Carey et al. 2012; Lehman et al. 2013). Common cyanobacteria genera can produce a suite of toxins, such as hepatotoxins (microcysts), cytotoxins (cylindrospermopsin), neurotoxins (anatoxin-a, antillatoxin, saxitoxins), and dermatoxins (lyngbyatoxins). Research in the Bay–Delta has primarily focused on Microcystis spp. blooms, which were first recorded in 1999, and occur annually (Kurobe et al. 2013; Lehman et al. 2013). Microcystis can thrive in highly altered and nutrient-rich habitats. They can produce microcystin, which may promote liver cancer in humans and wildlife (Ibelings and Havens 2008). Studies conducted on Threadfin Shad and Sacramento Splittail demonstrated that consumption of Microcystis adversely affected their nutritional status, and resulted in severe liver and gonadal lesions (Acuña et al. 2012a, 2012b). Cyanobacterial blooms can further affect wildlife by lowering dissolved oxygen concentrations, and can also cause taste and odor problems in drinking water (Paerl et al. 2001). In a synthesis of Delta data from 2004 to 2008, Lehman et al. (2013) found that dry years resulted in higher microcystin concentrations in the water and mesoplankton tissues. Miller et al. (2010) concluded that microcystin-contaminated freshwater that entered Monterey Bay was bioaccumulated by bivalves, resulting in the death of 21 southern sea otters. This raises concern for risks to sturgeon and other species that consume bivalves, including humans.

**UPDATE ON HUMAN HEALTH CONCERNS**

**Contaminant Exposure through Fish Consumption**

Contamination of sport fish by two legacy contaminants, mercury and PCBs, is a high priority management issue in the Bay–Delta. Concentrations of mercury (in its highly toxic form, methylmercury) and PCBs are high enough that the California Office of Environmental Health Hazard Assessment has issued advisories that cover the entire estuary. These advisories have been updated and expanded in recent years (Gassel et al. 2007, 2011). Methylmercury is a major driver of advisories in the Bay–Delta, and though PCBs are also a major driver for Bay advisories, they are lesser studied in the Delta. Risks to human health and the resulting consumption advisories are an important part of the bay mercury (SFBRWQCB 2006) and Delta methylmercury (Wood et al. 2010) control plans as well as the bay PCBs TMDL (SFBRWQCB, 2008).

Spatial patterns in sport fish methylmercury in the Delta have been fairly well-characterized, but very few data are available on inter-annual variation and long-term trends. Existing time-series at specific Delta locations are far from ideal, because of inconsistencies in sampling location, sample sizes, size ranges, and species, but the data do suggest consistent spatial patterns over time, with relatively high concentrations at the sites around the periphery of the Delta, and lower concentrations in the Central Delta (Davis et al. 2000, 2008, 2013; Melwani et al. 2009). Time-series based on repeated, directly comparable measurements are needed to rigorously characterize long-term trends that would serve as a performance measure to evaluate the effectiveness of the methylmercury TMDL.

Striped Bass is an important indicator species for methylmercury contamination throughout the Bay–Delta because of their high trophic position, consequentially high bioaccumulation, and popularity for consumption. Striped Bass from the Bay–Delta have the highest average mercury concentration in US estuaries (Davis et al. 2012). A historical dataset exists for Striped Bass in the bay, allowing trends over 39 years from 1971 to 2009 (Figure 3) to be evaluated. Concentrations measured in recent years are not significantly different from those measured in the early 1970s.

**Cyanobacteria**

The magnitude, frequency, and distribution of cyanobacterial blooms are expected to increase in the Bay–Delta as a result of climate change (Carey et al. 2012) and excessive discharge of nutrients. Humans can be exposed to cyanotoxins from recreational contact as well as consumption of fish and shellfish, and drinking water; effects range from skin irritation to death. In the Bay–Delta, incidents of human health effects are poorly captured, but the World Health Organization has documented effects from all over
Contaminants in Drinking Water

Contaminant effects on drinking water are very different from contaminant effects on the ecosystem because treated water complies with drinking water standards before its distribution. However, contaminants can affect the ability to meet standards, treatment requirements, aesthetic qualities of drinking water, water management programs, and drinking water provision costs. The Delta is an important source of drinking water for 25 million people in the San Francisco Bay area, Central Valley, and southern California. Priority drinking water quality issues described by Luoma et al. (2008) included salinity, bromide and natural organic matter, and remain important issues for Delta drinking water supplies. New drinking water regulations adopted or proposed by the USEPA and the SWRCB’s Division of Drinking Water are driving additional monitoring and science needs for the Delta. Additional contaminants receiving attention include pathogens, cyanotoxins, PPCPs, and emerging disinfection by-products and their precursors.

Organic carbon reacts with drinking water treatment disinfectants to form carcinogenic byproducts, which are regulated at low levels to protect public health. Salinity affects the aesthetic qualities of drinking water and creates water-management challenges for blending, groundwater recharge, and water reuse. Bromide, a component of salinity, also contributes to the formation of carcinogenic disinfection byproducts during the water treatment process. Levels of these constituents in Delta water vary significantly because of hydrology and water project operations. Organic carbon concentrations increase during wet weather because of higher loading from stormwater, agricultural, forested land, or other runoff sources. An analysis of Delta water quality at Banks Pumping Plant in wet and dry years indicated that salinity and bromide concentrations were significantly higher in dry years, especially in the summer months (Archibald Consulting 2012) when seawater intrusion into the Delta is more pronounced (CDWR 2010). The recent severe drought (2012 to 2015) resulted in Delta water quality conditions that posed water...
management and water-treatment challenges for drinking water agencies. Wei–Hsiang et al. (2010) evaluated the potential long-term changes in Delta water quality from sea level rise and levee failures of subsided western Delta islands, and found that under these long-term scenarios increases in salinity and bromide concentrations would significantly increase treatment costs for Delta water supplies.

Pathogens, such as Cryptosporidium spp. and Giardia spp., enter surface waters from animal and human sources, and can cause illness if consumed. Controlling the formation of disinfection by-products, from high concentrations of organic carbon and bromide in the source water, while implementing disinfection to inactivate pathogens, is an ongoing operational challenge for drinking water agencies that treat Delta water supplies. In recent years, the USEPA and SWRCB’s Division of Drinking Water adopted more stringent drinking water regulations for disinfection by-products and pathogens, and the regulatory requirements are tied to concentrations of organic carbon and pathogens in the source water. Under these regulations, drinking water agencies are required to monitor their source water for Cryptosporidium to determine if more advanced treatment is needed to reduce pathogen levels in drinking water supplies.

In July 2013, the CVRWQCB adopted a new Drinking Water Policy for Surface Waters of the Sacramento–San Joaquin Delta and Upstream Tributaries (CVRWQCB 2013). The policy includes a narrative (i.e., non-numeric) water quality objective and monitoring requirements for Cryptosporidium and Giardia. The Delta Regional Monitoring Program (Delta RMP)5 initiated a 2-year pathogen monitoring study in April 2015 to characterize ambient concentrations in the Delta concurrently with Cryptosporidium monitoring performed by drinking water agencies at their treatment plant intakes. The coordinated pathogen monitoring study is expected to characterize ambient background conditions and potential sources of pathogens in the Delta to fill an important data gap.

Cyanotoxin levels in the Delta are also a concern for drinking water, and cyanotoxins such as microcystin and cylindrospermopsin in drinking water may require regulation. The USEPA has proposed to include cyanotoxins on the draft fourth Drinking Water Contaminant Candidate List (CCL 4) and as part of the Unregulated Contaminant Monitoring Rule. The purpose of these programs is to identify priority contaminants that need further study and regulation, and to require public water systems to monitor for suspected drinking water contaminants. In June 2015, to protect public health, the USEPA published non-regulatory Drinking Water Health Advisories for young children and adults that provide technical guidance on microcystin and cylindrospermopsin.

The California Department of Water Resources (CDWR) initiated microcystin monitoring in the SWP facilities in 2006. Between 2006 and 2012, dissolved microcystin was detected in 1% of samples, but in 2013, the CDWR changed laboratories and methods. The new method measures total microcystin, including the microcystin contained in cyanobacteria cells, resulting in more frequent detections at more locations and at higher concentrations. Consequently, microcystin has been frequently detected throughout the SWP at levels that exceed the health advisories. From July 2013 to August 2015, most samples from Clifton Court Forebay, in the south Delta, exceeded the microcystin health advisory that protects young children. Some drinking water treatment facilities can remove microcystin, but cyanotoxins are still a concern for drinking water supplies from the Delta.

Emerging water quality concerns for Delta drinking water supplies include PPCPs and additional disinfection byproducts of public health concern, such as nitrosamines, which may be human carcinogens. Guo et al. (2010) conducted a source, fate, and transport study of EDCs and PPCPs that included several sampling locations in the Delta. The six most frequently detected contaminants were carbamazepine, diuron, sulfamethoxazole, caffeine, primidone, and tris (2-chloroethyl) phosphate (TCEP), with the highest concentrations occurring at sites downstream of wastewater treatment plant discharges. The investigators concluded there is no evidence of human health risk from low levels of PPCPs detected; however, more toxicological studies are needed. Lee et al. (2015) conducted a monitoring study in the Delta to evaluate the presence and

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5 http://www.waterboards.ca.gov/centralvalley/water_issues/delta_water_quality/comprehensive_monitoring_program/index.shtml
source of N-nitrosodimethylamine (NDMA), other nitroamines, and their precursors. They found that wastewater treatment plants are an important source of NDMA precursors in the Delta, because they found higher levels of NDMA precursors downstream from the Sacramento Regional Wastewater Treatment Plant and the Stockton Regional Wastewater Control Facility. NDMA was not detected in river samples, likely because of dilution and photolysis.

In summary, contaminants continue to be detected in the Bay–Delta at concentrations that exceed regulatory standards, potentially causing detrimental effects. For example, mercury still occurs at levels that pose a risk to human health through consumption of contaminated fish, and Delta drinking water supplies are threatened by contaminant issues such as cyanotoxins, pharmaceuticals and personal care products, and new carcinogenic disinfection byproducts.

SCIENCE CHALLENGES, GAPS, AND RECOMMENDATIONS

Lacking the Right Monitoring Endpoints

Contaminant monitoring in the Bay–Delta, particularly in the Delta, falls short of answering priority questions to adequately inform water quality management. We cannot assume that reduced acute toxicity from one chemical or chemical class protects beneficial uses. Today’s management questions are deeper and more far-reaching. Are contaminants delaying salmon from moving upstream when they need to? Are contaminants limiting productivity of nutritious fish food? And if so, is this constraining them to areas of greater risk for entrainment, predation, or other hazard? Are our control methods for aquatic vegetation and other invasive species affecting our primary productivity of the beneficial species? Managers need to consider multiple needs for multiple resources, and the cascading effects of contaminants. With more advanced monitoring, more integrative synthesis, and better input from multidisciplinary teams, resource managers will be better equipped with the information they need to make decisions.

Extensive water quality monitoring in the bay, has made it one of the most thoroughly-monitored estuaries in the world. The Bay RMP began monitoring in 1993, and has succeeded in its aims so well that funding has grown and been sustained. High quality monitoring data and special studies from the Bay RMP have guided dozens of important decisions about water quality management in the bay (Trowbridge et al. 2016). Monitoring has also provided an essential performance measure to evaluate the success of management decisions in meeting water quality goals. Collaborative governance by diverse interests allows the Bay RMP to optimize the use of funds, and to adapt to stay relevant as the ecosystem changes, new issues emerge, and knowledge advances.

A Delta RMP has been established and monitoring began in 2015. This program should be supported in becoming a long-term, robust, and comprehensive monitoring program that informs regulatory measures and management decisions.

More Spatial and Temporal Coverage

Better spatio-temporal coverage is critical to understand how water management changes can affect contaminant transport, fate, and effects. Monitoring for the Bay–Delta should include sites that are upstream and in back sloughs where more toxicity has been exhibited, rather than in larger channels (Werner et al. 2008; Werner et al. 2010b; Markiewicz et al. 2012). This will facilitate identification of toxicity and sources. Models like the Co-Occurrence Pesticide Species Tool (Hoogeweg et al. 2012) should be used to select monitoring stations where the greatest risk is posed by the likely co-occurrence of pesticides and sensitive species.

Fixed stations, like the CDWR monitoring station on the Sacramento River at Hood, should be installed in key areas to facilitate a combination of real-time physicochemical and flow-through biological monitoring. Such stations offer a more controllable test environment, thereby enhancing linkages between laboratory- and field-based study results. To better understand how instream chemical concentrations and abiotic stressors affect multiple species, methods from multiple disciplines spanning levels of biological hierarchy could be employed simultaneously (Biales et al. 2015).
Monitoring and assessment for Delta drinking water supplies should be expanded to include PPCPs and implemented at sufficient spatio-temporal scales to inform water management and drinking water treatment operations.

Time-series for mercury in Delta sport fish, based on repeated, directly comparable measurements, are needed for the rigorous characterization of long-term trends that would serve as a performance measure to evaluate the effectiveness of the Delta methylmercury TMDL.

**Diversified Testing**

Integrative toxicological studies using multiple species of diverse sensitivities, in conjunction with non-target chemical analyses, can be used to evaluate the ecological effects of contaminants, including unknown compounds. Simply monitoring for chemicals and comparing them to benchmarks of individual chemicals has proven inadequate to assess the ecological effects of contaminants (Bispo et al. 1999). Targeted chemical analyses have been shown to under-estimate mixture toxicity by a factor of two to ten compared to non-targeted analyses (Moschet et al. 2014). Employing non-targeted analyses could greatly advance the understanding of contaminant effects in the ecosystem by proactively assessing waters for new chemicals without limiting them to known compounds. Monitoring of contaminant presence should be paired with monitoring of effects on relevant species using sublethal effect endpoints. For example, the use of *Ceriodaphnia dubia* was appropriate in the past because they are sensitive to OPs and carbamates. However, *C. dubia* are relatively insensitive to several replacement insecticides; therefore, more appropriate species should be included to evaluate ecological effects. In a summary of toxicity tests from the Central Valley, researchers found that larval fish tests showed a higher frequency of toxicity than either invertebrates or algae, even though insecticides were determined to be the main cause of toxicity (Markiewicz et al. 2012). Had monitoring only included invertebrates, these sublethal effects on fish would have gone undetected.

To monitor surface waters, a comprehensive set of effect-based tools should be further developed that include important species, endpoints, and mechanisms of toxicity, and which consists of a combination of whole organism and *in vitro* tests. These effect-based assays are essential to address effects of low-level mixtures and unknown contaminants. In addition, biological assays can be tailored to comprehensively assess water quality by simultaneously evaluating contaminant effects on multiple modes of action (Escher et al. 2008, 2014). Environmental assessments should include techniques that are able to link physiological, biochemical, and molecular endpoints to organismal health condition. Although the association of sublethal effects and reproductive output has been demonstrated in non-native and surrogate species, analogous responses for threatened Bay–Delta species need to be confirmed. Sublethal effect concentration studies, particularly on early life stages of Bay–Delta species with contaminants of concern, should be performed.

**More Synthesis, Analysis, and Use of Adaptive Management**

Increased synthesis and analysis of monitoring data would allow for better trend analysis as well as improved assessment of ecosystem health. Past syntheses and analyses have been infrequent or incomplete because most Delta contaminant evaluations were special studies rather than systematic or comprehensive monitoring programs. Periodic events can go undetected by rigidly scheduled environmental monitoring (Brooks et al. 2012), thus monitoring needs to adapt to changing use patterns and be focused to identify risk to aquatic organisms. Regular analysis and synthesis are required to adaptively manage monitoring so management needs, and spatial and temporal variability, can be addressed.

Adaptive management has been successfully applied to contaminants. The successful reduction of PBDEs in the bay food web discussed above is an excellent example of adaptive management. In another example, a program was initiated in 1983 to address fish toxicity and drinking water taste and odor concerns associated with rice pesticides. By 2003, the Department of Food and Agriculture, Central Valley Regional Water Quality Control Board (CVRWQCB), the CDPR, County Agricultural Commissioners, and rice growers successfully worked together to identify management practices, water quality standards,
monitoring requirements and risk evaluations (e.g., use changes), and a communication and coordination mechanism to ensure that management practices would meet the performance goals and would not result in adverse effects on water quality.

Contaminants risks from past and present land use should be considered during planning and execution of habitat restoration efforts. With tens of thousands of acres of habitat restoration planned for the Delta and Suisun Marsh, those efforts could easily be confounded by contaminants. Many contaminants are sediment-bound, so sediment disturbance could cause resuspension and increased bioavailability. Additionally, repeated wetting and drying could increase risk over continual wetting (e.g., mercury methylation). Because wetlands are zones of deposition, sediment-bound contaminants will accumulate and may reduce the productivity and effectiveness of restoration efforts.

Robust planning, monitoring, analysis, reporting, and adjustments with cooperative participation and communication among regulators, industry, and other stakeholders can lead to reduced risk and improved water quality. Use of performance-based goals focused on contaminant effects alongside detection would better represent ecosystem health and function.

Integrate Efforts

Monitoring efforts of the Bay and Delta RMPs, the IEP’s Environmental Monitoring Program (EMP), the CDWR’s Municipal Water Quality Investigations (MWQIs), the SWRCB’s Surface Water Ambient Monitoring Program (SWAMP), and the USGS’s National Water Quality Assessment Program should be better integrated with each other and special studies. Their integration would not only make monitoring in the Delta more efficient, but it would facilitate multi-disciplinary evaluation of data across programs to allow for a better understanding of how water quality affects multiple levels of biological organization. Use of real-time monitoring data could provide in-depth information to interpret in situ testing results for multiple species at key sites, which would significantly enhance Bay–Delta monitoring efforts.

CHALLENGES

Managing Multiple Contaminants

A challenge is that the Water Quality Control Plan for the Sacramento and San Joaquin River Basins calls for protection of water quality whether the toxicity is caused by a single substance or the interactive effect of multiple substances, yet control programs and regulatory tools have typically addressed single chemicals or classes of chemicals. The CVRWQCB Basin Plan addresses the need to consider cumulative effects, and the policy assumes potential additive toxicity when pollutants are known carcinogens, or manifest their toxic effects on the same organ systems or through similar mechanisms of action. However, the ability to evaluate such effects is limited by lack of sufficient data on mechanisms of action for many contaminants; therefore, chemical-specific criteria are often used, which are under-protective of aquatic populations. Chemical mixtures of compounds with unknown interactions, or those having no specific criteria, are not adequately addressed (Johnson et al. 2010; Brooks et al. 2012).

Consistent Resources

Resource managers need consistent financial support from state and federal entities to better address contaminants and water quality in the Bay–Delta. With all the gaps identified above, even with additional integration of existing programs, current contaminant monitoring programs and research are desperately underfunded.

CONCLUSION

Contaminant issues that were of concern in 2008 persist, and contaminants continue to be detected in ambient water samples at concentrations that cause detrimental effects. Enhanced monitoring in the Delta is a critical need. The limited, existing Delta contaminant monitoring is reactive—measuring what we know is of concern rather than proactively addressing new potential threats—while the use of pesticides, pharmaceuticals, and personal care products changes frequently, creating an ever-evolving cocktail of contaminants. Contaminants

6 http://www.waterboards.ca.gov/centralvalley/water_issues/basin_plans/
thus do not occur as single compounds, but rather as mixtures that can interact additively, synergistically, and antagonistically with each other as well as with water quality parameters and food web processes, creating variable exposures and effects in time and space. The intertwined multiple-stressor effects of contaminants and their effects on Bay–Delta species requires a dedicated research program.

Contaminants likely played a significant role in the POD, but the specific role of contaminants in the health of the ecosystem will not be adequately understood until relationships are identified and tested through comprehensive studies that also relate the presence of contaminants to biological responses. Important Bay–Delta species are in decline, and their reduced abundance cannot be fully explained by other stressors. Direct effects on surrogate and important species range from decreased disease resistance and altered swimming behavior to lethality; and indirect effects are likely occurring through the food web. Multiple contaminants are detected in Bay–Delta waters, particularly in areas known to once support important species. Quantification of correlative relationships points to the need to include contaminant effects in ecosystem evaluations. The weight of evidence therefore suggests that numerous contaminants detected throughout the Bay–Delta have detrimentally affected the Bay–Delta ecosystem.

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NOTES


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ABSTRACT
Increasing clarity of Delta waters, the emergence of harmful algal blooms, the proliferation of aquatic water weeds, and the altered food web of the Delta have brought nutrient dynamics to the forefront. This paper focuses on the sources of nutrients, the transformation and uptake of nutrients, and the links of nutrients to primary producers. The largest loads of nutrients to the Delta come from the Sacramento River with the San Joaquin River seasonally important, especially in the summer. Nutrient concentrations reflect riverine inputs in winter and internal biological processes during periods of lower flow with internal nitrogen losses within the Delta estimated at approximately 30\% annually. Light regime, grazing pressure, and nutrient availability influence rates of primary production at different times and locations within the Delta. The roles of the chemical form of dissolved inorganic nitrogen in growth rates of primary producers in the Delta and the structure of the open-water algal community are currently topics of much interest and considerable debate. Harmful algal blooms have been noted since the late 1990s, and the extent of invasive aquatic macrophytes (both submerged and free-floating forms) has increased especially during years of drought. Elevated nutrient loads must be considered in terms of their ability to support this excess biomass. Modern sensor technology and networks are now deployed that make high-frequency measurements of nitrate, ammonium, and phosphate. Data from such instruments allow a much more detailed assessment of the spatial and temporal dynamics of nutrients. Four fruitful directions for future research include utilizing continuous sensor data to estimate rates of primary production and ecosystem respiration, linking hydrodynamic models of the Delta with the transport and fate of dissolved nutrients, studying nutrient dynamics in various habitat types, and exploring the use of stable isotopes to trace the movement and fate of effluent-derived nutrients.

KEY WORDS
Sacramento–San Joaquin Delta, nutrients, primary production, ammonium, nitrate, *Microcystis*, macrophytes, nutrient sensors
INTRODUCTION

The paradigm concerning nutrients in California’s Sacramento–San Joaquin River Delta (Delta), developed over several decades of research, is that nutrient levels are high, nutrient limitation is rare, and factors other than nutrients regulate the rates of primary production (Jassby et al. 2002). Annual rates of phytoplankton primary production varied 5-fold between 1975 and 1995. Jassby et al. (2002) explained the strong annual variability as resulting from at least four processes: (1) the effects of invasive clams on phytoplankton biomass; (2) a long-term decrease in total suspended sediments that affect the light regime; (3) changes in river inflows to the Delta that reflect climate variability; and (4) an additional unknown pathway. The State of Bay–Delta Science 2008 (Healey et al. 2008) did not contain a chapter on nutrients because of this ruling paradigm, and there was little mention of nutrients in chapters on water quality and aquatic ecosystems.

Thoughts about the roles of nutrients in the Delta have changed considerably over the past decade. The emergence of blooms of the toxic cyanobacteria, *Microcystis aeruginosa* (*Microcystis*), starting about 1999, has generated questions about the sources of nutrients needed to sustain these blooms (Lehman et al. 2015). The proliferation of invasive aquatic macrophytes within the Delta, especially *Egeria densa* (Brazilian waterweed) and *Eichhornia crassipes* (water hyacinth), has also raised questions about the sources of nutrients that support summertime coverage of up to 15% of the Delta’s waterways by submerged and floating aquatic vegetation (Santos et al. 2009). The growing water clarity in the Delta (Schoellhamer et al. 2013; Hestir et al. 2013) contributes to the increasing role for primary producers in Delta food webs, and the interactions among primary producers, light, nutrients, and hydrodynamics are emerging as critical Delta research topics (Schoellhamer et al. 2016). The chemical forms and concentrations of inorganic nitrogen (i.e., ammonium and nitrate) and the stoichiometry of nitrogen and phosphorus (N:P ratios) are also hypothesized to affect rates of primary production, the composition of the primary producer communities, and the structure of the aquatic food web (Dugdale et al. 2007; Parker et al. 2012a; Glibert et al. 2016). Nutrients and their multiple roles in the Delta landscape have emerged as important topics for monitoring and research in a changing Delta.

This paper addresses the following topics relating to nutrients within the Delta:

- nutrient inputs into the Delta from tributary rivers and the San Francisco Bay,
- nutrient inputs, transport, and fate within the Delta,
- rates and controls on phytoplankton primary production and nutrient uptake,
- changing Delta clarity and the effects of this change on primary production and nutrient uptake,
- whether the form of inorganic nitrogen potentially affects the productivity of phytoplankton, and if the elemental stoichiometry of nitrogen and phosphorus affects algal community composition,
- interactions between nutrients and harmful algal blooms such as *Microcystis*,
- nutrients and invasive aquatic vegetation like *Eichhornia crassipes* (water hyacinth) and *Egeria densa* (Brazilian waterweed) in the Delta,
- technological advances that allow continuous measurement of certain soluble nutrients in Delta waters, and
- thoughts on research needs and directions for future studies regarding nutrients in the Delta.

This paper focuses on nitrogen, the better-studied nutrient within the Delta, and also considers phosphorus. Nutrient concentration and nutrient cycling pathways within the Delta are changing. An upgrade to the Stockton Regional Wastewater Control Facility (SRWCF) in 2007 changed the dominant form of effluent nitrogen from ammonium to nitrate, and a major plant upgrade of the Sacramento Regional County Sanitation District (Regional San) in 2021 will change both the load and form of nutrients discharged into the Delta. An overview of current knowledge about nutrient dynamics within the Delta is timely and relevant.
The Sacramento and San Joaquin rivers deliver the largest loads of nutrients to the Delta, with municipal and agricultural discharge contributing the bulk of these loads (Kratzer et al. 2011). Agricultural discharges include both irrigation return flows that transport fertilizer and soil-derived nutrients and discharge from dairies and feedlots. Although the Sacramento River delivers the largest nutrient loads to the Delta, loading from the San Joaquin River is particularly large, given its size. This is most evident in summer, when the San Joaquin contributes almost half the total nitrogen load to the Delta despite flows less than 20% of the Sacramento (Kratzer et al. 2011). However, water exports from the south Delta divert much of the San Joaquin River water, reducing the effect of these nutrients on the greater Delta (Schlegel and Domagalski 2015). Although San Francisco Bay and the ocean can contribute a small fraction of water to the western Delta, their contribution to the Delta nutrient supply is negligible (Novick et al. 2015). The Sacramento River, therefore, plays the largest role in supplying nutrients to the Delta.

Municipal discharge is also a significant nutrient source to the Delta. Publicly Owned Treatment Works (POTWs) are estimated to supply approximately 25% of the total nitrogen (TN) and 20% of the total phosphorus (TP) loads to the Delta, combining all upstream and in-Delta sources (Domagalski and Saleh 2015; Saleh and Domagalski 2015). The Sacramento Regional Wastewater Treatment Plant (SRWTP) contributes about 90% of the annual total ammonia load to the Delta with effluent discharge occurring in the Sacramento River at Freeport in the north Delta (Jassby 2008).

Concentrations and Trends

Trends in nutrient inputs to the Delta via the Sacramento and San Joaquin rivers have been assessed (Kratzer and Shelton 1998; Cloern 2001; Kratzer et al. 2004; Kratzer et al. 2011; Schlegel and Domagalski 2015). It is important to note that these compilations use as their furthest-downstream reporting station Freeport on the Sacramento River, and Vernalis on the San Joaquin River, which are upstream of the major wastewater treatment plants that serve the Sacramento region and Stockton, respectively.

In the Sacramento River, the annual mean nitrate concentration declined 20% from values near 10 \( \mu \text{M} \) between 1980 and 2000, to values below 8 \( \mu \text{M} \) in 2013 (Schlegel and Domagalski 2015). The flow-normalized trend in annual loading to the Delta, however, remained relatively constant. The annual mean concentration and mean loading of ammonium from rivers that enter the Delta were reduced by a factor greater than five in the 1970s, and both values have continued to decline modestly since (excluding the SRWTP). The annual mean concentrations of TP were almost halved from values above 3.5 \( \mu \text{M} \) in the 1970s to values near 2.0 \( \mu \text{M} \) in 2013, with somewhat lower ongoing declines in total annual loads (Schlegel and Domagalski 2015).

Unlike the concentrations in the Sacramento River, the annual mean nitrate concentrations in the San Joaquin River show no decline from the 1970s to 2013, maintaining a concentration near 80 \( \mu \text{M} \), with similarly little change in annual loads. Ammonium concentrations in the San Joaquin River over the same period, however, did show a monotonic decline from values above 14 \( \mu \text{M} \) to less than 3.5 \( \mu \text{M} \) in 2013, with similar declines in total annual loads. The annual mean TP concentration at Vernalis also declined from highs of over 8 \( \mu \text{M} \) in the 1970s, 1980s, and 1990s to values near 5 \( \mu \text{M} \) in 2013, with little appreciable change in annual loading to the Delta (Van Nieuwenhuyse 2007; Schlagel and Domagalski 2015).

Schlagel and Domagalski (2015) also noted a strong, discharge-dependent seasonality in nitrate and TP in both the Sacramento and San Joaquin rivers. The highest concentrations in nitrate were observed in the Sacramento River during high discharge in winter months. Although the highest concentrations also occurred in the San Joaquin River during winter, they were greatest at intermediate discharge values. Presumably, the nitrate is diluted in the San Joaquin River at high flows, but this effect is not observed in the Sacramento River, where landscape yields of nitrate can keep pace with increased discharge. In contrast, Schoellhamer et al. (2012) observed TP concentrations to be highest on average in the fall, for both the Sacramento and San Joaquin rivers,
Nutrient Sources within the Delta

Although nutrient loading to the Delta is large and responds to long-term and seasonal changes, nutrient loading within the Delta is thought to be relatively small and relatively constant, comprising loadings from the internal municipal sources (included above) and nutrients introduced through island drainage, which are thought to be balanced between water withdrawal and island discharge (Novick et al. 2015).

Nitrification—the biological transformation of ammonium into nitrate—plays an important role in the Delta because it represents an internal source of nitrate to the Delta even though it does not change TN (Damashek et al. 2016; Foe et al. 2010). As the ammonium the SRWTP discharges is transported down the distributary channels of the Sacramento River into the central Delta, the north Delta, and into the lower Sacramento River, the ammonium concentrations are observed to decline, accompanied by a concomitant increase in nitrate (Foe et al. 2010; Novick et al. 2015; Parker et al. 2012b), with wastewater ultimately being the major source of nitrate in the central Delta (Novick et al. 2015). The finding that nitrification of wastewater-derived ammonium is significant in determining Delta nitrate concentrations is supported by much available N isotope data (Novick et al. 2015), but it clearly is not the sole determinant.

Nutrient Concentration Trends and Seasonality in the Delta

Trends in nutrient concentrations in the Delta generally have been flat or downward since 1998, with nitrate declining in the Western Delta but little trend elsewhere (Novick et al. 2015). Novick et al. (2015) observed declines in ammonium concentration in the north, central and western Delta. They attributed the observed declines to management source control efforts because the declines run counter to the increasing population density and agricultural intensity of the Central Valley. During this period, phosphate generally remained flat or declined, following the longer-term declines Jassby (2008) reported, which are attributable to declines in POTW discharge of phosphate (Kratzer et al. 2011).

Landscape Yields

Saleh and Domagalski (2015) studied nitrogen sources and transport in the rivers and streams of California, including the major inputs to the Delta, using the SPAtially Referenced Regressions On Watershed (SPARROW) attributes modeling framework. They found that agricultural lands comprised the largest source (47%) of TN to the Delta in the Sacramento River drainage, with point sources (chiefly POTWs) accounting for 32%. In the San Joaquin River drainage, agricultural lands contributed a greater fraction of TN loading to the Delta (62%), with point sources concomitantly lower (19%). TP loading to the Delta was also investigated using the same modeling approach (Domagalski and Saleh 2015), and this study suggested that agricultural lands contributed 65% and 58% of the total loading from the Sacramento River and San Joaquin River drainages, respectively. Point sources respectively accounted for 21% and 15% of TP loadings for these two drainages. The SPARROW model results also showed evidence of nitrogen retention within the aquatic system in small and medium-sized streams (Mulholland et al. 2008), but little retention in rivers. Retention for both P and N in reservoirs was also found to be small, largely because the reservoirs are located above most cultivated lands (Domagalski and Saleh 2015; Saleh and Domagalski 2015).

Nutrient Inputs, Transport and Losses Within the Delta

Spatial and temporal variation of nutrient concentrations within the Delta are driven by multiple influences: long-term changes in climatic conditions and anthropogenic inputs, and seasonal and climatic variation in flow and temperature as well as in internal biological processes (Novick et al. 2015). Over shorter time-scales, nutrient concentrations within the Delta can vary markedly because of source-related, tidal or biological processes (Pellerin et al. 2009).
Substantial seasonal and spatial variability in nutrient concentrations is observed in the Delta, driven by a combination of the location of the various inputs as well as internal processes (Figure 1). Concentrations of TN, dissolved inorganic nitrogen (DIN), nitrate, and ammonium are all generally higher in winter months, with lower values in summer—an expression of the multiple controls on concentration. As noted above, the loads of nitrate and phosphate delivered to the Delta are highest during periods of high runoff, and decline during lower flows (Kratzer et al. 2011). These inflows both positively and negatively regulate initial concentration values; winter concentrations in the Delta largely resemble river inputs and spatial variation is muted.

During lower-flow periods in the spring and summer, internal processes become more dominant in determining nutrient distributions. During this period, a characteristic feature of the spatial distribution of nutrients in the Delta is that concentrations of both nitrate and ammonium decline appreciably, particularly in the central and western Delta (Novick et al. 2015). This effect is unrelated to the source of water or initial concentration, and persists into Suisun Bay. The reason for this drawdown of N during transit through the Delta is discussed in the next section.

In-Delta Losses

As is commonly observed in river, delta, and estuarine systems globally, river-borne nitrogen is appreciably attenuated in the Delta (Seitzinger et al. 2006; Wollheim et al. 2008). This effect is generally attributed to changes in water residence time as systems become tidal, uptake into biomass, and denitrification (Seitzinger et al. 2006). Novick et al. (2015) have quantified this effect in the Delta using a simple box model as well as a one-dimensional

![Figure 1](http://dx.doi.org/10.15447/sfews.2016v14iss4art4)
hydrodynamic model (DSM2) to characterize losses. They estimated internal N losses to be near 30%, with significant spatial variation in the extent of loss across the Delta. Losses were greatest in summer and fall, during periods of high temperature, lower flows, and increased residence time. Further, they found that losses were greatest in the north, south, and central Delta, attributing this effect to the presence of flooded islands, wetlands, and higher mean residence times. Phosphate has not been similarly analyzed.

Higher residence time leads to higher loss because it provides greater opportunity for uptake into aquatic organisms as well as microbial transformation through denitrification—the conversion and subsequent loss to the atmosphere of nitrate to N₂ gas. Denitrification requires an environment depleted in oxygen, such as those found in sediments and wetlands (Seitzinger et al. 2006). Cornwell et al. (2014) measured denitrification rates in Delta sediments and established that they fell within the range found in estuaries around the world. Scaling these results to the area of Delta sediments, Novick et al. (2015) found that denitrification could account for 25% to 30% of the estimated nitrogen loss in the Delta. In a similar fashion, they estimated that uptake into biomass could account for half of the nitrogen loss in the Delta.

However, these heuristic estimates of the drawdown of nitrogen in the Delta do not account for the effects of wetlands; wetlands can be “hot spots” for denitrification and biomass uptake, as well as efficient traps for phosphate (Harrison et al. 2012; Wolheim et al. 2014). Changes in the type, location, and density of wetlands has the potential to significantly alter the attenuation of nitrogen that passes through the Delta, and, perhaps more importantly, where and when the drawdown occurs (Smyth et al. 2013). A study in Elkhorn Slough, California, found that restoration resulted in 50% to 70% reductions in nitrate, ammonium, and phosphate, with the effects most prominent near the restoration project (Gee et al. 2010). Understanding nutrient uptake into Delta wetlands is a significant gap in our knowledge, given the plans for large-scale wetland restoration.

**Rates and Controls on Primary Production and Nutrient Uptake**

The potential role of inorganic nutrients as a control on phytoplankton primary production within the San Francisco Estuary (estuary) and Delta has been the subject of substantial debate since *The State of Bay–Delta Science 2008* was published (SBDS 2008, Healey et al. 2008). The issue is critical for ecosystem management because phytoplankton carbon biomass likely plays a disproportionate role over terrestrially derived carbon, supplying much of the organic matter to the estuarine pelagic food web through photosynthesis (Sobczak et al. 2005).

Before the 1980s, Suisun Bay supported persistent, large diatom blooms in late summer that were accompanied by complete exhaustion of inorganic nitrogen (NO₃ and NH₄) (Di Toro et al. 1977; Ball and Arthur 1979; Dugdale et al. 2013). That condition changed dramatically with the invasion of the Asian overbite clam, *Potamocorbula amurensis* in the mid-1980s. Today, despite generally abundant nutrients, there has been persistently low phytoplankton biomass throughout the northern estuary and Delta (Cloern 1996), and measurements of primary production in Suisun Bay (Kimmerer et al. 2012; Parker et al. 2012c; Wilkerson et al. 2015) are among the lowest of estuarine–coastal ecosystems in the world (Cloern et al. 2014). The northern estuary and Delta have been characterized as “high-nutrient, low-growth” environments (HNLG, Wilkerson et al. 2015; Sharp 2001).

**Light and Grazing as Primary Controls on Phytoplankton Production and Standing Stock**

High suspended-sediment concentrations and the resulting turbidity that characterize the estuary serve as the primary control on phytoplankton production by reducing the availability of light needed to drive photosynthesis (Cloern 1999). This is evidenced by the successful application of a simple productivity model that relates rates of primary production to the availability of light for photosynthesis and phytoplankton biomass (Harding et al. 2002). The model requires calibration for individual systems to derive a “light utilization efficiency” term (Ψ), which has been shown to be robust, and can be used in other light-limited estuarine systems (Cole...
and Cloern 1987; Parker et al. 2012c). Much of our understanding of productivity is based upon this model and, as developed, the model output is best viewed as an index of the relationship between light availability and phytoplankton biomass through time (Jassby 2008). Changes in $\Psi$ may signal alterations in phytoplankton community composition, altered grazing pressure, or nutrient effects. This suggests that continued direct measurements of phytoplankton production and application of the light-productivity model with regular calibrations are not only necessary but could serve as an indicator of secondary controls on phytoplankton production.

The light-limited conditions that characterize much of the estuary provide a certain resilience against the common effects of high nutrient loads and concentrations, such as accumulations of phytoplankton biomass as “blooms” (Cloern 2001). Delta-wide, phytoplankton biomass is inversely related to freshwater discharge through control of water residence time (Jassby 2008). The intense benthic grazing pressure by $P. \text{ amurensis}$ in the brackish and marine reaches of the estuary and by $\text{Corbicula fluminea}$ in freshwater reaches, along with grazing pressures from copepods and microzooplankton (Kimmerer and Thompson 2014), appear to keep phytoplankton biomass low. However, given that light availability serves as a primary control of phytoplankton production, increasing water column transparency in the Delta should result in higher primary productivity that supports accumulation of phytoplankton biomass and the potential for phytoplankton blooms, should grazing pressure fail to keep pace with phytoplankton supply.

Small but significant increases in Delta chlorophyll have been documented in recent decades, along with declines in diatoms and increases in green algae, cyanobacteria, and flagellates (Jassby 2008; Lehman 1996), although the quality of some of the phytoplankton enumeration data collected has recently been called into question (Cloern et al. 2014). Ephemeral blooms in excess of 30 $\mu$g Chl-a L$^{-1}$ have been routinely detected in Suisun Bay with more temporally intensive sampling (e.g., Wilkerson et al. 2006, 2015; Dugdale et al. 2012), and a persistent bloom, rivaling pre-1980 blooms ($\sim$70 $\mu$g Chl-a L$^{-1}$), was observed in the lower Sacramento River in the spring of 2016. Elevated nutrients create at least the potential for phytoplankton to reach nuisance levels of biomass and production (Jassby 2008).

Recent Measurements of Phytoplankton Primary Production in the Delta

Over the past decade, several investigators have measured primary production directly in Suisun Bay and the Sacramento and San Joaquin rivers using incubation techniques, including the use of C isotope tracers or changes in dissolved oxygen concentrations—with some interesting patterns revealed.

Suisun Bay. Primary production has been measured in two comprehensive studies in Suisun Bay (Kimmerer et al. 2012; Wilkerson et al. 2015). Kimmerer et al. (2012) made temporally intensive measurements of phytoplankton biomass and production during spring and summer in the low salinity zone (generally Suisun Bay) over 2 years (Figure 2). The authors found little variation in biomass or primary production seasonally or inter-annually despite large differences in freshwater flow between years. Periodic phytoplankton blooms do occur in Suisun Bay (Wilkerson et al. 2006, 2015; Dugdale et al. 2012). Wilkerson et al. (2015) examined primary production during spring over 2 years (2011 and 2012) and found lower rates during high freshwater discharge in 2011. Spatially, primary production was lowest in mid-Suisun Bay and substantially higher (7- to 10-fold) at a shoal station in Grizzly Bay, when compared to the deeper channel in Suisun Bay. The depth-integrated production measurements varied considerably from 112 mg C m$^{-2}$ d$^{-1}$ to $>$490 mg C m$^{-2}$ d$^{-1}$. The timing of highest net production was earlier in upstream locations compared to downstream locations (Jassby and Cloern 2000).

Sacramento River. Primary production has been directly measured in the Sacramento River (Lehman 2007; Kress 2012; Parker et al. 2012b) and in the Yolo Bypass (Lehman 2007), where substantially higher rates of primary production were observed and were attributed to improved light conditions. Differences in phytoplankton community composition were also noted, with diatoms and green algae characterizing the Yolo Bypass while cryptophytes were dominant in the Sacramento River. Transects of the Sacramento River...
from above the confluence of the American River in Sacramento to the confluence with Suisun Bay have revealed what has been described as a “U-shaped” pattern in phytoplankton biomass and primary production (Foe et al. 2010; Parker et al. 2012b; Kress et al. 2012; Glibert et al. 2014). The lowest rates of primary production were observed between Hood and Rio Vista. Kraus et al. (forthcoming) have attributed this pattern to top-down controls that include grazing and the sinking of diatoms. Kress et al. (2012) noted a dominance of flagellates in the Sacramento River, except for during an “extreme” freshwater flow event during the spring of 2011, when the Sacramento River was dominated by diatoms and supported elevated phytoplankton biomass (Kress 2012).

San Joaquin River. Declines in phytoplankton biomass and primary production in the San Joaquin River were documented downstream of Stockton before the SRWCF was upgraded (Lehman 2007). The lower primary production was associated with higher biomass-specific C uptake, indicating that the changes were mostly from declines in phytoplankton biomass. Based on these results, it was speculated that less turbid, slower moving water or ammonium concentrations may have led to shifts in the phytoplankton community from diatoms to flagellates. After the 2007 upgrade to advanced secondary treatment at the SRWCF that included reductions in ammonium, Kress (2012) also observed declines in phytoplankton biomass in the vicinity of the area of effluent discharge, with the phytoplankton community primarily made up of diatoms and chlorophytes. The Stockton Deep Water Ship Channel, one of the few regions of the Delta where bottom water hypoxia commonly occurred, was also linked to ammonium load from the SRWCF with the low dissolved oxygen attributed to a combination of nitrogen-fueled increases in phytoplankton biomass and to biochemical oxygen demand (Jassby and Van Nieuwenhuyse 2005).

Measurements of Phytoplankton Nutrient Uptake in the Delta

Unlike the decades-long history of primary carbon production studies in the Delta, we are not aware of direct measurements of phytoplankton nitrogen uptake in the estuary or Delta before 1999. Over the past decade several studies have reported rates of nitrate and ammonium uptake, providing insight into the interactions of varying forms of inorganic nitrogen and their relations with primary production.

The Northern Estuary and Suisun Bay. Weekly to monthly measurements of chlorophyll, nitrate uptake, and ammonium concentration between 1999 and 2003 (Figure 3; Wilkerson et al. 2006) revealed seasonal inorganic nitrogen uptake rates that were similar between spring and summer with nitrate contributing roughly a third to inorganic nitrogen uptake during spring but only ~15% in all other seasons. Phytoplankton blooms appeared to be associated with periods when phytoplankton were assimilating the much larger pool of inorganic nitrogen in the form of nitrate at rates that were substantially higher than rates observed for ammonium uptake. The results have been interpreted to suggest that one condition necessary for bloom formation was phytoplankton use of nitrate. Similar patterns have been described for phytoplankton blooms between 2010 and 2013 (Dugdale et al. 2012; Wilkerson et al. 2015).

Sacramento River. Transects completed in the Sacramento River showed nitrate uptake to have occurred only in the reach above the SRWTP at
Freeport (Parker et al. 2012b). In the upper reach nitrate uptake represented ~80% of phytoplankton inorganic nitrogen assimilation, because ammonium concentrations were often quite low (<1 µM). Similar to observations made for primary production, phytoplankton nitrogen uptake rates declined in the middle reaches of the Sacramento River between Hood and Rio Vista, and phytoplankton relied on ammonium as their primary inorganic nitrogen source. Nitrogen uptake often rebounded south of Rio Vista, and phytoplankton used both ammonium and nitrate to meet cellular N demand.

Currently, few published results explore the influence of irradiance on the assimilation of nitrogen in the Delta, and this remains an area for future scientific effort. The broader oceanographic and estuarine literature suggests that phytoplankton N uptake is light-dependent and displays Michaelis–Menten (hyperbolic) kinetics in response to irradiance that varies by the form of inorganic N being used (MacsIsaac and Dugdale 1972). Specifically, the uptake of nitrate by phytoplankton appears strongly light-dependent, whereas ammonium uptake in the dark frequently occurs at rates between 30% and 95% of uptake in the light (MacsIsaac and Dugdale 1972; Pennock 1987; Boyer et al. 1994). In central San Francisco Bay, hyperbolic N uptake versus irradiance curves indicate that both nitrate and ammonium uptake were light-saturated at relatively low irradiances (>15% of surface values), with lower light intensities required for phytoplankton to begin assimilating ammonium (Dugdale et al. 2016). Using water collected from the Sacramento River upstream of Freeport, Glibert et al. (2014) performed N amendment experiments at both high and low light, and suggested that river diatoms may be favored in conditions with high nitrate and low light.

To our knowledge, there are no measured rates of phytoplankton phosphorus (P) uptake in the Delta; clearly, this is a data gap for nutrient management in the Delta. Van Nieuwenhuyse (2007) analyzed trends in P in the lower San Joaquin River between 1975 and 2005, and noted a steep decline in P loading during the mid-1990s associated with mandated bans of P-based detergents. The rapid decline in P coincided with similar declines in chlorophyll–a concentrations, indicating a potential role for P in regulating phytoplankton. The author could not associate the decline of chlorophyll–a to documented changes in zooplankton or clam grazing, but ruled out light limitation as a cause by noting that the decline occurred at the same time the photic zone depth increased. N:P ratios were low (generally <10), suggesting a greater likelihood for N rather than P limitation. The author suggested that either the assumed N:P ratio of the algal cells was wrong or the bioavailability of the inorganic N and P pools were not the same as overall inorganic N:P ratios.

**NUTRIENT EFFECTS ON PHYTOPLANKTON PRODUCTION**

Although the paradigm in estuarine management is that anthropogenic nutrient loading will increase the probability of cultural eutrophication and associated declines in estuarine water quality, the reality is that estuaries exhibit a broad spectrum of responses to nutrient enrichment (Glibert et al. 2010). Given the diversity of habitats that occur in the estuary and Delta, it is likely that a range of nutrient-related responses may be observed (Figure 4). Phytoplankton-nutrient processes in the estuary represent a largely unexplored research area. However, monitoring and
management frameworks for nutrients are currently under development for the bay (Senn and Novick 2014a) and the Delta (CVRWQCB 2014).

At present much of the published research on nutrients in the Delta addresses one set of related conceptual models: the Ammonium Paradox/Ecological Stoichiometry. These conceptual models have linked elevated ammonium concentrations to the estuarine food web through multiple pathways (Brown et al. 2016). The section that follows describes the published research on the “Ammonium Paradox” and points to extensive challenges to the proponent’s interpretations. Senn and Novick (2014b) provide a nice synthesis of arguments for and against the ammonium paradox for Suisun Bay, and the reader is directed there for more detailed discussion. Perhaps because of the level of scientific uncertainty and the resulting controversy, the Ammonium Paradox has dominated nutrient research in the Delta at the expense of a broader discussion of potential nutrient-related effects on phytoplankton in the Delta. Studies that more completely consider the continuum of estuarine responses to elevated nutrients are still needed.

The Ammonium Paradox

One observed response to elevated nutrients in estuaries runs counter to the conventional wisdom of cultural eutrophication. This is the observations of lower phytoplankton growth with higher levels of nutrients (e.g., Yoshiyama and Sharp 2006). Borrowing from oceanography, these systems are referred to as “high-nutrient low-chlorophyll” (HNLC; Cloern 2001) or HNLG (Sharp 2001). Dugdale et al. (2007), and several publications that followed have promoted the HNLG paradigm for the estuary and have suggested that estuarine managers must separately consider the impacts of anthropogenic
nitrate and ammonium on estuarine production in the estuary. The hypothesis that lower production results from ammonium enrichment has come to be known as the “Ammonium Hypothesis” (Dugdale et al. 2007; Wilkerson et al. 2015) or the “Ammonium Paradox” (Dugdale et al. 2012; Wilkerson and Dugdale 2016). The paradox is that ammonium is necessary for phytoplankton growth, but ammonium can also reduce growth relative to phytoplankton with access to the generally larger pool of DIN that is in the form of nitrate. The Ammonium Paradox was briefly mentioned in the SBDS 2008 as an area of emerging research (Kimmerer et al. 2008), and the proposed mechanisms behind these hypotheses have been more completely described since that time.

Acknowledging that light serves as the primary control of estuarine production, Dugdale et al. (2007) conceptualized “productivity windows”: situations in which the light field becomes favorable for blooms during which anthropogenic increases in nutrients—specifically ammonium—could result in declines in estuarine primary production by cutting off phytoplankton access to nitrate. It is debated that not all inorganic nitrogen is equal with respect to phytoplankton physiology; some phytoplankton (i.e., diatoms) are nitrate opportunists that can accelerate or “shift up” nitrate transport and assimilation based upon the external supply of nitrate (Dugdale et al. 2006). Dugdale et al. (2007) also hypothesize that in the estuary, as well as in other estuaries that receive large anthropogenic ammonium loads, ammonium concentrations are sufficient to inhibit phytoplankton access to nitrate, resulting in a lower probability of phytoplankton blooms. When light conditions are favorable for phytoplankton blooms, dissolved inorganic nitrogen concentration represents the potential fuel for primary production. However, when ammonium inputs are sufficient to maintain concentrations that inhibit nitrate uptake, primary production is supported only by ammonium and the nitrate, along with potential additional primary production, is exported to the coastal ocean through the Golden Gate. Therefore, under most conditions, nitrate is for all practical purposes a conservative property in the estuary.

The Ammonium Paradox is based upon three underlying hypotheses with each receiving scrutiny and challenge. The first hypothesis is that elevated ammonium concentrations inhibit nitrate uptake. It is well established from phytoplankton studies (for reviews see Dortch 1990; Glibert et al. 2016) that phytoplankton will take up nitrate before ammonium. The interaction of ammonium and nitrate is sometimes described as “preference” because the energetic cost to phytoplankton cells to assimilate ammonium into protein is less than that required for nitrate (Syrett 1981). However, the presence of ammonium has been shown to inhibit the transport of nitrate into phytoplankton cells (e.g., He et al. 2004; Song and Ward 2007) as well as inhibiting the manufacture of the enzymes necessary for nitrate assimilation (Eppley et al. 1969; Vergera et al. 1998), and so the terms “inhibition” or “repression” have also been used to describe the phenomenon. In locations where it has been tested ([1] the three embayments of the northern estuary, Wilkerson et al. 2006; Dugdale et al. 2007; [2] the Sacramento River, Parker et al. 2012b, and [3] the San Joaquin River, Lehman et al. 2015; Parker, unpublished; Figure 5), results suggest that ammonium inhibition of nitrate is a universal feature of the estuary and Delta. A review of ammonium in Suisun Bay (Senn and Novick 2014b) concludes that there is “strong support” that ammonium inhibition of nitrate uptake does occur in the estuary.

A second hypothesis embedded in the Ammonium Paradox is that nitrogen uptake is higher when phytoplankton use nitrate rather than ammonium. This higher uptake is attributed to differences in algal communities that occur in response to the supply of nitrate and ammonium, even at nitrogen concentrations considered saturating for algal growth (Glibert et al. 2016 and references therein). Studies have suggested that diatoms thrive in nitrate-rich systems, whereas chlorophytes and cyanobacteria are often associated with ammonium-rich systems (e.g., Blomqvist et al. 1994; Hyenstrand et al. 1998; Glibert and Berg 2009; McCarthy et al. 2009; Domingues et al. 2011). Experimental manipulations conducted using algae and water from the Delta reproduced these observations (Glibert et al. 2014). Diatoms, when supplied with high concentrations of nitrate in upwelling systems, can rapidly assimilate this form of nitrogen and produce large phytoplankton blooms. Called “shift up,” diatoms appear to up-regulate the cellular machinery necessary to assimilate
nitrate (Smith et al. 1992; Lomas 2004; Allen et al. 2006) in response to increasing concentrations of nitrate (Dugdale et al. 1981; MacIsaac et al. 1985; Wilkerson and Dugdale 1987). No such shift up has been demonstrated for ammonium (Glibert et al. 2016). Here, too, there is skepticism about whether the observed differences in nitrogen uptake rates for nitrate versus ammonium have been sufficiently demonstrated or rather reflect experimental artifacts (Senn and Novick 2014b; Reed et al. 2014).

Finally, the third hypothesis to the Ammonium Paradox requires that phytoplankton primary production is lower when phytoplankton use ammonium rather than nitrate. Surveys conducted in the urban Delaware River (Figure 6), the Hong Kong Harbor, and the Sacramento River provide support for the hypothesis that elevated ammonium may inhibit C and ammonium uptake (Yoshiyama and Sharp 2006; Xu et al. 2012; Parker et al. 2012b). However, a recent study (Kraus et al., forthcoming) found no evidence of ammonium inhibition of primary production when

Figure 5  (A) Nitrate uptake versus ammonium concentration in the three sub-embayments of the northern estuary. Nearly compete inhibition of nitrate uptake occurs at ammonium >4 µmol L⁻¹. (B) Ratio of nitrate to ammonium uptake versus ammonium concentration. Bubble size is proportional to the concentration of chlorophyll-a. Nitrate/ammonium uptake occurs at ammonium <4 µmol L⁻¹; elevated chlorophyll-a concentrations occur at high nitrate: ammonium uptake. Source: Dugdale et al. (2007).

Figure 6  Evidence of ammonium inhibition of estuarine primary production during summer from the Delaware Estuary. Maximum volumetric primary production per unit Chl-a (P : B ratio) versus (A) nitrate and (B) ammonium. Source: Yoshiyama and Sharp (2006).
tracking phytoplankton growth in the Sacramento River in the presence and absence of SRWTP effluent. The water parcel absent of this effluent was created by diverting wastewater into storage basins. Controlled laboratory culture studies suggest that phytoplankton, including cyanobacteria and diatoms, grow as well on ammonium as on nitrate (Collos and Harrison 2014; Berg and Sutula 2015; Figure 7). In one field study, Esperaza et al. (2014) were able to produce a diatom bloom that was supported by ammonium during a wastewater hold in a slough adjacent to Suisun Bay. Dugdale et al. (2012, 2013), however, argued that these observations did not consider the interaction between river flow and nutrient concentration. Further complicating this picture is that unknown contaminants could serve as an anthropogenic stressor for phytoplankton C, and N uptake may co-occur with the ammonium loads in anthropogenic settings such as the Delta. This is because much of the ammonium load comes from municipal wastewater. Under this scenario, ammonium serves as a “tracer” of the effect of unidentified contaminants rather than as the direct cause. The studies from Suisun Bay and the Delta (Parker et al. 2012a, 2012c) and the Delaware River (Yoshiyama and Sharp 2006) have raised this possibility.

Cloern et al. (2014) used long-term monitoring program data from Suisun Bay to look for ecosystem-scale evidence for the Ammonium Paradox or nutrient stoichiometry as drivers of phytoplankton declines, especially for diatoms. Based on annual mean data from two stations, the authors concluded that there was no correspondence between increasing ammonium concentrations and declining chlorophyll-α or diatoms. This result may be from the coarse temporal scale used as a result of annual data aggregation, which might have obscured the processes described to initiate phytoplankton blooms in the Ammonium Paradox. Still, Cloern et al. (2014) raised an important challenge to the Ammonium Paradox, namely to place the declines in primary production attributed to anthropogenic ammonium within the context of the demonstrated phytoplankton losses resulting from grazing by clams. They conclude that though sewage inputs may play a role in declining production, it is overwhelmed by other processes. The management implications articulated in Cloern et al. (2014) are important for both sides of the controversy. Controls on phytoplankton processes are likely regulated by many factors, including light availability, grazing, freshwater flow, and nutrients; and the relative importance of these drivers likely vary temporally and spatially across the diverse hydrographic landscapes of the estuary and Delta. Efforts to manage a single “master variable” (e.g., light, grazing, flow, or nutrients) are unlikely to improve ecosystem conditions for phytoplankton throughout the Delta. Continued studies of potential

![Figure 7](http://dx.doi.org/10.15447/sfews.2016v14iss4art4)
phytoplankton responses (Figure 4) to the Delta’s high nutrient conditions are still needed, and management of nutrient loading remains an important goal for the estuary (Jassby 2008).

**MICROCYSTIS AND CYANOBACTERIAL HARMFUL ALGAL BLOOMS**

**Overview**

Growth of cyanobacteria has become increasingly prevalent in waters around the world, including the Delta, in the past 2 decades. These harmful algal blooms (cyanoHABs) are linked to anthropogenic nutrient enrichment that originates from agricultural, industrial, and urban development (Heisler et al. 2008; Li et al. 2014; Paerl and Huisman 2008). CyanoHABs are associated with a number of adverse consequences to aquatic ecosystems and human wellbeing. Blooms reduce water clarity, ultimately to the detriment of aquatic habitat (Paerl and Otten 2013). As blooms die, bacterial decomposition can cause hypoxia or anoxia and potentially fish kills (Paerl and Otten 2013). CyanoHABs also produce toxins that can lead to mortality and sublethal effects on wildlife and which require expensive treatment of drinking water supply in order to prevent negative effects on humans (Paerl and Otten 2013). Blooms are expensive to mitigate and can reduce tourism near affected water bodies, thus affecting local economies.

Several major factors affect cyanobacterial blooms, including salinity, irradiance, stratification, temperature, water residence time, and nutrient availability (Berg and Sutula 2015; Lehman et al. 2013). Salinity gradients do not control the geographic distribution of cyanoHABs; common cyanobacteria have a broad range of salinity tolerance and can survive in brackish waters (Berg and Sutula 2015). Cyanobacteria have a photo-protective function that allows them to thrive in high light levels, and their positive buoyancy helps to ensure that they have adequate irradiance (Berg and Sutula 2015). At elevated temperatures (25°C and higher), cyanobacteria grow well and outcompete diatoms and green algae (Jöhnk et al. 2008; Paerl and Huisman 2008; Reynolds 2006), and toxic strains of *Microcystis* dominate over nontoxic strains (Davis et al. 2009; Paerl and Otten 2013).

Long residence times and strong vertical stratification can lead to persistent blooms in nutrient-enriched waters (Berg and Sutula 2015; Paerl and Otten 2013), although high flushing rates, vertical mixing, and turbulence negate some of the competitive advantages for cyanoHABs. The effects of stratification, (e.g., warmer temperatures, higher irradiance, and diminished loss rates) likely promote cyanoHABs, rather than the stability of the water column itself (Berg and Sutula 2015; Elliott 2010). When a period of high flow, providing a large influx of nutrients, is followed by a period of low flow, leading to higher residence times, bodies of water are prone to cyanoHABs (Paerl and Otten 2013). Additional factors that affect cyanoHABs include dissolved inorganic C cycling, zooplankton grazing, iron availability, turbidity, pH, sediment-water column exchange of stored nutrients (Paerl and Otten 2013), nutrient recycling by heterotrophic bacteria, viral lysis, exposure to herbicides and pesticides, and dissolved silica (Paerl and Otten 2013; Spier et al. 2013).

Finally, an ample supply of nutrients (N and P) is important. Although algal blooms persist with reduced N and P (Paerl and Otten 2013), the blooms will eventually die back without adequate nutrient availability. Elevated concentrations of dissolved macronutrients favor the growth of the toxigenic ecotypes of *Microcystis* (Downing et al. 2005; Paerl and Otten 2013). In addition, Harris et al. (2016) recently found that microcystin-producing cyanobacteria were favored, as was toxin production at low N:P ratios, in midwestern U.S. reservoirs. Some cyanobacteria are capable of nitrogen fixation, though most of their demand is met through fixed N: ammonium, nitrate, nitrite, urea, amino acids, and cyanate (Berg and Sutula 2015).

Stable isotope analysis suggests ammonium as the primary source of nitrogen for cyanoHABs in the Delta (Lehman et al. 2015), though many different forms of inorganic and organic N are bioavailable (Lee et al. 2015). *Microcystis* abundance appears to be more tied to absolute amounts of N and P than the N:P ratio (Lehman et al. 2005). Relatively high nitrate concentrations in the Delta dominate the N:P molar ratio, and this evidence further supports that nitrate concentrations have little influence on *Microcystis* blooms.
The Delta

CyanoHABs in the Delta were first observed in 1999 and have become commonplace since, though not necessarily present every year (Berg and Sutula 2015). Blooms generally begin in the central Delta and extend seaward (Lehman et al. 2005, 2008, 2010, 2013) with an increasing geographic range (Figure 8; Berg and Sutula 2015; Lehman et al. 2005). Although a number of cyanobacteria that form harmful blooms have been observed in the Delta (Cloern and Dufford 2005; Kurobe et al. 2013), Microcystis is most common (Berg and Sutula 2015). Lehman et al. (2013, 2015) found that Delta Microcystis originates in the Old River and the San Joaquin River. Compared with other regions known for prevalent cyanoHABs, the coverage and biomass of Microcystis during a bloom in the central Delta is low (Berg and Sutula 2015; Mioni et al. 2012). Guidelines for cyanotoxin levels in California are not yet determined, but microcystin concentrations in the Delta are within the range of potential harm to aquatic health according to the California Office of Environmental Health Hazard Assessment (Berg and Sutula 2015; OEHHA 2009).

Blooms in the Delta have been associated with high irradiance, warm water temperatures, timing of flows, and high nutrient concentrations (Lehman et al. 2005; Paerl and Otten 2013; Spier et al. 2013). Because nutrient ratios do not differ before and during the bloom, it is inferred that neither N nor P limits Microcystis growth (Lehman et al. 2013). It is possible that the Delta is occasionally at sub-saturated levels for N, so an increase in N enrichment, especially ammonium, could lead to increased N uptake in Microcystis communities (Lee et al. 2015). Overall, nutrient levels are non-limiting, and nutrient concentrations do not correlate well with cyanobacterial cell abundance. Therefore, it is unlikely that nutrients play a major role in seasonal or inter-annual variability in cyanoHABs (Lehman et al. 2008, 2013). Lehman et al. (2013) observed increasing Microcystis abundance and toxin concentration between 2004 and 2008, with elevated occurrences during dry years. These observations

Figure 8  The Sacramento–San Joaquin Delta region. Red bubbles mark locations with greatest Microcystis associated surface Chl-a concentrations (largest bubble = 554 µg Chl-a L⁻¹). Sources: figure from Berg and Sutula (2015); data from Lehman et al. (2005).
are especially well correlated with lower streamflow, suspended solids, higher water temperature, and nutrient levels. Dry years have elevated ammonium concentrations because less water dilutes effluent in the Sacramento River (Jassby 2005), and the ammonium concentration of the effluent increases. Although nutrients may play a critical role in the magnitude and extent of cyanoHABs, the factors that likely promote the proliferation of *Microcystis* in the Delta are increased water temperatures (Paerl and Paul 2012) and water column clarity (Lehman et al. 2013). If these two factors occur early in the cyanoHAB season (June to November), algal blooms could initiate earlier and grow for a longer period of time (Berg and Sutula 2015; Peeters et al. 2007). The effects of vertical mixing (ultimately reducing the availability of light) can temper these conditions. In fact, artificial mixing is a technique used to address blooms in some systems (Burford and O’Donohue 2006; Reynolds et al. 1983), and natural mixing that occurs in the Delta may help restrict cyanoHABs (Berg and Sutula 2015).

Because cyanobacteria are not routinely monitored in the Delta (though many indicator variables are, including salinity, turbidity, temperature, chlorophyll-*a*, and phytoplankton species composition), status and trends are difficult to determine and the relative importance of nutrients remains unclear (Berg and Sutula 2015). In spite of these uncertainties, nutrient management could be considered as a potential method for cyanoHAB management. Increasing flushing rates could effectively control cyanoHABs, if the incoming water has low nutrient concentrations (Paerl et al. 2011; Paerl and Otten 2013); however, this solution may be politically infeasible in the Delta, where water is a precious resource and the region is regularly affected by drought. One practical method of cyanoHAB control may be reducing nutrient inputs through wastewater treatment facility upgrades, or reducing nutrient loading through removal strategies such as wetland restoration (Paerl and Otten 2013). There has been a call for more research concerning cyanobacteria community composition, population dynamics, and nutrient biogeochemistry both in general and in the Delta specifically. The SRWTP upgrade presents a valuable research opportunity in the Delta because ammonium levels and N loads will decrease in the next decade, which may reduce the growth rate of *Microcystis* and decrease the frequency and intensity of blooms (Lee et al. 2015). Phytoplankton biomass and primary productivity are often low compared to available nutrients in the Delta, so the effect of this “large-scale ecosystem experiment” on *Microcystis* is an important standing question.

**INVASIVE AQUATIC VEGETATION**

Dramatic increases in coverage of invasive aquatic vegetation have occurred across the Delta over the last decade, particularly for two species: *Egeria densa* (Brazilian waterweed) and *Eichhornia crassipes* (water hyacinth, Figure 9). Coverage of *E. densa*, now the dominant submerged aquatic plant species in the Delta (Santos et al. 2011), increased 50% from 2000 hectares in 2007 to 2900 ha in 2014, while coverage of *E. crassipes* increased 4-fold from ~200 hectares on average from 2004 to 2007 to ~800 hectares in 2014 (reviewed in Boyer and Sutula 2015). A growing threat is being recognized in a suite of related species in the genus *Ludwigia* (water primrose, Figure 9), which have now attained coverage approximately equal to that of *E. crassipes* despite being relatively unknown a decade ago.

Globally, *E. densa* and *E. crassipes* are recognized as nuisance species throughout much of the temperate world (Bini and Thomaz 2005; OTA 1993). These species are especially problematic to human activities in the Delta because they impede navigation for commercial or recreational purposes, but they are also implicated in altering habitat and negatively affecting native species. These invasive macrophytes are considered ecosystem engineers (Yarrow et al. 2009; Wright and Jones 2006; Jones et al. 1994), because their presence in an environment affects the availability of resources for other species through alteration of biotic or abiotic materials. Importantly, these species often create conditions that are more favorable to their own growth in a positive feedback loop. For example *E. densa* has been shown to reduce turbidity and water velocity—conditions that facilitate growth and range expansion (Hestir et al. 2015). As such, much attention has been placed on understanding factors that drive the growth and distribution of these species worldwide (e.g., nutrients, light, temperature, salinity).
Nutrients and Macrophyte Growth

Two recent studies document the importance of salinity (Borgnis and Boyer 2016) and turbidity and depth (Durand et al. 2016) on the growth of invasive vegetation in the Delta, but the relative importance of nutrients in driving the growth and expansion of these species remains largely unknown. For example, no local studies have assessed the effects of nutrients on macrophyte growth under various scenarios of light or temperature. Generally, increasing nutrient concentrations result in increased biomass of *E. densa* or *E. crassipes* (Feijoó et al. 1996, 2002; You et al. 2014), though at least some studies have found no such response, possibly because they were conducted at relatively high nutrient concentrations (at or above ~360 µM N–NO₃⁻ or N–NH₄⁺). The magnitude of the effect of elevated nutrients on *E. crassipes* biomass depends on various factors including the N:P ratio and temperature (You et al. 2014). Concentrations of nutrients used in all of these studies are typically much higher than those reported for the Delta (on the order of 36 µM DIN and 2 µM DIP; Foe et al. 2010), so extrapolation of results must be done with caution.

Aquatic macrophytes can generally obtain nutrients from either the water column or sediments, though the proportion of nutrient uptake by source has been debated for different species and nutrients (Chambers et al. 1989). Truly free-floating vegetation like *E. crassipes* primarily obtain nutrients from the water column. For submerged or emergent species rooted in the sediment, such as *E. densa* and *Ludwigia* spp., the primary source of nutrients is often the upper layer of the sediment (Barko and Smart 1981). However, at least one study has found that *E. densa* shoots can be the primary tissue for nutrient uptake (Feijoo et al. 2002). To date, no studies have investigated these processes in the Delta. Understanding which tissues primarily absorb nutrients is essential to understanding if and how nutrient management could affect macrophyte growth in the Delta. For example, management of water column nutrients may have a more straightforward effect in floating species such as *E. crassipes* compared to species rooted in the sediment, where sediment nutrient cycling would still need to be considered.
Both N and P, the primary nutrients that control plant growth worldwide, are available in Delta waters at concentrations well above levels likely to be limiting (Cloern 2001). In the Delta, ammonium concentrations and N:P ratios increased over a 30-year period that ended in 2006 (Glibert 2010). However, in the decade that has followed—the period of dramatic expansion of these macrophytes—those trends are no longer evident (Larry Walker Associates 2015). TN and TP also show no clear trends over this time period. Regardless of Delta-wide trends, no studies have directly assessed trends in species growth or coverage against nutrient concentrations at specific sites.

Changing forms and proportions of nutrients are an important stressor in aquatic systems throughout the world, including the Sacramento–San Joaquin Delta (Glibert 2012). Wilson et al. (2005) estimated that N becomes limiting for *E. crassipes* at inorganic N:P ratios less than 7. Because the N:P ratio of Delta waters averages above 8 (Foe et al. 2010; Gilbert 2010), *E. crassipes* is not likely to be limited by N. *Egeria densa* is believed to grow well even under high water column N:P ratios because of its ability to tolerate high levels of N, particularly when in the form of ammonium, and because it can acquire P from sediments as well as the water column (Feijoo et al. 2002). Dense submerged macrophyte beds also can produce low dissolved oxygen and high pH conditions, which can stimulate P release from sediment (Barko and Smart 1980; Cornwell et al. 2014; Gilbert 2012) and promote growth in a positive feedback loop. Phosphate also appears to be more readily absorbed in *E. densa* than either ammonium or nitrate (Feijoo et al. 2002).

The ratio of available N forms has been shown to affect phytoplankton blooms (Wilkerson et al. 2006), but any similar relationship for aquatic macrophytes has not been studied in the Delta. *E. crassipes* readily absorbs added N regardless of form (Carignan and Neiff 1992; Heard and Winterton 2000; Moran 2006), though at least one study has found the species to be more efficient at taking up ammonium than nitrate when both were supplied in equal proportions (Reddy and Tucker 1983). Similarly, *E. densa* has been shown to take up ammonium more readily than nitrate in studies occurring in Brazil (Feijoo et al. 2002) and in Florida (Reddy et al. 1987). However, there was no significant effect of N source on biomass in any of these studies.

**Nutrient Cycling**

*E. densa* and *E. crassipes* are both known for their abilities to take up and store nutrients (Gopal 1987; Reddy et al. 1987). In fact, *E. crassipes* has been employed in water treatment projects to remove nutrients from water bodies around the world (Malik 2007). *E. crassipes* is capable of higher N and P removal than other co-occurring species, including *E. densa* (Reddy and DeBusk 1985). Despite these species’ large capacities for nutrient uptake, their effects on water column nutrient concentrations is hypothesized to be low because of the relatively low total coverage of these species Delta-wide (Hestir et al. 2008; Boyer and Sutula 2015; 3% for *E. crassipes*, 11% for *E. densa* in 2014, Khanna and Ustin, as cited in Boyer and Sutula 2015).

Decomposition of large mats of *E. crassipes* could have a large effect on nutrient cycling. In the Delta, only one study has investigated decomposition’s effect on nutrient cycling. Greenfield et al. (2007) assessed the effects of mechanical shredding on water column nutrient concentrations and found elevated TP, organic P, and TN. Flow conditions influenced the duration of the effect; under low flow, the elevated nutrients lasted for several weeks. Seasonal die-back of *E. crassipes* might produce a similar effect. *E. densa* does not die back seasonally in the Delta (Boyer et al. 2013; Santos et al. 2011), but die-back of dense submerged vegetation beds after chemical control treatments could lead to similar spikes in nutrient concentrations. To fully characterize the roles of these species on nutrient cycling in the Delta would require information on productivity rates, nutrient sequestration, and nutrient cycling among plant tissues, the water column, and sediments (Boyer and Sutula 2015).

Can nutrient management reduce the distribution and coverage of aquatic macrophytes in the Delta? A major limitation to understanding the mechanisms that underlie the expansion of invasive macrophytes in the Delta is that Delta-wide surveys of vegetation occur only sporadically. There has never been a consistent monitoring program to assess Delta-wide trends in spatial coverage of invasive aquatic
CONTINUOUS SENSING OF NUTRIENTS WITHIN THE DELTA

New Developments in Sensor Technology

Recent technological advances now permit collection of nutrient data—nitrate, phosphate and ammonium—in situ, and frequently enough to resolve processes on diurnal, tidal, and individual-event time-scales, dramatically improving our understanding of processes that shape aquatic systems (Johnson et al. 2010; Kirchner et al. 2004; Pellerin et al. 2016).


One fundamental consequence of finding higher-than-expected variability is that it calls into question classical techniques for calculating loads as a function of intermittent concentration data and continuous discharge data. Comparison of nutrient fluxes and loads calculated using less-frequent grab sample data to that calculated from high-frequency data has demonstrated that data collection at more frequent intervals improves accuracy, even in large rivers that are assumed to be buffered from short-term nutrient pulses (Carey et al. 2014; Cassidy and Jordan 2011; Pellerin et al. 2014). Assessments of these types of nutrient data do not yet exist for the Delta.

Although improved load estimates are important, the highlight of high-frequency data is that it provides novel insights into nutrient sources and cycling, and improves the ability to quantify these processes. There are many excellent recent examples in the literature (Bowes et al. 2015; Cohen et al. 2012; Collins et al. 2013; King et al. 2014; Voynova et al. 2015). One such example from the Delta is for quantifying nitrification rates in the Sacramento River (O’Donnell 2014). O’Donnell (2014) used data from the USGS monitoring stations located at Freeport and Walnut Grove on the Sacramento River to estimate nitrification rates by determining the change in nitrate concentration between stations (Figure 10). To account for exogenous inputs and in-river uptake of nitrate, the nitrate change was also determined during the times the SRWTP effluent was diverted into storage basins for maintenance or testing. The difference between the calculated nitrate change in the presence and absence of wastewater effluent was taken to be nitrification of wastewater-derived ammonium. As expected from the differences in temperature, estimated nitrification rates were 70% greater in the summer than in the winter, and within the range of published rates from other aquatic ecosystems (O’Donnell 2014). Results indicate that if the measured rates are representative, it would take between 11 and 17 days to convert the entire ammonium pool to nitrate, similar to transit times from the Sacramento River to the estuary, demonstrating why improving our understanding of nitrification rates is warranted.

Additional Nutrient Sensors

Although most high-frequency nutrient studies to date involve nitrate sensors, adoption of commercially available in situ analyzers for phosphate and prototype sensors for ammonium is growing (Rozemeijer et al. 2010; Cassidy and Jordan 2011; Bende–Michl et al. 2013; Cohen et al. 2013; Gilbert et al. 2013; Outram et al. 2014; Bowes et al. 2015). We are aware of few studies that report results from in situ high-frequency ammonium analyzers, and—not surprisingly—these studies found that ammonium, nitrate, and phosphate variability are not necessarily linked (Bende–Michl et al. 2013; Gilbert et al. 2013). Although there are no published studies from the Delta using in situ measurements of ammonium and phosphate, data collected by the USGS on the Sacramento River at Walnut Grove demonstrates their variability and the complex relationship they share with nitrate (Figure 11).
Recent studies have also demonstrated how high-frequency sensors may be used to improve our understanding of environmental processes by mapping spatial variability in rivers, lakes and estuaries, often in conjunction with fixed-station measurements (Downing et al., forthcoming; Gilbert et al. 2013; Hensley et al. 2014; Wild–Allen and Rayner 2014; Crawford et al. 2015). These examples include the Columbia River Estuary, where fixed station and mapping data allowed researchers to identify nutrient sources and transformations across a salinity gradient, and thus identify key transition zones (Gilbert et al. 2013). In Florida, longitudinal profiling of several rivers permitted nutrient removal “hot spots” to be located (Hensley et al. 2014). In the north Delta, Downing et al. (forthcoming) mapped the spatial variation in water isotopes, from which they calculated water residence time (Figure 12), an important ecological parameter related to many biogeochemical processes—and one previously not possible to quantify from field measurements. Using concurrent measurements of concentrations, they calculated rates of ecosystem uptake of nitrate, comparing how rates varied in areas with different amounts of wetlands. To assess the outcomes of the ongoing upgrades of the SRWTP and other management actions, quantitative estimates of nitrate uptake and transformation by various environmental compartments in the Delta are needed to model nutrient cycling.

Continuous Nutrient Measurements in the Delta

Continuous sensor measurements have begun in the Delta only recently. There are no published compilations or assessments of the data, although the data are available on the web in real time (http://waterdata.usgs.gov/nwis) and in daily reports via subscription. At present, eight continuous nutrient monitoring stations currently operate in the Delta (Table 1), with nitrate the only nutrient parameter reported from every site, and phosphate and ammonium reported intermittently on an event basis (Figure 11). The objectives for establishing these stations include determining drivers for available nutrients and quantifying nutrient dynamics as related to phytoplankton uptake. The stations...
Figure 11  Time series of nitrate, phosphate and ammonium from March 27 to April 8, 2014 in the Sacramento River at Walnut Grove, CA

Figure 12  Concurrent high speed mapping of nitrate [NO$_3^-$], Chlorophyll fluorescence [fCHLA], Phycocyanin fluorescence [fBGAPC], pH, Dissolved oxygen [DO%] and water residence time (days) show how environmental gradients are related to residence time. Data were collected on October 1, 2014, over a period of approximately four hours starting at the flood to ebb transition.
also directly support efforts to elucidate effects of wastewater effluent from the SRWTP on downstream nutrient concentrations and food web dynamics. Additional continuous nutrient measuring stations are planned.

Sensor Networks

Several recent papers have explored the advantages of building networks of nutrient monitoring stations to improve understanding of how ecosystems function at the landscape scale (Johnson et al. 2007; Crawford et al. 2015; Pellerin et al. 2016). Data from sensor networks can be used to quantify constituent sources, calculate transport times, and calculate transformation rates, which together can generate new insights and quantitative estimates of ecosystem processes. Rigorous evaluation of these data can inform current monitoring programs by quantifying the uncertainty and bias obtained from lower-frequency measurements, and can help design future sampling programs that take into account cost and accuracy (Hirsch 2014; Jiang et al. 2014).

However, as use of in-situ sensors becomes more common, to ensure data comparability and quality, there must be a concerted effort among users to develop, improve, document and adhere to community protocols for operation, maintenance and calibration. Further, users need to develop tools to effectively carry out quality assurance/quality control on the large volumes of data such networks generate, and to improve the visualization and analysis tools necessary to make these data useful to managers, policy-makers, and other scientists in a timely manner, preferably in near real time (Johnson et al. 2007; Pellerin et al. 2016).

A long-term commitment to high-frequency monitoring in the Delta will improve the ability to quantify how ecosystem processes are affected by events such as storms (Saraceno et al. 2009) and drought (Outram et al. 2014). This will help to more rapidly identify abrupt state changes and to recognize long-term change against a background of continuous variability, as well as the data necessary to resolve processes at short time-scales (Pellerin et al. 2009). These data can also be used to calibrate, validate, and improve models that water managers and policy-makers rely on to make decisions and to identify periods where water quality parameters either exceed or are below critical thresholds (Carey et al. 2014). Further, these data can help reduce uncertainties in models, and thereby improve assessments related to water management actions and/or climate variability (Pellerin et al. 2016). As an integrated understanding of ecosystem processes results from such data and synthesis, the development, implementation, and evaluation of water-management strategies and policies will be advanced (Cassidy and Jordan 2011; Outram et al. 2014; Pellerin et al. 2014).

RESEARCH NEEDS AND DIRECTIONS FOR DELTA NUTRIENTS

There are many research needs and directions for studying nutrients within the Delta, and recent reports, workshops, and symposia have put forth many useful recommendations. We focus on four

Table 1  Current USGS nutrient monitoring stations in the Delta

<table>
<thead>
<tr>
<th>Site name</th>
<th>Site abbreviation</th>
<th>NWIS station number</th>
<th>Date established</th>
<th>Latitude</th>
<th>Longitude</th>
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<td>8/21/2013</td>
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<td>121.517222</td>
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<td>1/31/2014</td>
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<td>8/19/2014</td>
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research areas that we believe will prove fruitful: (1) coupling continuous sensor data for nutrients with similar sensors for dissolved oxygen and temperature to estimate daily rates of primary production and ecosystem respiration along with nutrient stoichiometry; (2) linking existing hydrodynamic models of the Delta with appropriate water-quality processes that involve dissolved nutrients; (3) measuring key rates of nutrient uptake and transformation in habitat types off main channels including sloughs, backwaters, tidal marshes, macrophyte beds, and aquatic sediments; and (4) exploring the use of stable isotopes to trace the movement and fate of effluent-derived nutrients within the Delta.

The Delta has an extensive network of monitoring stations that measure flow (Burau et al. 2016), conductivity, temperature, turbidity, chlorophyll, pH, and dissolved oxygen (http://waterdata.usgs.gov/nwis). Daily dissolved oxygen patterns—when coupled with data on temperature, barometric pressure, and light regime—can be used to estimate gross primary production and ecosystem respiration in aquatic ecosystems as pioneered by Odum (1956) and now widely used in various aquatic ecosystems (Grace et al. 2015). Metabolism estimates can also be coupled to continuous nutrient sensor data to examine both elemental stoichiometry and the coupling of primary production and ecosystem respiration to the uptake of nutrients such as ammonium, nitrate, and phosphate (Cohen et al. 2013). Few places anywhere have the combined measurements of flows, fixed stations for basic water quality, and the network of high-frequency nutrient stations (Table 1) that are found in the Delta. This presents an excellent opportunity to estimate daily rates of primary production and ecosystem respiration that are linked to nutrient uptake and stoichiometry.

Hydrodynamic modeling of the Delta is well developed and mature with several existing models applied to portions of—or the totality of—the Delta (Trowbridge et al. 2016). The linking of these hydrodynamic models to water-quality models that focus on the transport and fate of nutrients is a logical next step. Trowbridge et al. (2016) offered recommendations to make this connection between hydraulics and nutrient biogeochemistry a reality for the Delta. Key recommendations included good data management, phased implementation of the coupled models, selecting the right model—or models—for the task, rigorous quality assurance, and regular workshops between modelers and biogeochemists. If resources can be found to develop these tools, an excellent opportunity exists to make progress at this interface between modeling and nutrient biogeochemistry.

Much of what we know about nutrients in the Delta has been gathered from samples collected where larger boats can travel. Locations that are less commonly sampled include sloughs, backwaters, tidal marshes, macrophyte beds, and aquatic sediments. Key processes that take up nutrients (e.g., uptake by attached bacteria, fungi, algae, and aquatic macrophytes) and transform nutrients (e.g., nitrification and denitrification) are concentrated in these rather than open-water habitats. Novick et al. (2015) have made an initial estimate of ~30% internal losses for N within the Delta, and the roles for various habitat types in the uptake by biota or loss to the atmosphere through denitrification remain an important unknown. Current planning and some initial implementation of large-scale restoration within the Delta through EcoRestore (http://resources.ca.gov/ecorestore/) heighten the need for rate measurements on key processes that cycle nutrients in these parts of the Delta.

A large-scale natural experiment is in the offing for the Delta because the Sacramento Regional County Sanitation District has broken ground on the EchoWater Project (http://www.regionalsan.com/echowater-project) that will upgrade the SRWTP, which serves about 1.4 million people. This upgrade will reduce ammonium inputs in the north Delta from the plant by 95% or more, and substantively reduce overall inorganic nutrient inputs by 2021. Effluent N has a distinctive isotopic signal that allows effluent nitrogen to be traced within the Delta. Effluent water is generally enriched in the heavier isotope of nitrogen ($^{15}$N) compared to the lighter isotope ($^{14}$N), and, therefore, a natural tracer is available to study the transport, uptake, transformation, and role in food webs of effluent nitrogen (Costanzo et al. 2005; Miller et al. 2010). This large-scale experiment, if adequately documented, could yield far-reaching insight into the dynamics of nutrient cycling within the Delta.
CONCLUSIONS

An ever-changing Delta has encouraged a re-evaluation of the roles for nutrients within it. At the turn of the millennium, nutrient levels were high, nutrient limitation was rare, and regulation of primary production resulted from factors other than nutrients (Jassby et al. 2002). As Delta waters became clearer (Schoellhamer et al. 2016), and Microcystis blooms became commonplace and invasive aquatic macrophytes became more pervasive, questions arose about the effects of concentrations, chemical form, and elemental stoichiometry of N and P on the structure and productivity of the open-water algal community. There has been a resurgence in interest in nutrients and their roles within the Delta. Good, long-term data on nutrients allow the status and trends of Delta nutrients to be assessed, and an emerging network of high-frequency nutrient sensors allows high resolution studies on the availability, transport, and fate of dissolved nutrients in Delta waterways. We highlight four emerging opportunities of interest relating to nutrients within aquatic ecosystems of the Delta. The first is how a major, nearly completed upgrade to the SRWTP will affect the primary producers and food webs of the Delta. The second is the opportunity to couple high temporal and spatial resolution estimates of primary production and ecosystem respiration to nutrient uptake and the elemental stoichiometry of C, N, and P. The third is addressing how tidal marsh restoration and floodplain re-connection within the Delta will affect nutrient biogeochemistry. The fourth is linking hydrodynamic models of the Delta to a growing understanding of biogeochemical processes within the tidally dominated Delta. Nutrients are most definitely of growing interest within the Delta, and future changes to the Delta allow some fundamental hypotheses about nutrient biogeochemistry of this rapidly changing landscape to be tested.

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NOTES

ABSTRACT

This paper reviews and highlights recent research findings on food web processes since an earlier review by Kimmerer et al. (2008). We conduct this review within a conceptual framework of the Delta–Suisun food web, which includes both temporal and spatial components. The temporal component of our framework is based on knowledge that the landscape has changed markedly from historical conditions. The spatial component of our framework acknowledges that the food web is not spatially static; it varies regionally and across habitat types within regions. The review highlights the idea of a changing baseline with respect to food web function. New research also indicates that interactions between habitat-specific food webs vary across the current landscape. For example, based on early work in the south Delta, the food web associated with submerged aquatic vegetation was thought to provide little support to species of concern; however, data from other regions of the estuary suggest that this conceptual model may not apply across the entire region. Habitat restoration has been proposed as a method of re-establishing historic food web processes to support species of concern. Benefits are likely for species that directly access such restored habitats, but are less clear for pelagic species. Several topics require attention to further improve the knowledge of food webs needed to support effective management, including: (1) synthesis of factors responsible for low pelagic biomass; (2) monitoring and research on effects of harmful algal blooms; (3) broadening the scope of long-term monitoring; (4) determining benefits of tidal wetland restoration to species of concern, including evaluations of interactions of habitat-specific food webs; and (5) interdisciplinary analysis and synthesis. The only certainty is that food webs will continue to change in response to the changes in the physical environment and new species invasions.

KEY WORDS

Food web, invasive species, production, submerged aquatic vegetation, pelagic, zooplankton, microzooplankton, phytoplankton
INTRODUCTION

The northern reaches of the San Francisco Estuary (the estuary), including the Sacramento–San Joaquin Delta, Suisun Bay, and Suisun Marsh, have undergone substantial shifts over the last 150 years in configuration, hydrology, species composition, and the dominance of humans on the landscape (Nichols et al. 1986; Whipple et al. 2012). As a result of these shifts, the upper estuary and most notably the Delta engender nearly intractable conflicts over land use, water use and ecosystem protection (NRC 2012; Luoma et al. 2015). Much of this conflict is between human water uses (e.g., agriculture and urban) and protection for the endangered Delta Smelt Hypomesus transpacificus, a Delta endemic, and other declining native fish species. Reasons for the declines are many, but a factor that contributes to the lack of recovery of these species is a shortage of food (Slater and Baxter 2014; Hammock et al. 2015). The consequent interest in the estuarine food web has led to new ideas and findings about why food web productivity is so low compared to that in other estuaries (Dugdale et al. 2012, 2013; Wilkerson et al. 2015; Cloern and Jassby 2012; Kimmerer et al. 2012), but also to conflicting ideas about the potential to reverse these declines.

The principal objective of this paper is to highlight recent research findings on estuary food web processes since the review by Kimmerer et al. (2008). Research during the previous three decades had led to considerable understanding of food web processes in the Delta, so recent improvements in understanding of food webs have been largely incremental. Because much of the recent work builds on previous work, we review earlier work as needed to provide appropriate background; however, it is not our intent to provide a comprehensive review of the estuary’s food web literature. We also do not consider factors that could affect food webs but for which data are insufficient for assessment. Specifically, we do not discuss pesticides, which are known to have episodic effects on organisms and mesocosm food webs, but sustained food web effects in the environment have not been demonstrated (Fong et al., submitted). We review the available information from temporal, spatial, and habitat-specific perspectives, all of which are needed to inform management actions intended to re-establish food web functions for the benefit of species of concern.

Our geographic scope ranges from the legal Delta seaward to Carquinez Strait, including Suisun Bay and Suisun Marsh (Figure 1). We also consider floodplain areas that have been shown to be important to Delta ecology. We discuss information from more seaward areas of the San Francisco Estuary and from other estuaries as needed. Because specific studies often focus on only one or two of these regions, we use “Delta–Suisun” to refer to all three regions combined, and otherwise refer to the Delta, Suisun Bay, or Suisun Marsh, separately.

We start this review by presenting our conceptual framework of the Delta–Suisun food web, which includes both temporal and spatial components. The temporal component is based on knowledge that the landscape has changed markedly from historical conditions. The spatial component of our framework acknowledges that the food web is not spatially static; it varies regionally and across habitat types within regions. We first present a brief history of some important changes in the Delta–Suisun region over time and a description of their current condition. We then consider food web findings for different habitats and Delta–Suisun regions, using a two-part conceptual framework. Finally, we discuss the relevance of these findings for habitat management actions that have been proposed to increase food web support for species of concern (e.g., Herbold et al. 2014). We do not consider other potential benefits of such actions here.

CONCEPTUAL FRAMEWORK

Our conceptual framework highlights food web differences among historical periods, habitats and geographic areas of the Delta–Suisun region. We first describe some of the key food web changes since the Gold Rush up until the current time. Understanding the history of ecological changes is important when actions intended to restore functions and services are considered (Lotze and Muir 2009; Lotze and Worm 2009), including food web functions. Since the majority of research on estuarine food webs in the estuary has focused on pelagic habitat, our discussion of historical periods is also pelagic-focused. Given that systematic scientific data
Figure 1 Map of the Sacramento–San Joaquin Delta, Suisun Bay, and associated areas discussed in this paper
collection did not begin until the mid-20th century, our characterizations of early food web processes are limited and speculative, but new work on the historical landscapes of the Delta–Suisun region (Whipple et al. 2012; Robinson et al. 2014) provides an increasingly solid basis for such speculation, and efforts are underway to estimate historic primary production (Robinson et al. 2016; Cloern et al. 2016).

The spatial part of the conceptual framework incorporates the idea of differences in food webs associated with different geographic regions (Figure 2) and habitats (Figure 3). We discuss pelagic and benthic food webs together because they are tightly linked through the grazing effects of benthic organisms—primarily clams—on pelagic organisms. We define geographic areas where we believe food web processes are relatively similar across the region (Figure 2). Within each geographic region we discuss the habitat-associated food webs for which data are available and that are likely important. Geographic differences in food web structure and function in an estuarine system this large and complex are to be expected. Importantly, geographic boundaries between the areas defined here are not sharp; they are blurred by the high variability characteristic of estuarine environments on tidal, seasonal, and annual scales (Kimmerer 2004). In this paper, we are mainly concerned with freshwater and low-salinity water areas because they dominate the Delta–Suisun region during most years and are most thoroughly studied. The low-salinity zone (LSZ) has been variously defined by researchers, but generally ranges between 0.5 to 1.0 salinity at the low end and 5.0 to 6.0 salinity at the high end. The position of the entire salinity field fluctuates across Delta–Suisun areas in response to freshwater flow on longer time-scales, and to tides on the shortest time-scales.

**DELTA–SUISUN FOOD WEBS: A CHANGING BASELINE**

The concept of changing baselines has received attention in the last 2 decades (Duarte et al. 2009). It has become apparent globally that the historical trajectories of human-dominated ecosystems cannot be retraced, and that the historical condition of these systems cannot be recaptured. Thus, the past serves as a guide but not a template for future actions to restore or improve the ecosystem.

**Pre-Gold Rush Era (<ca. 1850)**

Until the California Gold Rush, the Delta landscape was dominated by tidal and seasonal wetlands and floodplains (Whipple et al. 2012). Recent research on historical landscapes of the Delta (Whipple et al. 2012; Robinson et al. 2014) has provided essential details on the historical distribution and diversity of habitat types. For example, the central Delta was dominated by freshwater tidal wetlands, with dead-end sloughs and tidal channels linking marsh plain to open waters. The north Delta included large flood basins behind natural levees, combined with a network of tidal channels, ponds, and perennial, tidal, and seasonal freshwater wetlands. The south Delta was dominated by a complex network of distributary river channels, oxbow lakes, and tidal sloughs embedded in extensive tidal and non-tidal freshwater wetlands and floodplains. Further to the west, the bays of the estuary were fringed by expansive tidal flats and wetlands and Suisun Marsh was the largest brackish water marsh in the western U.S.

Although no direct scientific accounts exist, the food webs in these landscapes were likely fueled by a combination of biologically available organic matter produced by native marsh and floodplain vegetation and phytoplankton produced in the channels and open waters of the system (Robinson et al. 2016). Hypotheses about primary production in the historical landscapes are currently being addressed (Robinson et al. 2016; Cloern et al. 2016). Many of the small dead-end sloughs in the tidal wetlands were likely very productive, especially if their length exceeded the tidal excursion (Geyer and Signell 1992) so that long residence time allowed for accumulation of biomass. Complexity theory (Campbell Grant et al. 2007) and historical research (Williams 2006; Whipple et al. 2012; Robinson et al. 2014, 2016) suggest that the change from a reticulate dendritic landscape, with channels of various sizes and mixed residence time, to a much less complex, channelized landscape probably had profound influences on the habitat value of the Delta; however, there are no quantitative data or estimates of the abundances of the native
Figure 2  Conceptual framework for the discussion of food webs in selected geographic areas. The two-headed arrows indicate that the position and extant of the low-salinity zone will change seasonally and annually in response to flow conditions.
Large changes in a variety of landscape metrics (Table 1; Robinson 2014) suggest comparable changes in sources, extent, transport, and fate of estuarine organic matter and primary production; although, these changes have not yet been quantified. Watersheds, dam construction, diking, and water diversions in the Delta changed flow dynamics to the estuary and severed connections between floodplains and rivers, thereby reducing the supply of biologically available organic matter and primary productivity from floodplains to the Delta (Ahearn et al. 2006; Lehman et al. 2008; Opperman 2012). A pulse of sediment from hydraulic mining shoaled the estuary by ~1 meter in some places, and the shoaling lasted over a century (Schoellhamer et al. 2013).

The influx of settlers also brought a wave of species introductions, notably deliberate introductions in the 1870s of anadromous Striped Bass *Morone saxatilis* and American Shad *Alosa sapidissima* from the Atlantic Coast. Both species rapidly became established and developed large populations in the estuary (Moyle 2002). The introduction of Striped Bass was likely a large perturbation to the food webs of the estuary, but whether Striped Bass simply replaced native predators or placed increased demands on lower trophic levels is unknown.

**Landscape Change and Development (ca. 1850–1960s)**

The landscape of the estuary and watershed changed markedly with the onset of land and water development in the 19th century, and especially after the huge influx of settlers during and after the Gold Rush, with negative effects on native biota (Brown and Moyle 2005; Whipple et al. 2012; Robinson et al. 2014). Diking and draining of wetlands in the Delta for farming and other human activities, along with widening of the main channels and permanent flooding of several subsided islands, completely altered the Delta. Before development, wetland area exceeded open-water area by about 14:1; today this ratio is 1:6, an 80-fold switch in dominant habitat types (Whipple et al. 2012). Loss of wetlands in the Delta is estimated to be about 97% of the pre-Gold Rush area (Whipple et al. 2012). Losses of San Francisco Bay tidal flats and wetlands have been estimated as 70% to 93% (Callaway et al. 2011). Losses of Suisun Marsh natural habitats from conversion to managed wetlands or other human land uses have also been extensive (Manfree 2014).
Numerous inadvertent introductions of invertebrates to the estuary, especially into San Francisco Bay, occurred with increased shipping commerce (Cohen and Carlton 1997; Choi et al. 2005) with substantial effects on the Delta–Suisun food web (described below). Over the last 150 years numerous other fishes, invertebrates and plants have invaded the region, in most cases with undocumented effects.

By the onset of scientific study and monitoring of Delta and Suisun Bay ecology in the middle 1960s, a broad array of non-native species had been established as competitors of and predators on native fishes and other aquatic organisms, diverting resources away from native species. Such invasions have continued with additional invertebrate (Table 2) and fish (Moyle 2002) invasions since monitoring began. Native planktivorous species including Delta Smelt and Longfin Smelt *Spirinchus thaleichthys* were still fairly abundant during this period, despite the introductions of several competitors including American Shad and Threadfin Shad *Dorosoma petenense*.

### Table 1

<table>
<thead>
<tr>
<th>Landscape Metric</th>
<th>Pre-Gold Rush</th>
<th>Landscape modification</th>
<th>Pre-Potamocorbula</th>
<th>Post-Potamocorbula</th>
<th>Post-pelagic organism decline</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic-benthic</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Open water (ha)</td>
<td>16,344</td>
<td>↑ 63%</td>
<td>26,554</td>
<td>~</td>
<td>~</td>
</tr>
<tr>
<td>Tidal wetland</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Freshwater emergent wetland (ha)</td>
<td>193,224</td>
<td>↓ 98%</td>
<td>4,296</td>
<td>~</td>
<td>~</td>
</tr>
<tr>
<td>Area subject to tidal inundation (ha)</td>
<td>~ 150,000</td>
<td>↓ 144,000</td>
<td>~ 6,000</td>
<td>~</td>
<td>~</td>
</tr>
<tr>
<td>Dendritic channels adjacent to marsh (km)</td>
<td>1,151</td>
<td>↓ 93%</td>
<td>84</td>
<td>~</td>
<td>~</td>
</tr>
<tr>
<td>Wet meadow/Seasonal wetlands (ha)</td>
<td>37,561</td>
<td>↓ 93%</td>
<td>2,445</td>
<td>~</td>
<td>~</td>
</tr>
<tr>
<td>Seasonally flooded habitat (ha)</td>
<td>117,000</td>
<td>↓ 85%</td>
<td>19,000</td>
<td>~</td>
<td>~</td>
</tr>
<tr>
<td>Fluvial and detached channels (ha)</td>
<td>2,225</td>
<td>↓ 87%</td>
<td>298</td>
<td>~</td>
<td>~</td>
</tr>
<tr>
<td>Submerged aquatic vegetation (ha)</td>
<td>?</td>
<td>?</td>
<td>lowest modern</td>
<td>↑ 4,000–8,000 a</td>
<td></td>
</tr>
</tbody>
</table>

*a Between 2004 and 2008, total submerged aquatic vegetation coverage declined from approximately 80 to 40 km², possibly as a result of an herbicide application program targeting *Egeria densa* initiated by the California Department of Boating and Waterways in 2001 (2015 Personal conversation between E. Hestir, UC Davis, and J. L. Conrad, unreferenced, see “Notes”). However, by fall 2014, submerged aquatic vegetation coverage had rebounded to approximately 70 km² (2015 Personal conversation between S. Khanna, UC Davis, and J. L. Conrad, unreferenced, see “Notes”).*
exported by south Delta pumps from 1975 to 1989, though annual estimates were highly variable (Jassby and Powell 1994).

Zooplankton biomass was dominated by rotifers and cladocerans in the Delta and by rotifers and the copepod *Eurytemora affinis* in the LSZ (Orsi and Mecum 1986; Winder and Jassby 2011). The abundant mysid *Neomysis mercedis* comprised much of the diets of young fishes (Heubach et al. 1963; Meng and Orsi 1991; Feyrer et al. 2003). However, from 1972 to 1980, the pelagic food web began changing for unknown reasons, as phytoplankton biomass and abundance of rotifers, *E. affinis*, and *N. mercedis* declined by greater than two-fold (Orsi and Mecum 1986, 1996; Kimmerer et al. 1994; Kimmerer 2005).

During this time-period, the benthic community and grazing pressure of the Delta–Suisun region responded to the differences in flow. The seasonally fluctuating salinity (a result of flow variability) in Suisun Bay suppressed benthic invertebrate numbers and grazing except during extended and severe dry or wet periods (Nichols 1985; Peterson and Vayssières 2010). The 1976–1977 drought in particular allowed marine species to invade Suisun Bay, resulting in a depression of phytoplankton biomass (Nichols 1985; Alpine and Cloern 1992). The freshwater clam *Corbicula fluminea*, introduced well before monitoring began, likely affected the food web in the freshwater Delta, given its substantial grazing effect on phytoplankton in the Delta and elsewhere (Cohen et al. 1984; Lopez et al. 2006; Lucas and Thompson 2012) but the quantitative effect is unknown.

Through the mid-1980s, the LSZ was an area of high phytoplankton biomass (Ball and Arthur 1979), as in other estuaries (Morris et al. 1978; North et al. 2005). In the estuary, the LSZ was important rearing habitat for Striped Bass and the native Delta Smelt and Longfin Smelt (Stralberg et al. 1985; Moyle et al. 1992; Hobbs et al. 2006). At this time, primary production in pelagic estuary habitats was shown to be mainly light-limited (Cole and Cloern 1984; Alpine and Cloern 1988), with production being low in comparison to many other estuaries (Cloern et al. 2014).


An abrupt change in the estuarine food web began in 1987, after the invasion of the “overbite” clam *Potamocorbula amurensis* (Nichols et al. 1990). Native to estuaries of mainland East Asia, *P. amurensis* thrives in brackish water and tolerates variable salinity (Paganini et al. 2010). Its spread may have been facilitated by drought after a high-flow year (1986) that swept Suisun Bay clear of other benthic grazers (Nichols et al. 1990). Phytoplankton biomass decreased about five-fold in Suisun Bay and the western Delta (Alpine and Cloern 1992), the size distribution of phytoplankton shifted toward smaller cells (Kimmerer et al. 2014a), and production by diatoms nearly ceased in this region (Kimmerer 2005). The abundance of brackish-water rotifers and *E. affinis* and other copepods declined, apparently from predation by and competition with clams (Kimmerer et al. 1994; Kimmerer and Lougee 2015). Abundance of the mysid *N. mercedis* in brackish water declined sharply, apparently because of reduced food supply (Orsi and Mecum 1996), as did abundance of Northern Anchovy *Engraulis mordax*, in an apparent behavioral response to poor feeding conditions in the LSZ (Kimmerer 2006). The taxonomic composition of the benthos changed to a more diverse assemblage dominated by recently introduced species, although the assemblage shifted with seasonal and interannual movement of the salinity field (Peterson and Vayssières 2010).

In retrospect, the series of events that began after the *P. amurensis* invasion constitute an ecological “regime shift” (see “Sidebar 1”; Scheffer et al. 2001; Carpenter 2003; Folke et al. 2004). The *P. amurensis* invasion was clearly a major driver of this regime shift, but other drivers likely contributed as well, such as a prolonged drought, ammonia loading (Dugdale et al. 2007, 2012, 2013; Wilkerson et al. 2015; Dahm et al., submitted), and changes in water inflows and exports from the estuary (Winder et al. 2011; Cloern and Jassby 2012). This regime shift affected the entire estuarine food web, but was most pronounced in the estuary’s LSZ. This regime shift provided the impetus for a substantial research effort on the pervasive influence of benthic grazing and its consequences.
Pelagic primary productivity and phytoplankton biomass are now generally low throughout the northern estuary, but recent work has focused on depressed production in the LSZ (Cloern and Jassby 2012; Kimmerer et al. 2012). Only about half of the spring–summer primary production in the LSZ is attributable to phytoplankton large enough (greater than ~5 µm) to be readily consumed by copepods and clams (Kimmerer et al. 2012). Phytoplankton provide less support to copepods and other consumers since 1993 than before 1993 (Kratina and Winder 2015). Growth and reproduction of all three species of copepod that are abundant during summer in the LSZ are food limited (Kimmerer et al. 2014a).

Over 6 years after the spread of *P. amurensis*, a spate of introductions of copepods and mysids in the fresh to brackish regions of the estuary (Table 2) resulted in a nearly complete change in species composition (Orsi and Walter 1991; Orsi and Ohtsuka 1999; Winder and Jassby 2011) that further altered the food web. The biomass of copepods in the LSZ decreased less than the biomass of phytoplankton, rotifers, or mysids, presumably because of the departure of Northern Anchovy, which had comprised about half of the biomass of planktivorous fish in this region (Kimmerer 2006), and because newly introduced species used resources previously used by *E. affinis*.

**SIDEBAR 1**

**Defining Change**

As we have described in the previous sections, many factors have brought about the changes to the Delta food web. These factors have varied in kind from species introductions to landscape and water flow alteration and the time-scales have varied from gradual (Whipple et al. 2012), over decades, to abrupt (Thomson et al. 2010; Alpine and Cloern 1992), over a few months or years. In many respects, the Delta fits the definition of a novel ecosystem. Morse et al. (2014) credit Chapin and Starfield (1997) with the first use of the concept, which is commonly used among ecologists to describe ecosystems with biotic or abiotic characteristics altered by humans (Hobbs et al. 2009, 2013). Morse et al. (2014) have suggested that novel ecosystems should also have crossed

“an ecological threshold that facilitates a new ecosystem trajectory and inhibits its return to a previous trajectory regardless of additional human intervention. The resulting ecosystem must also be self-sustaining in terms of species composition, structure, biogeochemistry, and ecosystem services. A defining characteristic of a novel ecosystem is a change in species composition relative to ecosystems present in the same biome prior to crossing a threshold.”

These additional characteristics make the concept very similar to that of multiple stable states or alternative stable states (Beisner et al. 2003). Interestingly, Morse et al. (2014) do not consider the San Francisco estuary to be a novel ecosystem; however, their consideration of the literature appears limited to a single paper (Carlton and Cohen 1997). The evidence in this paper suggests that the Delta, as well as the entire estuary, meets the definition for a novel ecosystem as defined by Morse et al. (2014).

Another term often applied to change in the estuary, particularly the Delta, is “regime shift” (Moyle and Bennett 2008; Baxter et al. 2010). Regime shifts can be defined as changes in the internal dynamics and feedbacks of an ecosystem that often prevent it from returning to a previous regime, even when the driver that precipitated the shift is reduced or removed (Schefter et al. 2001; Carpenter 2003; Mac Nally et al. 2014). The undesirable characteristics (for humans) of regime shifts have led to interest in predicting when such shifts might occur (Biggs et al. 2009). Clearly, a system where introduced species have comprehensively replaced native flora and fauna has undergone a regime shift; however, few examples of change in estuarine systems are able to meet all the formal statistical requirements for establishing a regime shift (Mac Nally et al. 2014).

Regardless of various labels and definitions, it is clear that the food web in the Delta and the larger estuary has experienced profound changes in species composition and trophic pathways, with repercussions throughout the ecosystem. Although it is unclear if the current system is stable, the probability of it returning to any previously observed state seems small or nonexistent. This has led to the suggestion of applying the concepts of reconciliation biology (Rosenzweig 2003) to the Delta (Moyle et al. 2012). In the context of reconciliation ecology, managers seek to improve conditions for native species while accepting that the system has been irrevocably altered by human use and will continue to be used to support human goals. So, improving ecosystem conditions for native species in the estuary must happen alongside continuing use of land and water by humans and continuing physical and biological change. Such continuing change will likely include changes in water system management and operations, new invasions, and various physical and biological manifestations of climate change, including increasing temperature stress for sensitive native species (Brown et al. 2013, 2016).
The different copepod species are roughly nutritionally equivalent (Kratina and Winder 2015), but are not equally available to consumers (Meng and Orsi 1991). For example, the most abundant of the newly introduced species, *Limnoithona tetraspina*, is not commonly consumed by fish (Bryant and Arnold 2007; Slater and Baxter 2014), and the mortality rate of adults is very low, presumably because of the small size and cryptic behavior of these copepods (Kimmerer 2015). The copepod *Pseudodiaptomus forbesi*, which now makes up over half of the diet of Delta Smelt and other planktivorous fishes (Bryant and Arnold 2007; Slater and Baxter 2014), is most abundant in freshwater. The LSZ receives *P. forbesi* from freshwater via advection and dispersion, and they tend to be retained in the LSZ through their tidally timed vertical migration behavior (Kimmerer et al. 2014b). *Acartiella sinensis* is consumed by fish (Slater and Baxter 2014) and is apparently able to survive in the LSZ by preying on other copepods, particularly *L. tetraspina* and *P. forbesi* (Slaughter et al. 2016).

Even with the introductions of several mysid species, total mysid abundance and biomass have remained low compared to former abundances of *N. mercedis*. The epibenthic amphipod *Gammarus daiberi*, introduced in 1983, increased in abundance after the introduction of *P. amurensis* (Peterson and Vayssières 2010). Based on results from benthic surveys, it is more abundant in freshwater than in the LSZ (Kimmerer 2004). *Gammarus* spp. generally feed by shredding detritus, although they can be predaceous (Kelly et al. 2002). *G. daiberi* moves into the water column at night and remains on the bottom by day (Kimmerer et al. 2002), which may explain its infrequent occurrence in diets of most visually feeding fishes (Feyrer et al. 2003).

Several hydromedusae (jellyfish, Table 2) have become established in the low-salinity waters of the system (Mills and Rees 2000), raising questions about possible competition with fishes for zooplankton. The introductions likely resulted from ballast water release (Rees and Gershwin 2000). Medusae are seasonally abundant in the harbors and sloughs of the estuary (Rees and Kitting 2002; Schroeter 2008; Wintzer et al. 2011a, 2011c) but are not sampled well by existing sampling programs. They consume copepods, especially nauplii (Wintzer et al. 2011b, 2013), and in Suisun Marsh their diets overlap with those of Threadfin Shad and Delta Smelt (Wintzer et al. 2011c). Medusae were reported in samples taken by the Fall Midwater Trawl Survey during 2001–2011 with an average annual catch per volume of 0.0008 m\(^{-3}\) with a maximum of 0.0030 m\(^{-3}\) (Osborn and Civiello 2013, catch based on data in their Table 1). Small (<5 mm) medusae were detected in fewer than 5% of zooplankton samples collected with a 150-µm mesh net in channels of the LSZ during spring–summer of 2006–2007 (Kimmerer et al. 2014a; W. Kimmerer, unpublished data, see "Notes"). If further investigations show these low densities to be typical, substantial predatory effects on the zooplankton of open waters seem unlikely.

The changes in lower trophic levels were followed by shifts in diets, distributions, and abundance of many fish species. For example, during 1998–1999, the stomachs of Striped Bass, Splittail *Pogonichthys macrolepidotus*, and three other species in Suisun Marsh were less full than those of fishes caught during 1979–1983 (Feyrer et al. 2003). Further, mysids made up a smaller proportion of the prey of Striped Bass and were a negligible proportion of other common fishes. The magnitudes of declines in fish abundance were positively related to the dietary importance of mysids before the declines (Feyrer et al. 2003). Abundance of Northern Anchovy declined in the LSZ but not at higher salinities, presumably because the fish abandoned this now low productivity area (Kimmerer 2006). Abundance of Striped Bass and Longfin Smelt declined soon after the clam invasion (Kimmerer et al. 2009; Thomson et al. 2010; Mac Nally et al. 2010). In addition, Striped Bass became less abundant in the deep channels and more abundant in shallow nearshore areas, presumably because of better foraging opportunities (Sommer et al. 2011). Delta Smelt are frequently captured in the north Delta (Sommer and Mejia 2013). Because the region was not well sampled until recently, it is unclear whether the high frequency of occurrence in recent samples there reflects a shift of the population toward the north Delta or simply an increase in sampling effort there.
Today: The Pelagic Food Web No Longer Supports Native Fishes

Over approximately the last 15 years the zooplankton species composition has been more stable than in the preceding decades. No introductions of copepods have been reported since 1993 or mysids since 1997 (Table 2), and no previously documented native species went extinct. Notable recent invertebrate invaders include Siberian prawn *Exopalaemon modestus* (Brown and Hieb 2014) and an invasive cladoceran *Daphnia lumholtzi* (Mueller–Solger 2001). More alarmingly to many observers, four pelagic fish species, including the native Delta Smelt and Longfin Smelt, underwent steep declines in abundance in the early 2000s (collectively known as the “pelagic organism decline,” [POD]; Sommer et al. 2007; Thomson et al. 2010). The POD may have been the manifestation of another ecosystem-wide regime shift (Baxter et al. 2010) or delayed repercussions of the regime shift that started in 1987. In any case, pelagic habitat in the LSZ and parts of the Delta appears to be less suitable for pelagic fishes than before 1987, and changes in the food web are a likely contributing cause (IEP–MAST 2015; Merz et al. 2016).

In contrast to the relatively high productivity in the LSZ before the *P. amurensis* invasion, this region

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**Table 2** Selected known introductions of copepods, mysids, and medusae to the San Francisco Estuary since monitoring began. *Eurytemora affinis* is not considered because it has existed in the estuary before monitoring began. Salinity is the reported value where the species was taken, not necessarily where it is most abundant.

<table>
<thead>
<tr>
<th>Species</th>
<th>Taxon</th>
<th>Date of first capture</th>
<th>Location</th>
<th>Salinity</th>
<th>Likely source</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Eurytemora affinis</em></td>
<td>Calanoid copepod</td>
<td>1920s?</td>
<td>Napa and Petaluma rivers, Pittsburg, Antioch</td>
<td>1-16</td>
<td>Ponto-Caspian</td>
<td>Matern and Brown 2005</td>
</tr>
<tr>
<td><em>Deltamysis holmquistae</em></td>
<td>Mysid</td>
<td>August 1977</td>
<td>Not stated</td>
<td>Not stated</td>
<td>Not stated</td>
<td>Bowman and Orsi 1992</td>
</tr>
<tr>
<td><em>Sinocalanus doerrii</em></td>
<td>Calanoid copepod</td>
<td>May 1978</td>
<td>Confluence</td>
<td>3.4</td>
<td>Asia</td>
<td>Orsi et al. 1993</td>
</tr>
<tr>
<td><em>Limnoithona sinensis</em></td>
<td>Cyclopoid copepod</td>
<td>August 1979</td>
<td>Stockton</td>
<td>–0</td>
<td>China</td>
<td>Ferrari and Orsi 1984</td>
</tr>
<tr>
<td><em>Oithona daviesae</em></td>
<td>Cyclopoid copepod</td>
<td>October 1979 or before 1963</td>
<td>Suisun Bay</td>
<td>&gt;12</td>
<td>Japan</td>
<td>Ferrari and Orsi 1984</td>
</tr>
<tr>
<td><em>Pseudodiaptomus marinus</em></td>
<td>Calanoid copepod</td>
<td>October 1986</td>
<td>Suisun Bay</td>
<td>6-8</td>
<td>Japan</td>
<td>Orsi and Walter 1991</td>
</tr>
<tr>
<td><em>Pseudodiaptomus forbesi</em></td>
<td>Calanoid copepod</td>
<td>October 1987</td>
<td>San Joaquin R.</td>
<td>–0</td>
<td>China</td>
<td>Orsi and Walter 1991</td>
</tr>
<tr>
<td><em>Aurelia</em> spp.</td>
<td>Scyphomedusa</td>
<td>Spring 1988</td>
<td>Foster City (South Bay lagoon)</td>
<td>Not stated, probably high</td>
<td>Unknown</td>
<td>Greenberg et al. 1996</td>
</tr>
<tr>
<td><em>Acanthomysis aspera</em></td>
<td>Mysid</td>
<td>August 1992</td>
<td>Suisun Bay</td>
<td>Not stated</td>
<td>Korea, Japan</td>
<td>Modlin and Orsi 1997</td>
</tr>
<tr>
<td><em>Hyperacanthomysis longirostris</em></td>
<td>Mysid</td>
<td>July 1993</td>
<td>Suisun Bay</td>
<td>Not stated</td>
<td>Korea, Japan?</td>
<td>Modlin and Orsi 1997</td>
</tr>
<tr>
<td><em>Tortanus dextrilobatus</em></td>
<td>Calanoid copepod</td>
<td>August 1993</td>
<td>Suisun Bay</td>
<td>3.6</td>
<td>China</td>
<td>Orsi and Ohtsuka 1999</td>
</tr>
<tr>
<td><em>Limnoithona tetraspina</em></td>
<td>Cyclopoid copepod</td>
<td>September 1993</td>
<td>Suisun Bay</td>
<td>1–3.8</td>
<td>China</td>
<td>Orsi and Ohtsuka 1999</td>
</tr>
<tr>
<td><em>Acartiella sinensis</em></td>
<td>Calanoid copepod</td>
<td>October 1993</td>
<td>Suisun Slough</td>
<td>2.8–4.6</td>
<td>China</td>
<td>Orsi and Ohtsuka 1999</td>
</tr>
</tbody>
</table>
today is a net sink for phytoplankton, organic matter, and zooplankton (Jassby 2008; Kimmerer and Thompson 2014; Kayfetz 2014). Biomass accumulation of phytoplankton can be controlled by grazing. During spring–fall, the grazing rate attributed to different taxa is usually highest for clams (Lucas et al. 2002; Lucas and Thompson 2012; Kimmerer and Thompson 2014), followed in declining order by microzooplankton (York et al. 2011) and mesozooplankton (Kimmerer and Thompson 2014). Phytoplankton biomass is higher in both the freshwater Delta and the seaward higher-salinity areas than in the LSZ, implying that grazing is depressing phytoplankton biomass in the LSZ and the upstream and downstream areas are providing a spatial subsidy to the LSZ (Figure 4) (Kimmerer and Thompson 2014). An excess of bacterial carbon demand over primary production in the LSZ implies a subsidy of organic carbon from the Delta or Suisun Marsh (Hollibaugh and Wong 1996). Mortality of P. forbesi and other copepod nauplii from P. amurensis grazing in the LSZ is high despite strong escape responses (Kimmerer 2015; Kimmerer and Lougee 2015), and is partially compensated for by a spatial subsidy of copepods from the Delta (Durand 2010; Kayfetz 2014). There has also been a shift from largely herbivorous and omnivorous to more predatory zooplankton species (Orsi and Ohtsuka 1999; Kratina and Winder 2015), which added an additional trophic step to the food web, further decreasing its efficiency and exacerbating food scarcity for pelagic fishes. Phytoplankton, organic matter, and zooplankton from the more productive northern channels of Suisun Marsh may provide part of the seaward subsidy, although long residence time and shallow water may limit the extent of the subsidy if clam grazing is a significant factor in Suisun Marsh (Lucas and Thompson 2012, see “Suisun Marsh” section).

**Does Ammonium Limit Primary Production?**

Here, we briefly discuss the potential role of ammonium in suppressing phytoplankton blooms, and whether this might influence higher trophic levels. This topic and other nutrient issues in the Delta and Suisun Bay are addressed in detail elsewhere (Dahm et al., submitted). The theory known as the “ammonium paradox” (Wilkerson et al. 2006, 2015; Dugdale et al. 2007, 2012, 2013) asserts that some phytoplankton can grow faster on nitrate than on ammonium, and therefore inhibition of nitrate uptake by ambient ammonium concentrations enhanced by wastewater discharge can prevent the development of blooms.

This theory has generated considerable controversy among the estuarine research community, partly because experimental work has shown that phytoplankton growth rates can be faster on ammonium than on nitrate in some phytoplankton species (e.g., Dortch 1990; Clark and Flynn 2000). However, the “phytoplankton” is composed of a
very phylogenetically and physiologically diverse group of organisms, making it difficult to generalize statements about the effects of ammonium. In addition, some reports that claim to refute the theory have instead fueled the controversy; for example, Collos and Harrison (2014) examined ammonium toxicity, which is not relevant for this theory. Esparza et al. (2014) examined phytoplankton community composition in a small channel where ammonium concentrations were generally far above the threshold theorized to affect phytoplankton growth, but did not address the response of estuarine phytoplankton to low ammonium concentrations, so those results provide no insights into this theory.

Ammonium may play a role in modulating spring diatom blooms in the Delta and Suisun Bay when clam biomass is low (Dugdale et al. 2012, 2013; Wilkerson et al. 2015). However, clam biomass and grazing rate are usually high enough from late spring to autumn to prevent bloom formation or persistence (Kimmerer and Thompson 2014). During the summer, increased ammonium loads appear to contribute to Microcystis aeruginosa blooms (Lehman et al. 2015a).

Moreover, the available data do not support an assertion (Glibert et al. 2011) that the ammonium effect has had a substantial long-term influence on higher trophic levels. For example, chlorophyll concentrations increased in the Delta during the 2000s when ammonium loading and concentrations were increasing (Jassby 2008). Further, phytoplankton biomass, diatom production, zooplankton abundance, and abundance of several fish species resident in the LSZ declined in a brief (1 month to 2 years) period after the spread of the clam P. amurensis, from mechanisms that are well understood (Cloern and Jassby 2012) and described above. Even if improved wastewater treatment reduced ammonium loading and increased phytoplankton growth rate, the additional production may be consumed mostly by clams and fail to reach species of concern. An upgrade of the largest wastewater treatment plant and greatest ammonium point source (Jassby 2008) in the Delta from secondary to tertiary treatment is currently under way, and reductions in ammonium (and nitrate) discharge into the Sacramento River at Freeport will start in 2019. This provides an excellent opportunity to greatly reduce uncertainties and controversies about the effects of ammonium discharge on the Delta and Suisun Bay food webs.

**Harmful Algal Blooms**

*M. aeruginosa* blooms have been widespread in freshwater and brackish regions in the Delta since 1999. The blooms appear to originate in the San Joaquin and Old rivers, based on the high abundance of *M. aeruginosa* (Lehman et al. 2015a). *Microcystis* species produce hepatotoxic microcystins (Carmichael 1996; Chorus and Bartram 1999; Paerl 2008), and recent research has documented microcystins in zooplankton, amphipods, and fishes (Lehman et al. 2010a). Harm to human and livestock health is also a concern. Because the POD occurred soon after the first observations of *M. aeruginosa* blooms, questions quickly arose about the contribution of these blooms to the POD and to the ongoing poor recruitment of Delta Smelt (IEP–MAST 2015). As a result, considerable effort has been invested to understand the role of the blooms in Delta ecology (Dahm et al., submitted).

Recent research in the estuary has been equivocal regarding the role of *M. aeruginosa* in causing toxic effects to food webs and organisms. Several species of fish suffered sublethal effects when fed *M. aeruginosa* as part of their diet (Acuña et al. 2012a, 2012b). Laboratory studies found that survival of the copepods E. affinis and P. forbesi was reduced by dissolved microcystin, but the concentrations tested were unlikely to occur in the field (Ger et al. 2009). However, survival of both E. affinis and P. forbesi was reduced when *M. aeruginosa* exceeded 10% of their diet (Ger et al. 2010). In contrast, survival and reproduction of P. forbesi were much less affected when the water used in experiments was obtained from the field, so that it contained a natural assemblage of microorganisms (DuMais 2014).

In addition to releasing cyanotoxins, *M. aeruginosa* blooms can inhibit production of other phytoplankton. *M. aeruginosa* contain gas vesicles that allow them to float to the surface of the water. During periods of high temperature, long water residence time, and low wind speed, *M. aeruginosa* can accumulate at the surface, forming dense mats that shade underlying non-buoyant phytoplankton, which reduces the diversity of the phytoplankton assemblage.
through competition for light (Paerl 1988; Paerl and Huisman 2008). Lehman et al. (2015a) showed that \textit{M. aeruginosa} is able to grow on the high concentrations of ammonium in the Delta. The ability of \textit{M. aeruginosa} to use available ammonium may be an important factor that contributes to the magnitude and frequency of blooms. More detail on Delta nutrient dynamics is provided by Dahm et al. (submitted).

Other cyanobacteria are also present in the Delta, but have been less well studied. Lehman et al. (2010a) found \textit{M. aeruginosa} blooms co-occurred with the cyanobacterium \textit{Aphanizomenon flos-aquae}. During the wet year of 2011, \textit{M. aeruginosa} did not form blooms until late in the year (DuMais 2014), and \textit{A. flos-aquae} and \textit{Dolichospermum} spp. (formerly \textit{Anabaena} spp.) were abundant (Kurobe et al. 2013). Like \textit{Microcystis} spp., \textit{A. flos-aquae} produces toxins that may impair the Delta food web, but unlike \textit{Microcystis} spp., \textit{A. flos-aquae} can fix nitrogen gas. Nitrogen fixation can allow cyanobacteria, which include some harmful algal bloom (HAB) species, to outcompete other phytoplankton species (Schindler 1977). The roles of these cyanobacteria in the Delta food web are unknown.

As long as conditions remain favorable—such as high nutrients, water temperatures, and long residence times—and factors driving those conditions continue to occur—such as climate change, water management, and drought (Paerl and Huisman 2008, 2009)—HABs are likely to continue indefinitely. Nearly 2 decades after they were first observed, HABs are still not routinely monitored quantitatively in the estuary, although qualitative categorical observations (e.g., none, low, medium, high, scum or mats) are made during routine fish and discrete water quality monitoring (IEP–MAST 2015). Quantitative monitoring should be developed and implemented so blooms and their effects on food webs can be better understood.

\textbf{Under-Studied Components of the Pelagic Food Web}

When the Interagency Ecological Program (IEP) zooplankton monitoring program began in 1972, the dominant model of the food web was the linear food chain from phytoplankton through zooplankton to fishes. Consequently, the program was designed to focus on larger zooplankton and phytoplankton. We now know that estuarine pelagic food webs are much more reticulate than this, and that non-photosynthetic microbes are important consumers of phytoplankton and organic matter. Microzooplankton such as ciliates and flagellates overlap in size with phytoplankton, so that their grazing on phytoplankton must be inferred indirectly from experiments (Calbet and Landry 2004; York et al. 2011). Many microbes are mixotrophs, contributing to both primary and secondary production (Flynn et al. 2012).

Archaea and bacteria mediate nearly all biogeochemical processes (Mosier and Francis 2008), and their potentially high production can contribute to higher trophic levels, often through a microzooplankton trophic step (Ederington et al. 1995). The high bacterial carbon demand in Suisun Bay, discussed above, implies a carbon subsidy from other areas and potentially high production of organic matter available to higher trophic levels. In open waters, this carbon source appears to be smaller than that arising directly from phytoplankton (Sobczak et al. 2002, 2005; Mueller–Solger et al. 2002). Similar analyses of carbon sources and processing have not been done for tidal wetlands or tidal marsh channels.

The IEP zooplankton monitoring program has collected pump samples for microzooplankton between 43 and 150 µm and has reported abundance of rotifers, copepod nauplii, and adults of small species such as the copepod \textit{Limnoithona} spp. (Orsi and Mecum 1986; Ferrari and Orsi 1984). However, until 2008, sub-samples were taken for processing that represented \(<2 L of sample volume (IEP–MAST 2016), such that the counts of copepods and some other taxa in the pump samples were often too low for reliable abundance estimates. Starting in 2008 the effective volume sampled was increased to provide more reliable counts.

The abundance of non-photosynthetic protists (e.g., ciliates and heterotrophic flagellates) and all metazoan zooplankton smaller than 43 µm, has never been routinely monitored in the estuary. Several short-term studies have shown their importance as grazers on phytoplankton (York et al. 2011;
Kimmerer and Thompson 2014) and as food for some zooplankton (Rollwagen Bollens and Penry 2003; Bouley and Kimmerer 2006; Gifford et al. 2007) and clams (Greene et al. 2011). These roles are well known from other marine, estuarine, and freshwater locations (e.g., Calbet and Landry 2004), and the continued neglect of microzooplankton reflects the persistence of the outmoded 1970s conceptual model of a linear food chain that guided the initial design of the long-term monitoring programs in the estuary. It also reflects the difficulty of changing legally mandated, long-term monitoring programs and adding additional components to already expensive monitoring surveys.

Because small metazoa, protists, bacteria and archaea are key ecosystem components, monitoring their abundance and distribution would likely reveal important processes and sources of variation that are currently unknown.

NEW INTEREST IN NON-PELAGIC FOOD WEBS

Tidal Wetlands

By the time scientific data collection began, tidal wetlands were a relatively minor component of the landscape in the Delta (Table 1), and much of the tidal wetland in Suisun Marsh had been converted to managed wetlands (Manfree 2014). Consequently, there are relatively few studies that illustrate food web function in tidal wetlands. Brown (2003) reviewed and summarized the available information but because of the lack of information reached few conclusions on the importance of tidal wetland habitats to fishes. Today, understanding food web processes in tidal wetlands and their interaction with other food webs has gained scientific attention because tidal wetland restoration has been identified as a management action for improving habitat and food web processes for fish species of concern (e.g., EcoRestore c2016). We discuss this new information here, and the potential role of tidal wetlands in subsidizing the proximate open-water food web later.

Stable isotope data have shown that tidal marsh production supports resident species and is consumed by transient species (Grimaldo et al. 2009b; Howe and Simenstad 2011; Schroeter et al. 2015; Young 2016). Juvenile salmon feed in shallow areas throughout the estuary including small remnant wetlands (Grimaldo et al. 2009b). Juvenile salmonids in other Pacific Coast estuaries access tidal marsh for feeding and rearing (Simenstad and Cordell 2000; Bottom et al. 2005); however, it is less clear that pelagic fishes such as Delta Smelt make extensive use of tidal marsh. Delta Smelt probably used tidal channel habitat in the pre-Gold Rush Delta, when tidal channels were a dominant habitat type (Whipple et al. 2012; Robinson et al. 2014), but sampling in the Delta suggests that Delta Smelt are now relatively rare in small tidal channels (Gewant and Bollens 2012). Research from elsewhere in the estuary indicates that channel order and geomorphology can affect how different species of fish use tidal channels (Visintainer et al. 2006). The flooded Liberty Island in the northern Delta is used by Delta Smelt and other fishes (Whitley and Bollens 2014), but it is dominated by open water, with emergent tidal marsh mostly limited to its northern end. However, the fishes captured from that area do have tidal wetland-associated organisms in their diets at times.

Although the ability of transient and resident fishes to feed within tidal wetlands is well established, the dynamics of lower trophic-level organisms are not. Cohen et al. (2014) measured primary production of phytoplankton, benthic microalgae, low marsh, and submerged aquatic vegetation (SAV), in four tidal marshes in the lower Napa and Petaluma rivers of northern San Pablo Bay and in two small tidal marshes in the western Delta (only one had SAV). Primary production per unit area decreased with producer group in this order: low marsh vegetation > SAV > phytoplankton > benthic microalgae. However, relative contributions of the different producer groups varied from site to site, based on the areal coverage of each producer group. Cohen et al. (2014) suggested ammonium inhibition and benthic grazing were possible reasons for low primary production in the Delta wetlands compared to the other sites. A study in fall 2011 (a wet year) near the confluence of Suisun Marsh with Grizzly Bay, indicated highly variable carbon sources among consumers ranging from clams to Striped Bass (Schroeter et al. 2015). Emergent aquatic vegetation and phytoplankton were commonly dominant sources, and SAV was a substantial contributor for amphipods, California bay shrimp, and Striped Bass.
The addition of introduced species (specifically polychaete worms, shrimp, and amphipods) increased food web complexity, illustrated by isotopic differences from their native counterparts (Schroeter et al. 2015). Similarly, Young (2016) explored food web processes in Lindsey and Cache sloughs in the north Delta. Contributions to consumers of various primary producers, including emergent and submerged vegetation, epiphytic green algae, benthic diatoms, and phytoplankton, were variable seasonally and spatially, and highly dependent on local conditions. The results of Schroeter et al. (2015) and Young (2016) show that there is no single food web structure for tidal wetlands. Instead, food web structure is likely to vary in space and in time, with contributions from multiple primary sources.

Zooplankton abundance in the large channels of Suisun Marsh is similar to that in Suisun Bay. Few differences were detected in zooplankton communities between tidal marshes at Browns Island and Sherman Lake and nearby open waters (Bollens et al. 2014). Although this implies that existing tidal marshes do not support a distinct zooplankton community because of high exchange rates with pelagic habitats, the more remote sloughs of northern Suisun Marsh and the northern Delta may harbor higher abundance because of long residence time.

**Submerged Aquatic Vegetation**

There is no historical information on the extent of submerged aquatic vegetation (SAV) in shallow subtidal habitat in the Delta–Suisun region. There are native aquatic macrophytes, sago pondweed *Stuckenia pectinata* and widgeongrass *Ruppia maritima* (Whitcraft et al. 2011), so SAV habitat was certainly present. Interest in SAV was stimulated by the observation that a number of introduced species, particularly Largemouth Bass *Micropterus salmoides*, were increasing in abundance as SAV habitat expanded (Brown and Michniuk 2007), with the SAV composed primarily of introduced Brazilian waterweed *Egeria densa* (Hestir 2010). Brown and Michniuk (2007) noted increased occurrence of SAV in the Delta between the early 1980s, when it was first perceived as a problem (Hestir 2010), and the early 2000s. Recent assessments have indicated fluctuation in SAV extent in the Delta within the 2000s (Table 1). During peak SAV biomass in fall, Brazilian waterweed can cover nearly 60% of central Delta waterways (Santos et al. 2011) and 6% to 11% of the total area of Delta waterways (Santos et al. 2016). Conversely, in the brackish Suisun Marsh and Bay regions, the submerged macrophyte community is composed primarily of the natives. The reduced abundance of introduced aquatic macrophytes in brackish areas is likely a result of a low tolerance for salinity (e.g., Borgnis and Boyer 2016).

Brazilian waterweed makes up the vast majority of the SAV biomass in the freshwater Delta because of its bimodal growth pattern, which allows it to overgrow other SAV species in the fall (Santos et al. 2011), and its ability to colonize under low-light conditions (Yarrow et al. 2009) unfavorable to native SAV species (Santos et al. 2012). The invasion and spread of Brazilian waterweed established a new food web. Aquatic macrophytes provide structural complexity and surface area, expanding habitat for epiphytic algae, invertebrates (Schultz and Dibble 2012), and fishes (Brown and Michniuk 2007; Grimaldo et al. 2009b, 2012; Conrad et al. 2016).

A diet and stable-isotope study indicated that nearshore fishes in the Delta (mostly centrarchids) consumed mainly amphipods living on the SAV, thereby gaining energy from SAV and associated epiphytic macroalgae (Grimaldo et al. 2009b). Fishes inhabiting open water (non-vegetated) shoals such as Chinook Salmon and Mississippi Silversides *Menidia beryllina* consumed some food organisms supported by SAV production and may have been foraging in the SAV on high tides. Non-native red swamp crayfish *Procambarus clarkii* may have also increased in abundance because of an increase in suitable habitat within Brazilian waterweed beds, and may also provide an important food source to Largemouth Bass (Nobriga and Feyrer 2007). The role of red swamp crayfish in the SAV food web and its interaction with SAV is an under-studied but important topic, given that the crayfish can affect water quality, sediment resuspension rates, macrophyte growth, and nutrient dynamics in other systems (Matsuzaki et al. 2009).

Foraging in the SAV by fish species normally more abundant in unvegetated shoals or open waters may
provide a path of energy exchange between the SAV and pelagic habitats (Grimaldo et al. 2009b). Mississippi Silversides typically inhabit open-water habitats near the shoreline, but also occur at lower densities offshore (Grimaldo et al. 2012; IEP–MAST 2015). The dietary shift in fishes after the invasion of *P. amurensis*, discussed above, included an increase in consumption of amphipods (Feyrer et al. 2003), which are supported by SAV-based energy sources (Grimaldo et al. 2009b). Thus, food resources originating in SAV may provide some support to fish populations outside of SAV beds. Similarly, as noted above, stable isotope studies in sloughs within areas of tidal wetlands that also contain SAV have shown that the SAV can be an important source of organic matter to consumers in those channels (Schroeter et al. 2015; Young 2016).

*E. densa* is considered an “ecosystem engineer” (Jones et al. 1994), which can affect other food webs and species through its effects on physical habitat. *E. densa* beds create drag and slow-moving water, reducing turbulence, and increasing net deposition of sediment, thereby increasing water clarity (Yarrow et al. 2009). Hestir et al. (2015) found that, 21% to 70% of the increase in water clarity in the Delta from 1975 to 2008 can be attributed to SAV expansion. The dense canopies of *E. densa* beds shade the water column, contributing to light limitation of phytoplankton and benthic algae. Research from shallow lakes has shown that *E. densa* has seasonal effects on chlorophyll-*a* concentrations, with lower concentrations inside plant beds in spring and fall (Mazzeo et al. 2003). The spread of *E. densa* may also have important but unquantified effects on water-column nutrients; however, the relationship between submerged macrophytes and nutrients is likely complex, because they can obtain and store nutrients from both the sediment and the water column (Boyer and Sutula 2015; Dahm et al., submitted).

Less is known about the food-web effects of invasive floating aquatic vegetation (FAV). Both native (pennywort *Hydrocotyle umbellata*) and introduced (water hyacinth *Eichhornia crassipes* and water primrose *Ludwigia* spp.) species are present in the Delta–Suisun region. These floating species form dense mats at the water surface that limit light penetration, which in turn may limit growth and establishment of SAV (Khanna et al. 2012), as well as growth of phytoplankton and benthic algae. Dissolved oxygen and densities of epibenthic and benthic invertebrates were lower beneath water hyacinth than beneath native pennywort mats in three flooded islands in the Delta (Toft et al. 2003). Senescent plant material beneath the floating mats can create anoxic conditions (Malik 2007); large-scale mechanical shredding of water hyacinth as a control measure is associated with localized fish kills, presumably because of the low dissolved oxygen associated with decomposition (Greenfield et al. 2007). Research on the effects of FAV and associated control measures on food webs, including the potentially allelopathic effects of FAV on phytoplankton (Schultz and Dibble 2012; Shanab et al. 2010), is extremely limited and deserves increased attention.

**Floodplain**

Similar to other historic habitats, only remnants of seasonally flooded habitats remain (Table 1). Historically, seasonally inundated wetlands included small floodplains along many tributaries to the estuary and large flood basins to its north. The Yolo and Sutter flood bypasses—as well as the Sacramento Deep Water Ship Channel and the northern reaches of the Cache Slough complex in the north Delta—cross and encompass large parts of the historical flood basins (Whipple et al. 2012). Although the bypasses are used mostly for agriculture and flood flow conveyance, they also function as managed flood plains. Smaller managed or restored flood plains also exist in the Delta, for example along the Cosumnes River, a small eastside tributary to the Delta.

The Yolo Bypass has been well studied for the benefits it provides to fishes that use it and for potential subsidies to the north Delta. In 2003, the Yolo Bypass contributed 14% (median) of the phytoplankton biomass (chlorophyll-*a*) exported to the estuary, with only 3% of the river flow through the combined Sacramento River and Yolo Bypass (Lehman et al. 2008). This effect mainly resulted from higher phytoplankton growth rate and longer residence time in the Bypass than in the river (Lehman et al. 2008). However, the actual chlorophyll-*a* flux and its contribution to Delta food webs depend on the extent, duration, and
timing of flooding. Recent data have suggested that increased flows through Yolo Bypass during the fall rice field drainage are associated with increased chlorophyll-\(a\) concentrations at Rio Vista (Figure 5). Flooding of Yolo Bypass stimulates rapid production of chironomids, which aestivate in floodplain sediments (Benigno and Sommer 2008). Invertebrate drift in the Yolo Bypass exceeds that observed in the Sacramento River (Sommer et al. 2001, 2004) and the chironomids are a major food source for juvenile salmon (Sommer et al. 2001) in the Bypass. Conversely, there was no difference between floodplain and river in densities of zooplankton, likely because of the low residence time of the water. Similar food web benefits have been documented in the Cosumnes River floodplain, including high phytoplankton, invertebrate, and fish productivity within the floodplain, and biomass exports to adjacent habitats (Ahearn et al. 2006; Grosholz and Gallo 2006; Moyle et al. 2007; Jeffres et al. 2008).

Of course, floodplain habitats are not immune to the effects of species invasions and other stressors. For example, the Siberian prawn became the dominant macroinvertebrate in the perennial ponds and waterways of Yolo Bypass just 1 year after it was first observed (Brown and Hieb 2014). Its ecological effects have not yet been studied in detail, but food web effects are likely. In the Cosumnes River Preserve, which includes restored floodplain and riparian habitat, the highly invasive herb perennial pepperweed *Lepidium latifolium* and non-native redeye bass *Micropterus coosae* pose considerable management challenges (Moyle et al. 2003).

![Figure 5](image-url)  
*Figure 5* Chlorophyll-\(a\) concentration, measured as relative florescence units (RFU), in the Sacramento River at Hood (SRH; just south of Freeport; see Figure 1) and the Sacramento River at Rio Vista (RVB) in relation to flow in cubic feet per second (cfs) through Yolo Bypass measured at Lisbon Weir (Lisbon Flow)
REGIONAL VARIATION IN FOOD WEBS ACROSS THE LANDSCAPE

Earlier, we have largely discussed habitat-specific food webs as individual entities; however, across the landscape there is variation in the types and degree of connection among habitat-specific food webs in specific geographic regions (Figure 2). Understanding these geographic differences is important when management strategies and expected outcomes for different parts of the Delta–Suisun system are considered. We omit the LSZ from this discussion because it has been treated as a specific region throughout this paper.

Suisun Marsh

Fishes (Moyle et al. 1986; Meng et al. 1994; Matern et al. 2002; O’Rear and Moyle 2014a, 2014b) and their diets (Feyrer et al. 2003; O’Rear 2012) have been well-studied in Suisun Marsh; however, lower trophic-level function has not been well studied, with the exception of invasive jellyfish (Schroeter 2008; Wintzer et al. 2011b, 2013). Within the context of our conceptual model (Figure 3), Suisun Marsh mainly represents the interaction of a tidal wetland food web, as represented by the smaller sloughs, with the pelagic-benthic food web of the larger channels (Suisun and Montezuma sloughs). The tidal wetlands component includes a perennial connection of smaller Suisun Marsh channels with “fringing marshes” along the edges and small areas of tidal wetland (O’Rear and Moyle 2014a). The smaller sloughs have higher residence times, higher concentrations of phytoplankton and zooplankton, and lower abundances of \( P. \textit{amurensis} \) compared to the larger sloughs (O’Rear and Moyle 2014a). \( P. \textit{amurensis} \) is most common in upper Suisun Slough, with most of the remaining clams captured in the region of lower Suisun Slough just upstream of the confluence with Grizzly Bay. The interaction between the benthic and pelagic food webs in the larger sloughs appears to be similar to that described in the LSZ.

Seasonal draining of managed wetlands affects Suisun Marsh food webs both positively and negatively. In some cases, draining the ponds can flush large numbers of small fish, such as Threespine Stickleback \( \textit{Gasterosteus aculeatus} \), that can be consumed by predatory fish such as Striped Bass (O’Rear and Moyle 2014a). Conversely, under some conditions, the decomposing organic matter in such drainage water can deplete the oxygen in sloughs, killing fish and reducing habitability of portions of some sloughs (Siegel et al. 2011); however, this situation appears to be improving with management changes (O’Rear and Moyle 2013). The value of the discharged organic matter to the food web is unknown, but it likely includes a great deal of refractory material that is less biologically available to the ecosystem than fresh material such as phytoplankton (Sobczak et al. 2002, 2005).

The small- and large-channel food webs appear to change on several time-scales. O’Rear and Moyle (2013) have noted that over the course of their surveys (started in 1980) fishes have used nearshore habitats for feeding as mysid abundances decline seasonally, consistent with the hypothesis of shifts to nearshore areas and smaller channels for feeding on amphipods. Schroeter (2008) found no change in Striped Bass abundance in small channels and shallow areas, compared to declines in deeper areas of large sloughs.

The relative importance of various primary producers to Suisun Marsh food webs has been little studied. Schroeter et al. (2015) conducted a stable isotope study in lower Suisun Slough, but the study was limited to a few sites during the fall of a single wet year. Primary producers included phytobenthos, SAV, emergent aquatic vegetation, and terrestrial vegetation. Consumers tended to be generalists, incorporating a mixture of primary producer sources into their diet, even those generally thought of as primarily using phytoplankton (e.g., \( \textit{C. fluminea} \) and mysids). A study with broader geographic and temporal scope would be useful to understanding how Suisun Marsh functions, and could provide important information for designing restoration projects and projecting outcomes.

Freshwater Delta

For the following discussion, we divide the freshwater Delta into two regions: We define the south-central Delta as the region of the Delta south of the Sacramento River (Figures 1 and 2) and the north Delta as the area north of the Sacramento River,
emphasizing the Cache Slough complex and Liberty Island, as well as the Sacramento Deep Water Ship Channel and Yolo Bypass.

South–Central Delta

Two food webs dominate the south–central Delta: the pelagic-benthic and SAV. Floating aquatic vegetation (FAV) is also abundant, but only a single study (Toft et al. 2003) has explored the food web effects of FAV. Therefore, we acknowledge that FAV may have large effects, but there are insufficient studies for synthesis. This region also receives a seasonal floodplain subsidy from the Cosumnes River restored flood plain. Studies have shown that the Cosumnes River flood plain can provide important spawning and rearing habitat and food web resources for native fishes (Ahearn et al. 2006; Grosholz and Gallo 2006; Moyle et al. 2007; Jeffres et al. 2008); however, the degree to which this production is transported into the tidal Delta has not been explored. Later, we further address his general topic of floodplain transport to the freshwater Delta (see “North Delta”).

Studies in the south-central Delta have been critical to our understanding of clam grazing effects, particularly *C. fluminea*. Studies in the open waters of several flooded islands, e.g., Franks Tract and Mildred Island (Figure 1), documented how grazing by *C. fluminea* could deplete phytoplankton biomass from these areas, and that such effects were situation dependent (Lucas et al. 2002; Lopez et al. 2006). Franks Tract, which is shallow and well-mixed, and supported a population of clams, was consistently a sink for phytoplankton. Conversely, Mildred Island, which was deep and had limited connection to surrounding channels through a few breaches, did not support a large population of clams, and could accumulate phytoplankton biomass; however, the biomass was largely consumed by *C. fluminea* when it was transported out of the island through shallow channels where *C. fluminea* was abundant. These results have been the basis for much of the thinking about flux of organic matter from habitat restoration areas to pelagic habitats (e.g., Lucas and Thompson 2012; Herbold et al. 2014).

Studies in the south-central Delta have also influenced current understanding of SAV, which has invaded channel edges and other shallow habitats. These studies have highlighted changes in the fish community (Brown and Michniuk 2007; Grimaldo et al. 2012; Conrad et al. 2016). This new habitat has also established new food webs (Grimaldo et al. 2009b; Young 2016). Grimaldo et al. (2009b) determined that pelagic phytoplankton, and SAV and associated epiphytic macroalgae, fueled largely separate food webs; however, there was some limited crossover. Invertebrates produced within the SAV, such as amphipods and chironomid pupae, were consumed by fishes, such as Chinook Salmon and Mississippi Silverside, feeding along the edge of SAV beds.

In addition to *C. fluminea* grazing and habitat change, this region is subject to several other possible food web modifications. HABs are generally centered in this region, indicating that organisms are exposed to greater concentrations of these cyanobacteria and their toxins for longer periods of time. Also, there has been periodic concern that water exports from the water projects in the south Delta may entrain large quantities of nutrients and lower trophic-level organisms (Jassby and Powell 1994; Arthur et al. 1996; Durand 2015). Management concerns over entrainment have focused mainly on entrainment of fish (Kimmerer 2008; Brown et al. 2009; Grimaldo et al. 2009a). The effects of diversions on nutrients and lower trophic levels would require consideration of the timing and duration of diversions, hydrodynamics and interactions with *C. fluminea* and the SAV food web.

North Delta

The north Delta has been of much recent interest because of its potential for extensive habitat restoration and because of high frequency of occurrence of Delta Smelt (Sommer and Mejia 2013). Lindsey and Cache sloughs have relatively intact terminal channel networks (Figure 1) that have tidal marsh, backwater, and SAV habitats (Young 2016); however, there is a water diversion near the upper end of Lindsey Slough. Liberty Island provides a large area of open water pelagic habitat. The region provides the pathway from the Yolo Bypass to the Delta, and thus receives episodic inputs of organic material and sediment when Yolo Bypass floods and drains. Invasive species, particularly SAV (Young 2016) and *C. fluminea* (Kramer–Wilt 2010), appear to
be less prevalent than in the south-central Delta. SAV and FAV are present and can be abundant in the channel networks, but the distribution of SAV may be limited by turbidity (Young 2016). The role of C. fluminetia in north Delta food webs has not been fully evaluated. Overall, the area appears to provide some of the best remaining habitat for native fishes (Moyle et al. 2012; Young et al. 2015), including Delta Smelt (Hammock et al. 2015).

Young (2016) conducted stable isotope analyses in upstream and downstream sites in the channels of Lindsey and Cache sloughs of the north Delta in spring and summer. Pelagic species in the north Delta channels consumed more organic matter of non-phytoplankton origin than those in the south-central Delta. The high degree of separation between pelagic and SAV food webs observed in the south Delta was most similar to conditions in the downstream channel reaches of Cache and Lindsey sloughs, where channels have rip-rapped banks, limited emergent vegetation and strong tidal exchange of water. This suggests that concepts developed in the south-central Delta should be applied to north Delta channel food webs with caution.

Liberty Island provides an expansive area of shallow pelagic habitat that supports an array of native and invasive fishes (Nobriga et al. 2005; Whitley and Bollens 2014). Liberty Island is dominated by tidal marsh at the northern end: connections with larger channels are important at the southern end and western margins. Fish diet studies in the northern tidal marsh area indicate important contributions of tidal marsh habitat to the diets of native and invasive fishes (Whitley and Bollens 2014). Whitely and Bollens (2014) noted that Delta Smelt used tidal wetland-derived prey during some seasons although zooplankton was always dominant in the diet; they did not address diets in the southern area of Liberty Island. The Liberty Island food web deserves additional study, perhaps using stable isotopes.

**CAN AQUATIC FOOD WEBS BE MANAGED TO BENEFIT SELECTED SPECIES?**

Given the condition of the present-day Delta–Suisun food webs, formulating management actions to provide food web support to fish species of concern—particularly Delta Smelt, Longfin Smelt, and endangered salmonids—is a formidable challenge. There appear to be two major approaches to food web management. First, if primary production is limited by excessive ammonium loading, limiting such loading from wastewater discharge is a potential approach to food web management. The plan for wastewater treatment upgrades and the opportunity this presents for understanding food web processes is discussed above and by Dahm et al., submitted. A second approach to food web management is habitat restoration, particularly of tidal wetlands and flood plains, to provide new production sources for fishes to exploit.

The premise of the restoration approach is that altering areas of the current Delta landscape to more closely resemble historical conditions will restore food web processes that benefit native fishes. Specifically, the expectation for restoration of food webs are two-fold: first, restored areas will produce phytoplankton and zooplankton that will benefit species of concern on-site; and second, food resources produced in restored areas will be exported such that they will subsidize pelagic habitat in adjacent waters or regionally.

A major challenge to achieving the objectives of the restoration approach is that the highly altered state of the ecosystem makes it difficult to predict outcomes, making restoration planning more complex (Seastedt et al. 2008). Species assemblages within novel ecosystems such as the Delta–Suisun system are unique, and environmental stressors differ from those of the historical ecosystem. Thus, restoration projects and their effects on the local and regional food webs must be approached as experiments, beginning with the knowledge of current ecosystem function, having clearly expressed objectives and careful design, and including feasibly measured metrics to determine if projects are achieving stated objectives (Seastedt et al. 2008). An adaptive management framework for restoration projects in the Delta–Suisun system is required by the Delta Plan (DSC 2013) and includes all of these elements. This adaptive management framework is crucial for charting and adjusting the progress of floodplain and tidal wetland restoration projects that are intended to support food web processes.
Floodplain Restoration

Wetland and floodplain restoration efforts are likely to play out differently for food web support simply because tidal wetlands are perennial, and floodplains are only seasonally and episodically inundated, to the extent that their food web effects are likely to be more ephemeral than those of tidal wetlands. However, the data summarized above (Figure 5) suggest that Yolo Bypass can quantifiably contribute to phytoplankton populations in the north Delta and contribute to phytoplankton blooms in the lower estuary. One option for food web management is to adaptively manage Yolo Bypass flows to support the pelagic food web downstream. For example, it may be possible to strategically augment natural high-flow events (via appropriately timed reservoir releases) to achieve Yolo Bypass inundation in winter and early spring to support spring blooms of the type that were common in the historic estuary until approximately the 1970s. Similarly, winter and early-spring Yolo Bypass flooding would likely benefit juvenile salmonids by providing a food-rich habitat during their emigration to the ocean (Sommer et al. 2001). During the autumn rice harvest in the Yolo Bypass, strategic management of the perennial agricultural drainage system could achieve flows out of Yolo Bypass substantial enough to promote phytoplankton blooms downstream of the Bypass (Figure 5). If these phytoplankton blooms support enhanced zooplankton production, such management could promote food web processes to support pelagic species, particularly Delta Smelt. Thus, while floodplain support of downstream food webs is indeed likely to be ephemeral, well-timed adaptive management actions may have beneficial food web results.

Tidal Wetland Restoration

Evaluating the benefits of tidal wetland restoration to fish requires the consideration of several overlapping spatial scales. The first spatial scale is that of the tidal wetland itself; benefits provided within the tidal wetland are available to resident fishes and transient fishes that can access the site. The second spatial scale is the near-field, local scale outside of a specific tidal wetland project but within roughly a tidal excursion (the “tidal footprint” of a restored area). The third spatial scale is regional and depends on tidal dispersive transport from the tidal wetland to more distant channels and bays where food is scarce (e.g., the LSZ). At the near-field-scale, we consider in detail the narrower perspective of a subsidy of food (primarily zooplankton) for native pelagic fishes in open waters, because this is one of the major areas of uncertainty (Herbold et al. 2014). We then take a more general perspective on regional ecosystem restoration. At all scales, characteristics of the wetland itself—such as elevation, vegetation types, inundation regime, colonization by SAV and clams, and connectivity to neighboring areas—are key to its structure and to functions such as productivity, and thus to its ability to support consumer species. To determine if a project is achieving its objectives, each scale of habitat restoration will require scientific monitoring efforts of specific metrics. For example, at the wetland scale, metrics might include densities of food resources and target fish species for management. At the tidal-excursion and regional scales, measurement of plankton fluxes between restoration areas and nearby open waters over the full tidal cycle, may be appropriate. Where possible, comparison of plankton biomass in all areas before and after restoration will also be helpful to evaluate success.

At the scale of the tidal wetland, stable isotope data have clearly shown that tidal wetland production supports resident species and is consumed by transient species (see “Tidal Wetlands”). Clearly, tidal wetland restoration projects are designed to allow fishes to access the newly created habitat, but invasive SAV and FAV may impose a physical or water-quality (e.g., low dissolved oxygen) barrier to fish movement into a restoration site, and SAV will provide habitat for predators that could seriously reduce the survival of fishes of concern. Extensive beds of *E. densa* and FAV, such as water hyacinth, may limit phytoplankton production through competition for light. Studies in Suisun Marsh indicate that smaller channels that are somewhat isolated from larger channels can be very productive (see “Suisun Marsh”); however it is unclear whether individual restoration projects will be large enough to maintain such channel systems. It is also unclear how likely some fishes are to access and use tidal channels within wetland restoration projects. Research from
elsewhere in the estuary indicates that channel order and geomorphology can affect use by different fish species (Visintainer et al. 2006). This uncertainty suggests that access-relevant metrics are needed to evaluate restoration projects. Monitoring metrics for the “opportunity” offered by restored habitat have already been proposed for wetland restoration efforts in the Pacific Northwest (Simenstad and Cordell 2000).

At tidal-excursion and regional scales, the potential for the export of food organisms from restoration sites to surrounding habitats is not well understood. To be used by fish in the surrounding habitats, this export must take the form of zooplankton, non-planktonic aquatic animals (e.g., epi-benthic amphipods and chironomids), or large particles of biologically available detritus. Few studies have examined zooplankton exchange between tidal wetlands and adjacent areas. A marsh at China Camp in San Pablo Bay was a net sink for mysids, probably because of predation within the marsh (Dean et al. 2005). Liberty Island, often cited as a habitat restoration model for Delta Smelt, is a net sink for organic and inorganic material over a whole tidal cycle (Lehman et al. 2015b) and seasonally (Lehman et al. 2010b). Within Liberty Island, limited areas of wetland provide a large proportion of the total phytoplankton production, which can support food web processes locally (Lehman et al. 2015b). Zooplankton fluxes in and out of Liberty Island were sometimes inward and sometimes outward, and strongly related to the tidal cycle (Lehman et al. 2010b).

Of the handful of studies of zooplankton flux to and from shallow estuarine areas outside of the estuary, all show evidence of import and none of export except for planktonic larvae of the wetlands’ benthic residents (Carlson 1978; Kimmerer and McKinnon 1989; Brucet et al. 2005, 2010; Badosa et al. 2006; Mazumder et al. 2009; Cooper et al. 2012). Fluxes of zooplankton into shallow areas likely result from consumption of zooplankton by small fish that seek food and shelter there, which has been shown in estuaries and lakes (e.g., Fulton 1985; Cryer and Townsend 1988; Gliwicz and Rykowska 1992; White 1998; Jeppesen et al. 1998).

Although phytoplankton is not directly consumed by fishes of concern, it can support secondary production that can be consumed by fishes; however, the likelihood of such a subsidy is uncertain. Lucas and Thompson (2012) described the complex interactions among residence time, depth, and grazing rates of invasive clams, and their combined influence on phytoplankton in shallow restoration sites. The principal lesson of that study was that benthic grazing can control the sign of the relationship between residence time and biomass. Without information on all three values, the outcome of restoration actions is difficult to predict.

Wetland restoration to enhance beneficial food web processes at the regional scale has been discussed in other estuaries. In Delaware Bay, stable isotope data have shown that several marine transient fishes benefit from production that originates in salt marshes, without necessarily occupying them (Litvin and Weinstein 2003; Weinstein et al. 2014). Based on these indirect benefits, researchers have strongly advocated for adopting a “whole estuary” approach to wetland restoration, in which tidal wetlands are considered as part of a larger “habitat mosaic,” and their functional connections with adjacent habitats are considered in the restoration planning phase (Weinstein and Litvin 2016). In the Delta–Suisun system, much of the tidal wetland restoration planning occurs at the single project level. To adopt the “whole estuary” approach proposed by Weinstein and Litvin (2016), the potential production of all wetland restoration projects—and their contribution to adjacent waters within their tidal excursion—would be considered jointly and at a landscape scale. The advantages of this approach may be the identification of (1) potential food web benefits that are apparent only with the joint contributions of proximate restoration areas, and (2) the need for restoration in other areas that are not yet planned.

Regardless of the scale of the planning approach, if tidal wetlands are to be restored to support pelagic fishes, much work is required to determine how to design restoration programs that are likely to achieve that purpose. The long-term sustainability of the benefits of restoration over future decades should also be considered, given the likelihood of colonization by introduced clams, SAV, and FAV, and the anticipated use of the area by the target species. The challenges
to achieving substantial zooplankton export suggest that this objective of restoration should be approached with a careful, step-wise experimental approach that includes hydrodynamic modeling, analysis of the effects of wetland vegetation and bathymetry on zooplankton production, and behavioral studies of zooplankton and their predators. In addition, studies of phytoplankton and nutrient fluxes will be necessary to determine if they support zooplankton production in open waters, or merely provide more food for clams.

**INTO THE FUTURE: EXPECTATIONS FOR CONTINUED CHANGE AND SCIENCE NEEDS**

Continued changes in the Delta food web should be expected. Although the pace of species introductions has declined, species invasions still occur in the estuary and its watershed. A likely future invertebrate invader is quagga mussel, which has already invaded several reservoirs in California. Quagga mussels and closely related zebra mussels can have effects on aquatic ecosystems at least as severe as that of *P. amurensis* in the estuary (Strayer 2009). The high calcium requirements of quagga and zebra mussels may limit their distributions to areas outside the influence of the Sacramento River (Cohen 2008). The ultimate distribution and severity of the food web effects of such an invasion are difficult to predict, especially given the high current grazing rates of clams.

Successful zooplankton introductions into the estuary have been associated with drought (Winder et al. 2011). Drought conditions are often associated with higher temperatures, salinity intrusions deep into the Delta, and low flows that could favor newly arriving species over native species, although abundance, growth, and reproductive rates of copepods in the estuary do not respond to changes in freshwater flow (Kimmerer et al. 2014). Also, the analysis of Winder et al. (2011) used a 3-year lag between drought and establishment, which is very long for species with life cycles of a few weeks. Ballast-water regulations that require at-sea exchange have been reasonably effective at limiting the supply of potential invaders (Choi et al. 2005), and no new zooplankton invasions have been documented. Further work on the conditions that favor introductions of zooplankton and other organisms would help clarify what has happened in the past, and provide information that would inform expectations for the future.

The expected effects of climate change on the Delta are addressed by in detail by Dettinger et al. (2016). Ecologically, the effects of climate change in the watershed (Yates et al. 2008; Cloern et al. 2011; Thompson et al. 2012; Null et al. 2013) and the estuary (Cloern et al. 2011; Brown et al. 2013, 2016) have largely been considered on a species-specific basis. Statewide, native species appear more susceptible to climate change than invasive species (Moyle et al. 2013). There have been several assessments of tidal marsh sustainability in the context of estimated rates of sea level rise (Orr et al. 2003; Stralberg et al. 2011; Swanson et al. 2013), but the outcomes have generally not been considered in the context of food webs.

The effects of sea level rise and concurrent salinity intrusion on food webs must be examined in the context of local topography and land use, regional human activities including management, and the vulnerability of Delta levees to failure (Mount and Twiss 2005). If tidal and seasonal marshes are given room to migrate landward as sea level rises, changes to the landscape and associated food web processes would likely be relatively minimal. If edges are hardened with seawalls or levees, sea level rise will more likely result in marsh drowning and the conversion of tidal marsh to subtidal habitats, with accompanying changes in trophic processes. Salinity intrusion would likely little affect food web processes in the open waters of the estuary because pelagic species move with the water, and other species move with the salinity gradient, either actively or through death and colonization. However, the geographic extent and location of waters with a specific salinity, such as the LSZ, would change, with possible effects on species abundance. Efforts to model and predict the outcomes of changes in aquatic habitat from climate change would help to indicate where habitat conservation and restoration should be focused.

The uncertainties in predicting the effects of climate change, the outcomes of management actions, and the occurrence of unexpected events such as levee failures should not be interpreted as a reason not to proceed with efforts to improve conditions for species
of concern. The continued development of modeling capacity (e.g., Cloern et al. 2011; Rose et al. 2013; Kimmerer et al. 2014b) and ecological understanding (IEP–MAST 2015; Kimmerer and Thompson 2014) can be applied to scenarios of future change (e.g., Cloern et al. 2011). Combined with monitoring and research in an adaptive management framework, implementing new management actions should improve our ability to understand what the future Delta and its food web will look like.

CONCLUSION

Over the last 2 decades our understanding of the structure and changes in Delta–Suisun food webs has improved, but the relative importance of various ecosystem drivers in causing those changes is unclear. These uncertainties urgently need to be resolved to support effective management actions that will halt or reverse the declines in species abundance and otherwise improve aquatic habitat. Management of a few drivers is clearly more feasible than simultaneous management of a large number of drivers; however, the quest for a ranking of drivers or selection of a single “key” driver is futile because they all matter at different times, in different places, and in different and often interactive ways (NRC 2012; IEP–MAST 2015).

Below, we identify the topics that we believe merit the initiation of research programs to identify effective management approaches that will support beneficial food webs. Our recommendations are presented in the order they appeared in the preceding sections, and we do not prioritize them.

1. Clarify the causes of low pelagic biomass.
   The relative contributions of clam grazing, light limitation and ammonium inhibition in limiting the accumulation of phytoplankton biomass should be resolved through a dedicated effort, including experimental and modeling components. Monitoring and research associated with expected decreases in ammonium inputs to the Sacramento River should be a key part of this program to inform the integration effort and to determine if any benefits are sustained over time.

2. Monitor and investigate harmful algal blooms (HABs). Despite the high frequency of HABs, we still lack a systematic monitoring program to document HAB distribution, abundance, and toxicity. Additional research on the effects of HABs on organisms of interest under field conditions would be useful to define the level of concern for aquatic resources and identifying management actions that might be effective. A monitoring program for HABs should be designed and implemented as soon as possible.

3. Modernize the monitoring program. Maintaining current long-term monitoring is essential. There is, however, a growing disconnect between what the existing monitoring programs provide and what is needed for effective management of the changing system. Most important, current monitoring focuses almost entirely on abundance and distribution of species, or indices of species groups such as chlorophyll–a concentrations. There is very little monitoring of process rates or fluxes, which are necessary to understand how drivers influence species of concern. Major habitats (e.g., tidal wetlands and SAV) and species groups (e.g., HABs, SAV, FAV, jellyfish, microzooplankton, and bacteria) are not being systematically monitored. Preparation for short-term studies during unusual or extreme events—such as the wet year of 2011 (Brown et al. 2014), the current drought, or the large phytoplankton bloom in spring 2016—would also be beneficial. Finally, effort is needed on an ongoing basis to incorporate modern methods into long-term monitoring.

4. Conduct tidal wetland restoration to benefit native species of concern as an experimental program. This is likely the most urgent issue because large investments are being made in planning and implementation without parallel efforts to fully understand outcomes. Too many uncertainties remain for confident predictions of the outcomes of restoration, especially at the scale of the Delta. Therefore, large-scale restoration should be implemented as adaptive management experiments, which will require considerable additional effort and funding beyond those required to simply construct wetlands. Adoption of an experimental approach integrated across the whole estuary or Delta but addressing all spatial scales discussed above, as well as different
wetland characteristics and configurations, would help to achieve maximum benefit from the time, money, energy and resources being invested. Although conceptual models are being developed and monitoring protocols designed and tested (IEP–TWMPWT 2016), a framework is needed to predict outcomes, synthesize results, or standardize monitoring.

5. **Study the functions of submerged and floating aquatic vegetation (SAV and FAV).** The importance of SAV to consumers of fish in Suisun Marsh and the north Delta tidal channels suggests that Delta–Suisun food webs are more thoroughly linked across the landscape than previously thought. Further, we lack established concepts about the role of FAV. A unified study should be designed to better understand the relative importance of SAV and FAV, as well as other habitat-specific food webs. This recommendation interacts with recommendation 4.

6. **Study the contributions of floodplains.** Determine the extent and fate of floodplain contributions to Delta food webs. This study should focus on Yolo Bypass, which provides a large area.

7. **Model regional processes.** It is unclear if individual, uncoordinated restoration projects can contribute to improvement of pelagic habitats in a specific region or across the Delta–Suisun region as a whole. Such processes should be explored using suitable simulation models to determine if significant long-term benefits seem likely.

8. **Expand efforts at interdisciplinary analysis and synthesis.** Ongoing analysis and synthesis of new knowledge should be an ongoing part of any sustained scientific effort, including the efforts outlined above. Such analysis and synthesis can lead to new understanding, and identify important data gaps (e.g., Brown et al. 2014; Herbold et al. 2014; IEP–MAST 2015). These analyses should increasingly apply modern modeling and statistical tools to improve the reliability and utility of results (e.g., Feyrer et al. 2007; Thomson et al. 2010).

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Delta Smelt: Life History and Decline of a Once Abundant Species in the San Francisco Estuary

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HAiku

Hey there, delta smelt
If you hang on for a while
You will fool us all.

ABSTRACT

This paper reviews what has been learned about Delta Smelt and its status since the publication of The State of Bay-Delta Science, 2008 (Healey et al. 2008). The Delta Smelt is endemic to the upper San Francisco Estuary. Much of its historic habitat is no longer available and remaining habitat is increasingly unable to sustain the population. As a listed species living in the central node of California’s water supply system, Delta Smelt has been the focus of a large research effort to understand causes of decline and identify ways to recover the species. Since 2008, a remarkable record of innovative research on Delta Smelt has been achieved, which is summarized here. Unfortunately, research has not prevented the smelt’s continued decline, which is the result of multiple, interacting factors. A major driver of decline is change to the Delta ecosystem from water exports, resulting in reduced outflows and high levels of entrainment in the large pumps of the South Delta. Invasions of alien species, encouraged by environmental change, have also played a contributing role in the decline. Severe drought effects have pushed Delta Smelt to record low levels in 2014–2015. The rapid decline of the species and failure of recovery efforts demonstrate an inability to manage the Delta for the “co-equal goals” of maintaining a healthy ecosystem and providing a reliable water supply for Californians. Diverse and substantial management actions are needed to preserve Delta Smelt.

KEY WORDS

Hypomesus transpacificus, Sacramento-San Joaquin Delta, endangered species, extinction, co-equal goals, pelagic organism decline (POD)

INTRODUCTION

The Delta Smelt (Hypomesus transpacificus) is a small, translucent fish endemic to the upper San Francisco Estuary (estuary). Until the 1980s, it was an abundant fish in the upper estuary, moving with tides and river flows between the freshwater Delta and brackish Suisun Bay (Moyle 2002). The rapid decline of its population led to its listing as threatened under
state and federal Endangered Species Acts in 1993 (Appendix A, Table A–1). Listing was controversial because the principal home of Delta Smelt is the center of California’s water supply system. The need for information on Delta Smelt has resulted in over 300 peer-reviewed publications since it was proposed for listing in 1989 (Figure 1), as well as countless reports, technical memos, and blogs on its biology and management.

Knowledge about Delta Smelt has been synthesized by Moyle et al. (1992), Bennett and Moyle (1996), Moyle (2002), Bennett (2005), and the IEP Management, Analysis, and Synthesis Team (IEP MAST 2015). In The State of Bay-Delta Science, 2008 (Healey et al. 2008), the Delta Smelt was treated mostly as part of the decline of pelagic fishes in the estuary, with the causes of Delta Smelt decline determined to be uncertain. Since that report, major advances in understanding Delta Smelt biology have occurred (Table 1). Here, we present a synthesis of recent studies, as the smelt dives towards extinction. We cover the following topics: (1) taxonomy and genetics; (2) historic and current distribution; (3) ecology; (4) population trends and dynamics; (5) conceptual models; and (6) causes of decline. We discuss why the present environment of the Delta no longer supports Delta Smelt and what conservation actions can be implemented. Survival of Delta Smelt

Table 1 Key scientific findings on Delta Smelt (DS) since Healey et al. (2008). Details of these findings are presented in the chapter text. Findings are not listed in order of importance.

1. There is a single population of DS, with recently reduced genetic diversity (Fisch et al. 2009, 2011).
2. Spring Kodiak Trawl surveys show DS population approaching extinction, reflecting trends in other sampling surveys (unpublished data, CDFW 2015).
3. Some DS remain year-round in fresh water, primarily in the north Delta (Merz et al. 2011; Sommer et al. 2011; Sommer and Mejia 2013). This was known previously but generally not recognized (e.g., Erkkila et al. 1951).
4. DS are sensitive to warm temperatures, with thermal stress beginning at about 4–5°C below the critical thermal maxima of 24–28°C, depending on life history stage and acclimation temperatures (Komoroske et al. 2015).
5. DS are strongly associated with turbid water in spring and summer, continuing into fall (Feyrer et al. 2007; Nobriga et al. 2008; Sommer and Mejia 2013). Increased turbidity in the Delta (around >10 NTU) is associated with entrainment of adult DS in the State Water Project and Central Valley Project pumps in the south Delta (Grimaldo et al. 2009).
6. DS movements track turbidity and salinity gradients (Feyrer et al. 2007; Bennett and Burau 2015). Increased turbidity in fall and early winter triggers adult movement toward spawning areas (Grimaldo et al. 2009; Sommer et al. 2011).
7. DS engage in “tidal surfing” behavior, moving laterally into shallow water to avoid peak ebb tide flows (Bennett and Burau 2015).
8. DS gonads exhibit multiple stages of oocyte development, indicating that females can spawn more than once, increasing total fecundity (Lindberg et al. 2013).
9. DS experience poor nutritional condition in Suisun Bay during summer months, and there was evidence of contaminant effects in freshwater (Hammock et al. 2015).
10. DS survival from summer to fall is correlated with biomass of calanoid copepods in the low salinity zone (Kimmerer 2008b). As calanoids decline after mid-summer, DS revert to smaller, less nutritious prey items (Slater and Baxter 2014; Kratina and Winder 2015).
11. Thomson et al. (2010) used Bayesian change point analysis to confirm the abrupt decline in delta smelt abundance in the early 1980s and in the early 2000s.
12. The UC Davis Fish Conservation and Culture Laboratory now rears DS through their entire life cycle while maintaining genetic diversity and making large numbers of fish available for laboratory studies (Lindberg et al. 2013).
depends on new, flexible approaches to management (Luoma et al. 2015) as does survival of endangered species worldwide (Helfman 2007).

**TAXONOMY AND GENETICS**

The Delta Smelt is a distinctive estuarine-dependent species (McAllister 1963; Trenham et al. 1998), whose closest relative is the Surf Smelt (H. pretiosus), a marine species that occurs in San Francisco Bay (Stanley et al. 1995). Delta Smelt comprise one interbreeding population. Although genetic diversity of the population is fairly high, there are signs of bottlenecks associated with reduced population size (Fisch et al. 2011). The related Japanese osmerid, Wakasagi (H. nipponensis) has invaded the estuary from upstream reservoirs, but very limited hybridization with Delta Smelt has been detected (Trenham et al. 1998).

**HISTORIC AND CURRENT DISTRIBUTION**

During the first systematic surveys of fish in the upper estuary, Delta Smelt were widely distributed throughout the Delta, Suisun Bay and Marsh, and western San Pablo Bay (Erkkila et al. 1950; Ganssle 1966; Radtke 1966; Moyle 2002). Despite being tolerant of meso-haline salinities (see “Ecology,” page 4), their distribution was largely confined to low salinity (<7 psu) tidal regions (Bennett 2005). The early surveys showed high abundance in Suisun Bay and Marsh, with the highest catches occurring in the Sacramento River channel near Sherman and Decker islands (Bennett 2005). Recent findings that Delta Smelt can reside in fresh water for their entire life cycle (Merz et al. 2011; Sommer et al. 2011; Sommer and Mejia 2013) indicate that their upstream limits are determined by tidal action to transport them to favorable habitats (cool, zooplankton-rich environments).

An analysis of data from seven widespread, long-term sampling programs and 23 regional and short-term (since 2000) sampling programs described the total distribution as extending from San Pablo Bay to the confluence of the Sacramento and Feather rivers in the north Delta and the fork of the Old and San Joaquin rivers in the south Delta, an area encompassing approximately 51,800 ha. Smelt of all stages were most abundant in the center of their range, from Suisun Marsh and Grizzly Bay up the Sacramento River to the Cache-Lindsey Slough Complex (Merz et al. 2011). Because the standard Delta fish surveys (Bennett 2005) sample mainly larger channels and embayments, the importance of peripheral areas as year-around habitat for smelt has been discovered only recently. These include the Napa River, the Cache–Lindsay Slough Complex, the Toe Drain, the Sacramento Deepwater Ship Channel and Liberty Island (Sommer and Mejia 2013). This discovery should probably be labeled as “rediscovery” because the first extensive survey of Delta fishes (Erkkila et al. 1951), caught smelt year-round during 2 years of sampling the fresh waters of the Delta. A similar pattern was shown in Radtke (1966).

The distribution of Delta Smelt changes on a seasonal basis with life stage (Figure 2) (Sommer et al. 2011). In winter, sub-adult and adult smelt move into fresh water for spawning and are sampled using the Fall Midwater Trawl (FMWT) and Spring Kodiak Trawl (SKT). In spring and summer, larval and juvenile smelt move into brackish water, primarily in Suisun Bay and Suisun Marsh (Dege and Brown 2004), and are sampled using the 20-mm Survey. Most Delta Smelt rear in low salinity habitat in summer and fall (Feyrer et al. 2007; Nobriga et al. 2008), allowing them to feed on abundant zooplankton; they are sampled by the Summer Townet Survey (TNS) and the University of California at Davis’ (UC Davis) Suisun Marsh Survey. This seasonal pattern of distribution is used as evidence that Delta Smelt are a migratory “semi-anadromous” species, following the original conceptual model of Moyle et al. (1992). Recent distributional studies, however, indicate that movement patterns of smelt are highly variable, depending on outflow, exports, channel configurations, and other factors.

As Delta Smelt abundance has declined and habitat conditions have changed, their distribution has become more restricted, excluding most of the central and south Delta (Merz et al. 2011). In both old and recent surveys, most smelt have been caught in the arc of habitat from the Cache–Lindsay Slough Complex in the north Delta, down the Sacramento River, to Montezuma Slough in Suisun Marsh. This arc of tidal habitat is connected by flows from the Sacramento River. An increasingly higher
Figure 2  Delta Smelt distribution and abundance, in relationship to four key surveys, calendar year, and the Delta Smelt life cycle. Larvae and post-larvae are sampled in the 20-mm Survey (shown in yellow); juveniles by the Summer Townet Survey (shown in orange); sub-adults by the Fall Midwater Survey (blue); and adult spawners and post-spawners by the Spring Kodiak Trawl (green). A small number of smelt survive a second year, allowing them another opportunity to spawn the following winter.

percentage of smelt caught in various surveys are found in freshwater areas, year around, such as the Sacramento Deepwater Ship Channel and the Toe Drain of the Yolo Bypass (Merz et al. 2011; Sommer et al. 2011; Sommer and Mejia 2013).

Currently, Delta Smelt rarely occur in the central and south Delta, especially during summer/fall because the water is too warm or too clear to sustain them. However, hydraulic conditions created by water operations can cause net flows to pull larvae and juveniles towards the south Delta, although survival is likely low (Kimmerer and Nobriga 2008; Grimaldo et al. 2009). Restrictions on pumping have limited occurrence of these events since 2009 (USFWS 2008; SLDMWA et al. vs. Salazar et al. 2009).

**ECOLOGY**

**Temperature**

Delta Smelt are commonly found at temperatures of 10 to 22 °C. Wild-caught Delta Smelt show a critical thermal maximum of 25 °C for juvenile fish acclimated to 17 °C (Swanson and Cech 2000). Cultured smelt acclimated to 16 °C have a critical thermal maximum of about 28 °C, but thermal tolerance decreases from late-larval to post-spawning fish (Komoroske et al. 2014, 2015). Molecular assays suggest that thermal stress begins at about 20 to 21 °C depending on life history stage. Delta Smelt are unable to compensate for thermal stress, meaning
short-term exposure to stressful conditions can lead to chronic stress (Komoroske et al. 2015).

These results are consistent with reduced smelt catch at higher temperatures. The majority of Delta Smelt juveniles (TNS) and pre-adults (FMWT) are caught at temperatures of $< 22 \, ^\circ C$ (Nobriga et al. 2008). This is consistent with the absence of Delta Smelt from the San Joaquin River and the south-central Delta during summer. Presumably, Delta Smelt of different life stages avoid areas (or die) where water temperatures are near thermal maxima, and are therefore less likely to be captured in surveys.

**Salinity**

Delta Smelt is a euryhaline species mostly inhabiting salinities from 0 to 7 psu, but can tolerate up to 19 psu (Swanson and Cech 2000; Moyle 2002) and even sea water for short periods of time (Komoroske et al. 2014). Data from the TNS and FMWT indicate that over 70% of juvenile and 60% of pre-adult Delta Smelt are caught at salinities less than 2 psu, with over 90% occurring at less than 7 psu (Bennett 2005). Recent studies indicate that there is a small part of the population that stays in fresh water for its entire life cycle. The fact that Delta Smelt can be reared in captivity in fresh water through their entire life cycle supports these findings. However, most smelt spend part of their life cycle near or slightly upstream of 2 psu in the entrapment or low-salinity zone (LSZ). Both the TNS (Nobriga et al. 2008) and FMWT (Feyrer et al. 2007) found peak occurrences of Delta Smelt in areas with low specific conductance, with somewhat lower occurrences in fully fresh water. This finding is consistent with the observation that most juveniles and sub-adults rear in the low-salinity region of the estuary, as they presumably did historically (Feyrer et al. 2007; Nobriga et al. 2008; Hasenbein et al. 2013; Komoroske et al. 2014).

**Turbidity**

Juvenile and sub-adult smelt are strongly associated with turbid water in spring and summer (Nobriga et al. 2008; Sommer and Mejia 2013), continuing into fall (Feyrer et al. 2007). The translucent body color and small size of Delta Smelt may make them less visible to predators in moderately turbid water. Turbidity reduces Largemouth Bass predation on Delta Smelt in mesocosm experiments (Ferrari et al. 2014).

Delta turbidity above 10 NTU coincides with increased entrainment of adult Delta Smelt (Grimaldo et al. 2009). This turbidity increases results from the flush of suspended material in rivers, after major storms. The “first flush” of the rainy season is likely a trigger for adult Delta Smelt to move toward spawning areas, including areas within the influence of the south Delta pumping plants (Sommer et al. 2011; Bennett and Burau 2015).

**Feeding Behavior**

Delta Smelt are visual zooplankton feeders, using suspended particles (i.e., turbidity) as a background to increase visual acuity in the near-field during daylight (Hobbs et al. 2006; Slater and Baxter 2014). As with all visual feeders, visual range and prey density determine feeding success. Optical attributes of the water column are affected by turbidity from organic particles, such as algae and detritus, and from inorganic particles, such as sand and silt (Utne–Palm 2002; Hecht and Van der Lingen 2012). Feeding of larval Delta Smelt is increased at high algae concentrations and light levels (Baskerville-Bridges et al. 2003). The addition of algae or other suspended particles is standard practice for successfully rearing Delta Smelt larvae in culture facilities (Mager et al. 2003; Baskerville–Bridges et al. 2005; Lindberg et al. 2013). Presumably, the suspended particles provide a background of stationary particles that helps the larvae detect moving prey. Hasenbein et al. (2016) observed highest feeding rates of late-larval Delta Smelt at beween 25 and 80 NTU. Feeding success of juvenile and adult Delta Smelt is reduced by high turbidity (250 NTU) when light levels are very low (Hasenbein et al. 2013), supporting observations that smelt feed largely in the daytime (Hobbs et al. 2006). However, such high turbidities are rarely observed in the wild.

**Swimming Behavior**

Laboratory smelt have maximum sustained swimming velocities of about 28 cm s$^{-1}$ (Swanson et al. 1998). A discontinuous “stroke-and-glide” behavior is used at water velocities of less than 10 cm s$^{-1}$, while sustained swimming occurs above
15 cm s\(^{-1}\). However, many fish will not swim above water velocities of 10-15 cm sec\(^{-1}\). Stroke and glide swimming may be advantageous for a diel-feeding planktivore because it minimizes continuous movement that might attract predators.

Despite this swimming performance, Delta Smelt can travel large distances by using tidal currents (Bennett and Burau 2015). Lateral turbidity gradients change with tides and around first flush events, and these gradients coincide with lateral Delta Smelt movements toward the mid-channel during flood tides and toward the shoreline during ebb tides. Delta Smelt are caught more frequently throughout the water column during flood tides. On ebb tides they are observed only in the lower half of the water column and along the sides of the channel. By behaviorally selecting positions on the edge or center of the channel and near the surface or bottom of the water column, Delta Smelt can use tidal currents to move upstream or downstream, or avoid such currents to maintain position (Bennett et al. 2002; Feyrer et al. 2013; Bennett and Burau 2015).

**Food and Feeding**

Delta Smelt feed mainly on small crustacean zooplankton, particularly calanoid copepods (Moyle et al. 1992; Lott 1998; Slater and Baxter 2014). This is true across decades of study. For example a study from 1972–1974 showed that the dominant food item was the calanoid copepod *Eurytemora affinis*, with cladocerans and mysid shrimp (*Neomysis mercedis*) being important at times (Moyle et al. 1992). By the late 1980s, *E. affinis* was largely replaced in smelt diets, except in early spring, by the similarly sized introduced calanoid *Pseudodiaptomus forbesi*. Smelt also eat other calanoids, including the larger *Acartiella sinensis* and the more evasive *Sinocalanus doerri*, but they are less commonly found in diets. In fresh water, a higher proportion of cladocerans and native cyclopoid copepods appear in diet studies (Nobriga 2002; Hobbs et al. 2006; Slater and Baxter 2014). In general, most copepod prey of Delta Smelt are of similar nutritive value (Kratina and Winder 2015). Larger smelt are capable of supplementing their diet with larger crustaceans such as mysids and amphipods, as well as with larval fishes (Moyle et al. 1992; Lott 1998; Feyrer et al. 2003). First food tends to be copepod nauplii or copepodites. The tiny invasive cyclopoid *Limnoithona tetraspina* is poorly represented in diets; this is presumably a function of capture evasion and low nutritional value (Bouley and Kimmerer 2006; Kratina and Winder 2015).

**Predators and Competitors**

Historically, Delta Smelt were likely occasional prey for aquatic predators such as Thicktail Chub (*Gila crassicauda*), Sacramento Perch (*Archoplites interruptus*), Sacramento Pikeminnow (*Ptychocheilus grandis*), Chinook Salmon and Steelhead smolts, Sturgeon, and perhaps avian predators as well (Grossman, this volume). After the Gold Rush, native predatory fish were largely replaced by a suite of non-native species, most conspicuously Striped Bass (*Morone saxatilis*), a pelagic piscivore (Stevens 1966; Thomas 1967; Moyle 2002; Grossman, this volume).

Predation rates on smelt were likely linked to their presumed abundance, given that their behavior and translucent color makes them difficult to target as a prey species. Currently, Delta Smelt are rarely seen in diets of fish predators. This is probably because low encounter rates make it difficult for active, mobile predators to detect it, as an uncommon, nearly invisible prey (Grossman et al. 2013; Grossman, this volume). The alien fish that may be the most significant predator on Delta Smelt is Mississippi Silverside (*Menidia audens*). Silversides feed diurnally along shallow water edge habitat and could be potentially important predators on smelt eggs and larvae (Bennett and Moyle 1996; Bennett 2005). Delta Smelt DNA has been isolated in silverside guts (Baerwald et al. 2012), suggesting that silversides do prey on smelt. However, no quantitative evidence exists for the overall effect of predation or competition by silversides.

The potential for alien piscivores to affect Delta Smelt abundance is suggested by (1) by inverse correlations of predatory fish (including Striped Bass, Largemouth Bass [*Micropterus salmoides*] and Mississippi Silverside) with Delta Smelt (Bennett and Moyle 1996; Brown and Michniuk 2007), and (2) by bioenergetics models of Striped Bass that show they can potentially be significant predators on smelt (Loboschefsky et al. 2012). However, empirical evidence and statistical modeling have shown scant
evidence for a cause-and-effect relationship between smelt and predator abundances (Mac Nally et al. 2009; Thomson et al. 2010; Maunder and Deriso 2011; Miller et al. 2012; O’Rear 2012; Nobriga et al. 2013), a pattern that follows much of the general literature on predation (e.g., Doherty and Richie 2016).

Historic competitors may have been other planktivorous fishes, especially Northern Anchovy (*Engraulis mordax*) and Longfin Smelt in Suisun Bay, and juveniles of other native fishes in the Delta. Potential alien competitors include American Shad (*Alosa sapidissima*), Threadfin Shad (*Dorosoma petenense*), Golden Shiner (*Notemigonus chrysoleucas*), juvenile centrarchids, and juvenile Striped Bass and Mississippi Silverside (an intraguild predator).

Competition may occur if zooplankton resources are limited during critical points of the smelt life cycle. Today, the most effective, if indirect, competitors are overbite (*Potamocorbula amurensis*) and Asian clams (*Corbicula fluminea*) (Kimmerer et al. 1994; Kimmerer 2006), which depress zooplankton abundance by grazing down phytoplankton and zooplankton nauplii (Durand 2015). Although Delta Smelt remained abundant in the Delta and Suisun Bay through the 1970s, long after most introduced fish predators and competitors had established populations (Grossman, this volume), smelt began rapidly declining after the invasions of overbite clam and Mississippi Silverside in the 1980s. At the same time, presumed predators and competitors, such as other planktivorous fishes and Striped Bass, began parallel long-term decreases in abundance.

**REPRODUCTION**

Delta Smelt have a protracted spawning season, given their life span, from late January through June. Larvae are seen from late February through early May (Moyle et al. 1992; Bennett 2005). They are thought to spawn on shallow sandy beaches, although spawning has not been observed in the wild. In laboratory culture, Delta Smelt spawn on the bottom and sides of tanks, indicating they need substrate for deposition of their adhesive eggs (Lindberg et al. 2013). The number of eggs per female is exponential to length for cultured fish, although the relationship is less clear in wild fish (Lindberg et al. 2013). The number of eggs per female for small fish (60 to 80 mm) ranges from 1,000 to 2,500. Larger females (80 to 120 mm) can have 2,500 to 12,000 eggs (Bennett 2005; Lindberg et al. 2013). Mature eggs have been found in females as small as 56-mm fork length (FL) in the wild (Kurobe et al. 2016).

Spawning behavior of captive Delta Smelt has been observed and studied using parentage genetic techniques in outdoor mesocosms. Females can spawn repeatedly with multiple males (LaCava et al. 2015) up to four times in a season, with resting periods of 40 to 50 days (M. Nagel, pers. comm., 2016, with J. Hobbs, unreferenced, see “Notes”). The capacity of females to produce multiple clutches of eggs in the wild when environmental conditions are favorable for reproduction could be important for maintaining population resilience during periods of low adult abundance (Kurobe et al. 2016).

**POPULATION TRENDS AND DYNAMICS**

Abundance indices for Delta Smelt are calculated from the catch-per-unit-effort of key agency surveys, shown in Figure 3 (IEP MAST 2015). Indices of abundance, rather than abundance estimates or absolute numbers, are used for two principal reasons. First, their distribution is patchy and mobile, and smelt may at times occupy regions that are difficult to sample. This is true even in areas of peak abundance, where likelihood of capture increases. Second, capture efficiencies of the various types of sampling gear are poorly known and difficult to compare among surveys. With the exception of the SKT surveys, none of the sampling programs were specifically designed to capture Delta Smelt and they all have biases related to how and where they sample. But together they survey a wide variety of habitats and regions of the estuary, at all times of year, providing a reasonable picture of smelt distribution and abundance (IEP MAST 2015).

Because actual smelt population size cannot be known with certainty, the indices are a convenient way to track population trends and their response to environmental conditions. Delta Smelt is mostly an annual species, resulting in highly variable year to year abundance (Bennett 2005). Small changes in vital rates such as growth, survival and fecundity
Figure 3  Abundance indices for life stages of Delta Smelt including larvae and juveniles (20-mm Survey), juveniles (Summer Townet Survey), sub-adults (Fall Midwater Trawl), and adults (Spring Kodiak Trawl). The initiation of each individual survey is indicated by the first bar; missing bars indicating years for which an index was not calculated. Indices for each survey were standardized by subtracting each yearly index from the global mean for each survey and dividing by the standard deviation. (Sources: Interagency Ecological Program unpublished data, CDFW and CDWR).
can have large effects on adult abundance (Bennett 2005), although long-term trends are apparent. Delta Smelt experienced a major decline in the early 1980s (Manly and Chotkowski 2006; Thomson et al. 2010), followed by a substantial but brief increase in 1998–1999 (Manly and Chotkowski 2006). This, in turn, was followed by an abrupt decline in the early 2000s, part of the so-called pelagic organism decline (POD) (Manly and Chotkowski 2006; Sommer et al. 2007; Thomson et al. 2010).

The population dynamics of Delta Smelt have been examined by calculating ratios of various indices to estimate survival or population growth within or between cohorts (Mauner and Deriso 2011; MacNally et al. 2010; Miller et al. 2012). The role of density dependence has been an area of emphasis for these population dynamic studies. An apparent decrease in the carrying capacity of the estuary in the 1980s resulted in an increase in density-dependent mortality from the juvenile stage in late summer through the adult stage in fall (i.e., between the TNS and FMWT indices) (Sweetnam and Stevens 1993; Bennett 2005; Mauner and Deriso 2011). This is supported by chronically low zooplankton prey density in summer (Miller et al. 2012). However, apparent density-independent survival from late larvae to juveniles (i.e., between the 20-mm Survey and TNS indices, Figure 3), suggests that carrying capacity is not a limitation during spring (Mauner and Deriso 2011). High summer temperatures are associated with reduced juvenile abundance and density-independent mortality from larval to juvenile stages (Mac Nally et al. 2009; Mauner and Deriso 2011).

Yearly recruitment based upon adult population abundance (stock size), has been described as density “vague” because, while there is some evidence of reduced survival with higher density, survival is highly variable from year to year (Mauner and Deriso 2011; Mac Nally et al. 2010; Miller et al. 2012). In general, the relationship between sub-adult (spawner) abundance indices and juvenile abundance in the following year is poor (i.e., between the FMWT and TNS indices, Figure 2), as is the relationship between subadult indices from one year to the next. This suggests that inter-generational abundance is driven more by environmental conditions, rather than by density-dependent factors.

As the Delta Smelt population approaches zero, density dependence becomes increasingly less likely. However, shrinking habitat volume, or declining food abundance in fall, might facilitate periods of density-dependent mortality, particularly between the juvenile to adult stage. The most useful explanatory variables for Delta Smelt stock–recruit relationships appear to be factors associated with juvenile survival, e.g., summer–fall food availability, summer temperatures, and egg production (Rose et al. 2013a, 2013b).

**CONCEPTUAL MODELS**

Conceptual models that link ecosystem functions with proposed management actions are valuable tools to highlight key uncertainties in fisheries management (Thom 2000; Ogden et al. 2005). In the Sacramento–San Joaquin Delta, conceptual models have been used extensively to synthesize knowledge of species–habitat relationships (Baxter et al. 2010; DiGennaro et al. 2012), developing predictions for adaptive management actions (USBR 2012) and evaluating outcomes of drought (L. Conrad, pers. comm. to L. R. Brown, 2016, unreferenced, see “Notes”). Many such efforts have been conducted for Delta Smelt since its listing in 1993, which was accompanied by the first smelt conceptual model (Moyle et al. 1992). Since then, a series of conceptual models followed (Bennett 2005; Nobriga and Herbold 2009; Baxter et al. 2010).

A new ecosystem-based conceptual model framework for linking environmental drivers to stage-specific Delta Smelt responses is based on a literature review of Delta Smelt biology and ecology (IEP MAST 2015). The model format is process-driven rather than descriptive, so it uses a box and arrow diagram. The model is structured around four quadrants that represent each life stage (life cycle module) and embedded within a series of hierarchical tiers. The tiers represent direct and indirect effects from landscape-level attributes, environmental drivers, and habitat attributes which drive vital rate responses (i.e., growth and survival) through the life cycle. For each life stage module, a more traditional box-and-arrow diagram links various habitat attributions to the transition of the smelt through the life stage. The diagrammatic representation is complex and comprehensive, and allows identification of disparate...
Linkages between environmental variables and vital rates of smelt. The framework is also flexible enough to adapt to most management scenarios and is being used as a management tool to assess the effects of the recent drought on Delta Smelt and to guide monitoring plans for tidal wetland restoration (IEP TWMPWT 2016).

Here, we present two diagrams that synthesize our understanding of Delta Smelt biology and ecology based on information in this report (Figures 4 and 5). Our synthesis is presented as a hypothesis, grounded in our combined expertise (collectively more than 100 years of accumulated experience in the system).

**Figure 4** describes physical controls on foraging effectiveness during the growth phase of Delta Smelt (March to December), as a different example of conceptual models that can capture various aspects of smelt behavior and life history. The vertical arrow on the left of the Figure represents a hydrodynamic gradient of mixed to stratified conditions or from low to high residence time. The horizontal arrow shows a depth gradient from deep to shallow. Delta Smelt move opportunistically across the environmental gradients to optimize their physiological needs, represented by the triangle, as a series of trade-offs. Smelt perform best in turbid conditions, but turbidity is likely to be greatest in highly turbulent areas, either from wind or high-velocity currents, where smelt, with limited swimming ability, have a hard time sustaining themselves (Bever et al. 2016). Foraging success can improve in some shallow regions of the estuary, where less turbulent conditions and higher residence time allow zooplankton to aggregate. This became particularly true after the 1986 invasion of the overbite clam, which led to greatly decreased zooplankton abundance. However, the ability of Delta Smelt to find turbid, food-dense regions is mediated by thermal tolerance, especially in summer/fall. Thermal refuges may exist away from abundant food resources (e.g., in deep water). We hypothesize that smelt must actively negotiate conditions to maximize their needs (i.e., turbidity, food abundance, and temperature) and are potentially most viable if they can find conditions where these needs converge.

**Figure 5** shows a life cycle diagram for Delta Smelt, using a format similar to that in Figure 2. Key stressors referenced in the text, both direct and indirect, are shown at vulnerable stages of the life cycle. Chronic low-level stressors are omitted. In December adults begin upstream movement, timed to the first pulse of outflow that is sufficient to increase turbidity (Bennett and Burau 2015). Spawning habitat has been reduced in part because of the effect of restricted inflow and increased export, which have reduced turbidity and promoted the colonization of alien species, in particular dense stands of Brazilian waterweed, *Egeria densa* (Durand 2015). This submersed aquatic vegetation (SAV) has created highly unfavorable habitat by occupying likely spawning areas, and by slowing the water further, making it clearer and warmer. This makes it even less suitable for Delta Smelt and more suitable for alien fishes such as Largemouth Bass and sunfishes.

Delta Smelt have generally been abundant in the north Delta, even before their decline, which is where most current reproduction is thought to occur. While the habitat is less degraded than in the south Delta, predation on eggs and larvae by Mississippi Silversides may lead to high mortality. Most post-larval fish move out of the north Delta in spring, at which point they may be vulnerable to entrainment in the south Delta pumps and, hence, high mortality. These conditions have been reduced when springtime pumping restrictions are put in place during periods of vulnerability.

Apparently, most juvenile smelt once reared in Suisun Bay and Marsh, where historically, they fed upon abundant plankton resources. Since the 1980s, phytoplankton declines resulting from poor water-quality conditions (possibly high levels of ammonium) and intensive grazing by overbite clam, have led to food limitation in much of Suisun Bay, especially in late summer and fall. Foraging success is further limited by decreases in outflow that constrain the LSZ to a deeper and more spatially constricted region, rather than the shallow habitats of Little Honker Bay and Suisun Marsh. Food limitation likely reduces fall juvenile growth, limits survivorship to adulthood, and limits reproductive output.
CAUSES OF DECLINE

“Uncontrolled drivers of change (population growth, changing climate, land subsidence, seismicity) means that the Delta of the future will be very different from the Delta of today.”

—Healey at al. 2008

The ultimate cause of decline in Delta Smelt is competition with people for water and habitat. The explosive growth of the California economy since the Gold Rush resulted in rapid and extensive habitat alteration, invasions of new predators and competitors, and changes in hydrology. Changes continue at an accelerated pace, tracking both population and economic growth (Hanak et al. 2011; Hanak and Lund 2011). In this section we briefly review proximate drivers of decline: entrainment, altered hydrology, food, predation, contaminants, habitat change, drought, and climate change. We finish by integrating the science into a synthetic understanding of Delta Smelt biology.
Figure 5  Simplified conceptual model of key hypothesized factors that limit Delta Smelt recovery. Note that direct factors of food limitation, predation, and spawning habitat loss are linked to underlying causes. Orange arrows represent the calendar year; yellow shapes represent predation by Mississippi Silversides on eggs and larvae; blue shapes represent entrainment of post-larvae and juveniles at the south Delta water export facilities; green shapes represent factors that affect food limitation on sub-adults; brown/grey shapes represent factors that affect turbidity (brown) and spawning habitat (grey). Low turbidity (brown box, middle) increases stress and interacts with other factors to decrease smelt success. For a more detailed model, see IEP MAST (2015).
Entrainment

The Delta has thousands of water diversions, but most entrainment of Delta Smelt is by the giant pumps of the Central Valley Project (CVP) and State Water Project (SWP) located in the south Delta. High exports of freshwater and low inflows to the Delta can create reverse flows and asymmetrically strong flood tides, which carry smelt, especially larvae, toward the pumps (Monsen et al. 2007; Grimaldo et al. 2009). Few Delta Smelt are entrained by small diversions found throughout the Delta (Nobriga and Herbold 2009). The pump intakes are generally small and close to shore, and most diversions take place at times and places when Delta Smelt, especially larval smelt, are not likely to be present.

The intakes to the south Delta pumping plants have louvers that divert fish to a capture facility where they are collected and trucked for release at downstream locations, a process known as salvage. A sub-sample of these fish is counted, but estimates do not include larvae and juvenile fish less than 20 mm total length (TL). For Delta Smelt, these counts provide a rough estimate of >20-mm-long fish killed by the operation because most smelt do not survive being salvaged (Miranda et al. 2010a, 2010b; Aasen 2013; Afentoulis et al. 2013; Morinaka 2013). Because of high pre-screen mortality (especially in Clifton Court Forebay, at the SWP) and a lack of estimates for fish <20 mm long, only a small percentage of all smelt entrained are counted (Castillo et al. 2012), and mortality estimates for handling and transport are biased low. Moreover, most smelt moved into the central and south Delta do not make it to the pumps; they likely die because of poor water quality or other factors. The population-level effect of removing spawning adults is likely high. Salvage mortality tends to be highest at times when the Old–Middle River flow is most negative (i.e., flows are reversed) and turbidity is high (USFWS 2008). Salvage also tends to be highest at times when exports are high relative to outflow, so a greater proportion of the water is moving towards the pumps; this changes the pattern of water movement through the central and south Delta (Kimmerer 2008).

Delta Smelt are most vulnerable to entrainment by the CVP and SWP pumps during upstream adult spawning movements and as larvae move downstream from fresh to brackish water (Sweetnam 1999; Sommer et al. 2011). In the early 1980s, when smelt were still abundant, high salvage occurred at all export levels, dominated by adults between December and March–April, and by larvae and juveniles from April through July (Kimmerer 2008; Grimaldo et al. 2009). Since the 1990s, May–June juvenile salvage has declined and July–August late juvenile and sub-adult salvage has nearly disappeared, because Delta Smelt no longer reside over summer in the central–south Delta.

During years of high exports, up to 25% of larval–juvenile smelt and up to 50% of the adult population may be entrained at the CVP and SWP, annually (Kimmerer 2008). Salvage increased greatly in winter of 2002, coincident with the first year-class of the POD (Figure 6). Modeling efforts suggest that these periodic entrainment losses may have adversely affected the Delta Smelt population (Kimmerer 2011; Maunder and Deriso 2011; Miller et al. 2012; Rose et al. 2013a, 2013b). In particular, the high entrainment of Delta Smelt in the winter of 1982, followed by high rates of pumping in the following spring, when larvae were most abundant, is associated with the beginning of the major smelt decline over the next 3 decades. Whether or not this is cause-and-effect needs further study. The drastic reduction in the population during the 1980s made it more difficult for it to recover from other events such as overbite clam and silverside invasions. Given the annual life cycle, any episodic salvage event may undermine population resilience by keeping numbers low, even when environmental conditions are good.

Food and Feeding

Food resources for Delta Smelt, particularly calanoid copepods and mysid shrimp, have decreased since the 1980s, corresponding to declines in phytoplankton abundance (Brown et al., submitted). POD species abundances are related to prey abundance, and decreases in prey have reduced the carrying capacity of the system to support fish (Sommer et al. 2007; Kimmerer 2012). Modeling exercises support the hypothesis that food limitation affects Delta Smelt population trends (Miller et al. 2012; Rose et al. 2013b).
Studies of smelt gut fullness, growth, condition, and histology provide additional evidence for food limitation, particularly in spring and fall (Feyrer et al. 2003; Bennett 2005; Bennett et al. 2008; Baxter et al. 2008; Hammock et al. 2015). A mismatch between smelt and their prey in spring may decrease juvenile recruitment (Bennett 2005). Both Lott (1998) and Slater and Baxter (2014) found that >30% of Delta Smelt larvae <14 mm FL had empty guts in April. The frequency of empty guts increased during late spring–early summer during metamorphosis of larvae to juveniles (fish ca. 20 to 24 mm FL). Low calanoid abundance in late summer may affect survival to fall abundance (Kimmerer 2008; Mac Nally et al. 2009; Thomson et al. 2010; Miller et al. 2012). Smelt diets in Suisun Bay revert to smaller prey items after mid-summer and into fall, including less nutritious nauplii (Kratina and Winder 2015) and the smaller Limnoithona tetraspina (Slater and Baxter 2014). Warm water temperatures during summer exacerbate stress from low food availability and may explain reduced survival from summer to fall in some years (Bennett 2005; Bennett et al. 2008). Hammock et al. (2015) found that Delta Smelt captured from Suisun Marsh, the north Delta, and the Sacramento Deep Water Ship Channel showed more stomach fullness and better condition and growth indices than did fish captured in Suisun Bay and the confluence. Food limitation effects on Delta Smelt growth and survival varied considerably with season, year, and location (Hammock et al. 2015).

**Predation**

Delta Smelt have adaptations that make them surprisingly unavailable as prey for other fishes, except as larvae. Both native and alien potential Delta Smelt predators are generalists that focus on abundant prey, rather than on species as rare as Delta Smelt today (Grossman et al. 2013, Grossman, this volume). There is no evidence that these predators had a major effect on Delta Smelt populations in the past (see earlier discussion). Presently, Mississippi silverside is probably the most important predator of Delta Smelt larvae because of their ability to prey on eggs and larvae and their high abundance in shallow environments.
areas where Delta Smelt spawn (Bennett and Moyle 1996; Bennett 2005; Baerwald et al. 2012).

In the south Delta, warm temperatures, high water clarity, low flows, and expansion of invasive aquatic vegetation have created a novel ecosystem that largely excludes Delta Smelt and favors alien fishes. The alien fishes feed on a variety of alien and native prey, including invertebrates such as crayfish and amphipods. Largemouth Bass will consume Delta Smelt in mesocosms (Ferrari et al. 2014), but are unlikely to be a major predator in the wild because of limited habitat overlap between the two species.

Contaminants

Delta Smelt are exposed to a variety of contaminants throughout their life cycle but the nature and degree of the effects of contaminants on Delta Smelt health are not well documented (Johnson et al. 2010; Brooks et al. 2012). If contaminants significantly affect smelt, the effects are likely chronic rather than acute (Werner et al. 2010) but overall effects on wild populations are not known. Because of their short life cycle, smelt are more likely to suffer contaminant effects from direct exposure than from cumulative effects (biomagnification). Contaminants are most likely to affect smelt in combination with other stressors, such as starvation. The categories of contaminants that may affect Delta Smelt are pesticides, ammonia and ammonium, heavy metals, polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs), contaminants of emerging concern, and mixtures of any of the above (Fong et al., submitted). An increasingly wide array of contaminants are present in the water in which smelt live and in the prey they eat. New, highly toxic compounds (e.g., pyrethroids) have appeared coincident with the decline of Delta Smelt. Their contribution to the decline, if any, is most likely through indirect effects.

Pesticides. Pesticide concentrations in surface waters of the Delta are typically highest during winter and spring in runoff from rainfall. Thus pesticides are most likely to affect freshwater life stages of smelt. Peak densities of larval and juvenile Delta Smelt can coincide with elevated concentrations of dissolved pesticides (Kuivila and Moon 2004). Pesticides can affect Delta Smelt in diverse ways, by altering swimming behavior, gene expression, immune response, detoxification, and growth and development (Connon et al. 2009; Jeffries et al. 2015).

Ammonia and Ammonium. Delta Smelt spawning and larval nursery areas in the northern Delta may be at risk from exposure to ammonia and ammonium, mainly from discharge by the Sacramento Regional Wastewater Treatment Plant into the lower Sacramento River (Connon et al. 2011). However, concentrations of ammonia used in laboratory studies to assess risk were higher than concentrations measured in the wild.

Heavy Metals. Heavy metals and other elements of concern in the Delta include copper, mercury, and selenium. Sublethal effects of these elements on fishes include reduced fertility and growth, impaired neurological and endocrine functions, and skeletal deformities that affect swimming performance (Boening 2000; Chapman et al. 2010). These elements are often associated with sediment and may affect adult and larval life stages of smelt, because sediment is transported with significant rain events, especially the “first flush.”

PAHs and PCBs. PAHs and PCBs from urban and industrial sources are found in excess of established water quality objectives in the Delta (Thompson et al. 2000; Oros et al. 2007), and are known to cause endocrine disruption in fish (Nicolas 1999; Brar et al. 2010).

Contaminants of Emerging Concern. Contaminants of emerging concern, such as pharmaceuticals, hormones, personal care products, and industrial chemicals are widespread in the aquatic environment, biologically active, and are largely unregulated (Kolpin et al. 2002; Pal et al. 2010). They are known to cause sublethal effects in fish including endocrine disruption, changes in gene transcription and protein expression, and morphological and behavioral changes (Brander et al. 2013).

Contaminant Mixtures. Interactions among contaminants can have both synergistic and antagonistic effects on fish physiology (e.g., Jordan et al. 2011). There is increasing evidence that compounds in mixtures show adverse effects at concentrations at which no effects are observed for
single toxicants (e.g., Silva et al. 2002; Walter et al. 2002; Baas et al. 2009).

**Habitat Change**

Delta Smelt are pelagic fish that primarily inhabit the Delta and Suisun Bay. Here, we discuss some of the more important factors that influence their pelagic habitat: hydrology, salinity and outflow, turbidity, harmful algae blooms, drought, and climate change.

**Hydrology.** Since the construction of Oroville Dam in the 1960s, upstream diversions of water and exports from the Delta have increased in most years, while inflow (and consequently outflow) has decreased (Lund et al. 2007). These dramatic changes in hydrology and related factors have made much of the south and central Delta unsuitable as habitat for smelt and have interacted with other factors to create unfavorable conditions for smelt survival (Moyle and Bennett 2008). Changes in the timing and magnitude of seasonal Delta outflows have changed the size and location of places where smelt can find adequate food resources, especially in the fall. Changes to hydrology have likely promoted alien invasions including the spread of Brazilian waterweed, Mississippi Silverside and Largemouth Bass. These combined changes have caused Delta Smelt largely to disappear from the central and southern Delta, signifying major habitat loss (Feyrer et al. 2007; Nobriga et al. 2008). Most adult Delta Smelt now either move into the Sacramento River and Cache Slough region for spawning and rearing or stay year around in fresh water.

**Salinity.** While Delta Smelt have a fairly broad salinity tolerance; they were historically most abundant in the low salinity zone (LSZ) of the estuary, the position of which is determined by outflow (Moyle et al. 1992; Kimmerer et al. 2013; Sommer and Mejia 2013). Moderate hydrological conditions in late winter and spring place the LSZ in the Grizzly Bay region of Suisun Bay (Jassby et al. 1995). These conditions were beneficial to the Delta Smelt population at least partly because of high food abundance. At present, there is little evidence of the benefit of summer and fall occupancy in the LSZ (IEP MAST 2015). The relationship of smelt life stage abundance indices to spring (February to June) X2 (location of the 2-ppt isohaline) shifted downward after each sharp “step” decline in overall abundance. However, the slope of the relationship before and during the POD has not changed significantly (IEP MAST 2015), suggesting that years that carry the LSZ in western Suisun Bay through spring (e.g., 2011) have a positive effect on Delta Smelt abundance. In addition to moving the LSZ to a favorable location for smelt, increased outflow influences habitat quality through its effect on food supply, dilution of contaminants, and turbidity.

**Turbidity.** Long-term declines in turbidity may be a key reason that juvenile Delta Smelt now rarely occur in the south Delta during summer (Nobriga et al. 2008). Turbidity is usually lower in the central and south Delta than in the Suisun Bay and north Delta regions (Nobriga et al. 2008; Durand 2014). This may result, in part, from changes in flow patterns, river inputs, and sediment trapping by SAV (Hestir 2010). Occurrence of adult Delta Smelt at the SWP salvage facilities is linked with high Delta turbidity associated with winter “first flush” events. Relatively high turbidity (mean of 27 NTU in 2009 and 2010) in the Cache Slough region (from tidal asymmetry, a limited tidal excursion, and wind-wave re-suspension) may help to make this region a year-round refuge for Delta Smelt (Morgan–King and Schoellhamer 2013). Overall, turbidity is recognized increasingly as an important influence on smelt distribution and, perhaps, abundance. The increasing clarity of Delta water in recent years may, therefore, have played a role in its decline, or at least limited the amount of suitable habitat.

**Microcystins.** Periodic blooms of toxic blue-green cyanobacteria, *Microcystis aeruginosa*, most commonly occur in August and September. They are an emerging concern for Delta Smelt (Lehman et al. 2005, 2013) because *M. aeruginosa* can produce toxic microcystins. Blooms typically begin on the San Joaquin River side of the Delta, away from the core summer distribution of Delta Smelt. However, some overlap is apparent during and after blooms and as cells and toxins are dispersed downstream after blooms (Baxter et al. 2010). *M. aeruginosa* distribution has expanded north during the drought (Morris 2013). Studies by Lehman et al. (2010) found microcystins in the tissues and food of pelagic fishes, including Striped
Bass and Threadfin Shad, suggesting that Delta Smelt could be exposed to microcystins, depending on the degree of co-occurrence of Delta Smelt with blooms. Microcystins are toxic to other fish of the region (Acuña et al. 2011, 2012). Laboratory studies have shown that *M. aeruginosa* also can have a negative effect on calanoid copepods, an important food source for Delta Smelt, although it is unclear how laboratory results translate to field conditions (Ger et al. 2009, 2010). Factors that are thought to cause more intensive *M. aeruginosa* blooms include warmer temperatures, longer water residence times, high nitrogen levels, and clear water (Lehman et al. 2005, 2013; Baxter et al. 2010; Morris 2013). These conditions, which are generally unsuitable for Delta Smelt, occur in the estuary during dry years (Lehman et al. 2013). The intensity and duration of *M. aeruginosa* blooms are expected to increase over the long-term, along with any negative effect on Delta Smelt, from the increased frequency of drought conditions associated with climate change (Lehman et al. 2013). In short, *M. aeruginosa* blooms have not been implicated in Delta Smelt decline but they may be influential in the future as an added stressor during generally unfavorable conditions.

**Water Temperature.** Unfavorable temperatures are increasingly characteristic of much of the Delta in summer, and are associated with the absence of Delta Smelt from the central and south Delta during summer. Delta Smelt do occur in freshwater habitats of the north Delta during summer months. This region is typically cooler than the central and south Delta as a result of cooler flows from the Sacramento River. Years with warm water conditions result in increased energetic demand and, given persistent food limitation, small increases in temperatures could have large effects on Delta Smelt. For example, several modeling and empirical studies have suggested the summer to fall transition period may be critical for Delta Smelt survival (Maunder and Deriso 2011; Mac Nally et al. 2010; Miller et al. 2012; Rose et al. 2013a, 2013b). This coincides with the warmest time of the year in both freshwater and low-salinity habitats. Because Delta Smelt are sensitive to warm temperatures (Komoroske et al. 2014), they may experience chronic stress during summer months. Climate change projections suggest that all regions of the Delta that currently maintain summer water temperatures suitable for smelt will be unsuitable in 50 or so years, depending on the models used (Brown et al. 2013).

**Drought.** Drought is a factor that influences smelt distribution and abundance because of its effects on water quality and smelt habitat. While long-term drought is part of the evolutionary history of smelt, modern droughts exacerbate human-caused changes to the estuary, creating conditions that are much worse than would have occurred historically. Under current conditions, not only does the water become warmer and clearer in response to drought, but there is likely less dilution of contaminants and increased likelihood of harmful algae blooms. The suppression of Delta Smelt populations in 2007–2009 and since 2012 is presumably at least partly an artifact of drought. The drought of the 1980s enabled the rapid invasion of the overbite clam, expansion of Mississippi Silverside populations, and the spread of Brazilian waterweed, which reduced the ability of much of the Delta to support Delta Smelt.

**Climate Change.** The effects of anthropogenic climate change on the Delta are covered in Dettinger et al. (submitted). Extreme weather patterns in recent decades indicate that climate change is already affecting the Delta ecosystem, making the water warmer and reducing outflows. Arguably, climate change is an additional stressor to smelt, one that is making it increasingly difficult for the species to recover. Changes in precipitation, air temperature, proportions of rain and snow, and runoff patterns are increasing. It is highly likely that water temperature will increase, and salinity intrusion will occur in the Delta (Cloern et al. 2011; Wagner et al. 2011). Brown et al. (2013) evaluated the effects of projected changes in water temperature, salinity field, and turbidity on Delta Smelt and determined that habitat suitability (see Feyrer et al. 2011) and the position of the LSZ during fall converged on values observed only during recent severe droughts (Brown et al. 2013). These more or less permanent changes are expected by mid-century. Higher water temperatures are expected to render much of the historic Delta Smelt habitat, from the confluence of the Sacramento and San Joaquin rivers and upstream, uninhabitable by smelt during summer and early fall. Such high temperatures will restrict distribution of smelt (Brown et al. 2013, 2016), likely interfering with maturation.
of fish and population fecundity (Brown et al. 2016), and inhibiting their recovery. Human responses to these events, such as changes in water project operations and new water infrastructure, are difficult to predict, but are not likely to favor Delta Smelt.

**Overview: Causes of Decline**

There is no “smoking gun” or single cause of the Delta Smelt decline. Instead, multiple factors have created habitat that is significantly less able to support smelt in large numbers (Figure 5). Moreover, the annual life cycle, relatively low fecundity, and current low abundance of this species increases probability of extinction due to stochastic effects in any given year. For example, droughts such as the one that began in 2012, worsen estuarine conditions for smelt, favor alien species, and generally create conditions that are likely to squeeze Delta Smelt between effects of natural stressors and anthropogenic stressors. Such droughts are likely to become longer and more severe as climate changes (Ingram and Malmud–Roam 2013). The lack of a single cause is not surprising considering that Delta Smelt is an annual species that lives in a highly dynamic and highly altered estuarine environment. The decrease of just one of its vital rates over a short period of time can cause a significant change in abundance.

Nevertheless, the outlook is not entirely bleak. The slight increase in Delta Smelt populations in 2011, a cool year with high outflows in spring and fall (Brown et al. 2013) suggests that outflows strongly interact with other factors, and can dilute toxicants, reduce temperatures, reduce entrainment, improve food supplies, and delay reproduction of potential predators and competitors. Higher outflows essentially allow more favorable habitat conditions for smelt to return to at least the north and west Delta. In addition, the capture of a few smelt in Montezuma Slough in Suisun Marsh every year suggests that some smelt move up and down the estuary even in dry years (W. Bennett, pers. comm. with P. Moyle, 2015, unreferenced, see “Notes”). There is also evidence that some smelt spend their entire life in the fresh waters of the north Delta, including the Sacramento Deepwater Ship Channel.

**DISCUSSION: THE FUTURE OF DELTA SMELT**

The Delta Smelt is well adapted for an estuary that no longer exists. Although we can only speculate on conditions for Delta Smelt in the historic estuary, it seems likely that Delta Smelt could always find abundant food and places to spawn and rear, whether in flood or drought, allowing it to remain abundant. The bulk of the population moved between the Delta and Suisun Bay, although presumably part of the population never left the fresh waters of the Delta, no matter what the conditions were like elsewhere. The Delta was originally a great wetland complex, absorbing freshwater outflows in winter and spring, and slowly releasing the water and the food it contained throughout the summer (Whipple et al. 2012). Delta Smelt were able to capitalize on rich food resources in a variety of habitats provided by the sloughs, backwaters, and channels of the entire historic Delta during winter and spring. As river inflows decreased and water temperatures warmed, larval and juvenile smelt could move, or be carried by the tides and rivers, into Suisun Bay and Marsh. There, the mixing of fresh and salt water created a concentration of planktonic organisms, ideal for plankton-feeding fishes, including Delta Smelt, Longfin Smelt, and Northern Anchovy. Similar conditions were probably present in the many isolated ponds present in the marshlands of the Delta and Suisun Marsh that may have sustained resident populations of smelt. No matter how wet, or how dry, a year might be, these conditions would have existed somewhere in the estuary, including the south and central Delta. In extreme wet years, most juvenile smelt might be advected to San Pablo Bay, while in dry years they might be retained in the Delta. Considering the dramatic changes to the estuary in recent decades, it is remarkable that Delta Smelt remained abundant through the 1970s; even though the estuary had changed markedly, the smelt still found the conditions they needed to thrive.

As discussed previously, human populations and water demand finally caught up with the smelt in the 1980s and its populations have spiraled rapidly downward as a consequence. The proximate causes of decline are interactions of multiple factors that have altered their habitat, making it increasingly unsuitable. None of these factors by themselves have caused the severe decline Delta Smelt has experienced
in recent years, but together they are devastating, transforming the Delta into a novel ecosystem (sensu Hobbs et al. 2009; Moyle et al. 2010; Morse et al. 2014) dominated by alien species, highly altered in structure, and generally inhospitable to Delta Smelt (Figure 7). This is the Delta described by Luoma et al. (2015) as a “wicked problem” with no single solution to its many conflicts and contradictions, requiring radically different management to have positive outcomes, such as prevention of Delta Smelt extinction.

Can the downward trend of the Delta Smelt be reversed? Does the Delta Smelt have a future in the estuary? We see three major alternative pathways: (1) complete extinction, (2) a conservation-reliant species with small populations, and (3) an uncommon species in an intensely managed arc of habitat in the north Delta and Suisun Marsh.

**Extinction**

The Delta Smelt appears to be on the pathway to extinction in the wild. All trends have been downward especially since 2002 (Figure 3). Delta Smelt have been almost undetectable in surveys since 2012. The discovery of freshwater resident smelt and continued persistence of small aggregations in Suisun Marsh provides some hope, but the population is likely so small that stochastic factors, such as continued drought, the loss of key spawning or rearing sites, or an increase in local abundance of competitors or predators (e.g., Mississippi Silverside) could cause extinction in the near future. The captive population at the UC Davis Fish Culture and Conservation Laboratory (Box 1) can prevent actual extinction for a while, but the loss of wild fish to interbreed with cultured fish to maintain genetic diversity will eventually result in domesticated smelt, best suited for survival inside the hatchery rather than outside of it. Reintroductions will have to be done within a few years of loss of wild fish, into an environment with better capacity to sustain them. One promising experimental approach would be to replicate culture techniques used in Japan for a similar smelt, Wakasagi (Mizuno 2012). Mats containing fertilized eggs that were spawned by

**Figure 7** The progression of the San Francisco Estuary to a novel ecosystem. Abiotic factors on the bottom axis, in concert with biotic factors on the vertical axis, have led to a system that supports a diverse array of introduced fishes, but has limited capacity to be restored to a condition that will support Delta Smelt (sensu Hobbs et al. 2009). The blue polygon represents historical conditions; the green polygon represents conditions that retain some feasibility of restoration to historic ecosystem structure or function; the orange polygons represent conditions that make restoration difficult or impossible; the red polygon represents conditions that dominate the novel ecosystem that typifies much of the Delta.
cultured fish (Box 1) could be placed in protected enclosures in food-rich environments, such as the flooded Yolo Bypass, or ponds such as those on Twitchell Island in the Delta. The eggs would hatch and larvae would live in the enclosures, before being released for a natural return to the Delta. This would be contingent, of course, on favorable conditions being present in the Delta.

Conservation-Reliant Species

“A species is conservation reliant when the threats that it faces cannot be eliminated, but only managed.”

—Goble et al. (2012), p. 869

This definition seems to fit the Delta Smelt well in its present circumstances. If it does avoid extinction, then it will only persist as a wild fish if its population is intensely monitored and managed. The focus may have to be on creating a more stationary freshwater sub-population, perhaps in a flooded island or in a reservoir outside the estuary. Alternatively, refuge areas could be created within Delta polders (islands) and perhaps the Napa River in which habitat quality is maintained, and potential competitors and predators controlled. The wild population would be critical for maintaining the genetic diversity of the captive population and the captive population may have to be used to help maintain the wild population during droughts. If increasingly unfavorable temperatures for smelt occur, predicted as a result of climate change, then special refuges may have to be created that can take advantage of cooler water in the Sacramento River or from water that is piped in from some other source.

North Delta Arc Species

Without massive investments in management, the south and central Delta are highly unlikely to continue to contain suitable habitat for Delta Smelt in most years. Realistically, habitat for a migratory population of Delta Smelt will have to be in the aquatic arc from Yolo Bypass, through the Cache–Lindsay Slough complex and the lower Sacramento River and into Suisun Bay and Marsh, a drastic reduction in its native range. Assuming temperatures stay cool enough, management programs will be necessary to maintain habitat quality including (1) invasive species control, (2) managing contaminants to keep concentrations low, (3) providing adequate flow down the Sacramento River at crucial times of year to promote environmental variability and transport of larvae, (4) providing high-quality habitat for spawning, (5) promoting production of the right food organisms in the right places for rearing, (6) keeping smelt out of the Central and South Delta and (7) thermal regime management. Such efforts, of course, could also provide major benefits to declining anadromous fishes such as Longfin Smelt, Chinook Salmon, and Green Sturgeon. In this scenario, the

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**BOX 1**

**Culture for Conservation**

As it became clear that the Delta Smelt was in severe decline, the UC Davis Fish Conservation and Culture Laboratory (FCCL) was established in 1996 at the State Water Project pumping plant in Byron, California. The purpose of the facility initially was to rear smelt in captivity for use in various experimental studies, because of their increasing unavailability in the wild. By 2004, the laboratory had the capacity to rear Delta Smelt through their entire life cycle. The program was remarkably successful in breeding a very delicate annual fish about which little was known in terms of culture (Lindberg et al. 2013). As a result, researchers had a ready supply of experimental fish. In 2008, the focus of the FCCL also became to establish a “refuge population” as a hedge against extinction in the wild. The breeding program was then set up to have strong genetic basis with reproductive success tracked for individuals and families. After starting with 2-year-old fish from the initial culture operation, wild fish were brought in every year to spawn with fish already in captivity, to enhance genetic diversity. The program has easily met its goals of having an annual spawning population of 500 fish, derived from a pool of 6,000 adults. An additional backup population was established at the Livingston Stone Hatchery below Shasta Dam. Ongoing studies are showing the difficulty of preventing domestication of cultured Delta Smelt, especially when wild adults are in short supply. For example, LaCava et al. (2015) showed a small but significant loss of genetic diversity after one generation of experimental breeding of Steelhead Trout.
number of smelt each year is likely to be directly proportional to the effort put into providing high-quality habitat for it.

CONCLUSIONS: LESSONS LEARNED FROM DELTA SMELT

The continued decline of Delta Smelt demonstrates a general failure to manage the Delta for the “co-equal goals” of maintaining the Delta as a healthy ecosystem while providing a reliable water supply for Californians. When the goals were first stated, the smelt and other native fishes were already in serious decline, so the ecosystem goal started on the path to co-equality from a position of great inferiority to the water supply goal. Efforts to manage Delta Smelt independently of its ecosystem, especially by reducing exports on an emergency basis when smelt approached the pumps in the South Delta, have reduced salvage but have not recovered the population. This is equivalent to treating the symptoms without acknowledging the disease. The condition of the smelt population is an indicator of the failure to manage the Delta as a valuable ecosystem that provides more than just fresh water for human use.

An opportunity for more ecosystem-based management of the Delta was presented in *Recovery Plan for the Sacramento–San Joaquin Delta Native Fishes* (USFWS 1995), the original recovery plan for Delta Smelt. The idea of the plan was to manage the Delta simultaneously for eight native fishes chosen because (1) they were known to be in decline, (2) they were important or historically important in the Delta ecosystem, (3) they depended on the Delta for a significant part of their life history, (4) the combined species required a wide range of conditions, so could collectively work for de facto ecosystem management, and (5) they were sufficiently well studied for managers to “make reasonable judgments as to measures that could reverse downward trends (USFWS 1995, p. 1).” At the time of the plan, the Delta Smelt was the only listed species; but even the section of the Recovery Plan devoted to just Delta Smelt had an ecosystem focus because it defined recovery by continued occurrence throughout the Delta as well as by total abundance. Ultimately, the plan was never adopted, because actions to protect Delta Smelt trumped actions for all other species under the Endangered Species Act. Since then, four of the seven remaining species have been listed as threatened or endangered.

Failure to implement a viable recovery plan has been instrumental in the decline of Delta Smelt and their virtual absence from the south and central Delta. Much of the Delta ecosystem has undergone irreversible changes, from estuarine conditions that favored native fishes to conditions that largely favor alien warm water fishes, invertebrates, and aquatic macrophytes (Moyle and Bennett 2008). The Delta is now a novel ecosystem, physically and chemically altered and dominated by alien species, to the point that going back to a past condition is no longer an option (*Figure 7*). Creating conditions that will allow native fishes such as Delta Smelt to exist in this novel ecosystem is a major challenge; it requires restoring at least some features of the historic environment, especially related to flows, and engaging in active management of other features (Moyle et al. 2010). As *Luoma et al. (2015)* state for the Delta in general, saving the Delta Smelt will require “finally and honestly embracing the equal value of water supply and ecological health (p. 5).”

The basic lesson from the collapse of Delta Smelt is that to save species, ecosystem-based actions have to be taken quickly to halt irreversible change, or at least to guide inevitable change in a more favorable direction. The longer the delay, the harder the decisions, and the less likely they are to produce positive results. For the Delta Smelt, the time to make key decisions for its survival in the Delta may have already passed.

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NOTES


Nagel M. 2016. Meredith Nagel, University of California, Davis. In-person conversation and e-mail with J. Hobbs about Delta Smelt clutch rearing and development in studies performed at the UC Davis Fish Culture and Conservation Laboratory.
ABSTRACT

As juvenile salmon enter the Sacramento–San Joaquin River Delta (“the Delta”) they disperse among its complex channel network where they are subject to channel-specific processes that affect their rate of migration, vulnerability to predation, feeding success, growth rates, and ultimately, survival. In the decades before 2006, tools available to quantify growth, dispersal, and survival of juvenile salmon in this complex channel network were limited. Fortunately, thanks to technological advances such as acoustic telemetry and chemical and structural otolith analysis, much has been learned over the past decade about the role of the Delta in the life cycle of juvenile salmon. Here, we review new science between 2006 and 2016 that sheds light on how different life stages and runs of juvenile salmon grow, move, and survive in the complex channel network of the Delta. One of the most important advances during the past decade has been the widespread adoption of acoustic telemetry techniques. Use of telemetry has shed light on how survival varies among alternative migration routes and the proportion of fish that use each migration route. Chemical and structural analysis of otoliths has provided insights about when juveniles left their natal river, and provided evidence of extended rearing in the brackish or saltwater regions of the Delta. New advancements in genetics now allow individuals captured by trawls to be assigned to specific runs. Detailed information about movement and survival in the Delta has spurred development of agent-based models of juvenile salmon that are coupled to hydrodynamic models. Although much has been learned, knowledge gaps remain about how very small juvenile salmon (fry and parr) use the Delta. Understanding how all life stages of juvenile salmon grow, rear, and survive in the Delta is critical for devising management strategies that support a diversity of life history strategies.

KEY WORDS

Chinook Salmon, telemetry, Steelhead, otoliths, Delta Cross Channel, survival, migration
INTRODUCTION

All anadromous salmonid populations in the Central Valley must traverse the Delta twice during their life cycle: once as juveniles migrating toward the ocean and once as adults returning to their spawning grounds. However the functional role of the Sacramento–San Joaquin River Delta in the life cycle of anadromous salmonids depends on the extent to which different life stages use the Delta. For adult salmon, the Delta is a migration corridor through which they must pass on their homeward journey. For juvenile salmon, variation in origin (hatchery versus naturally produced), run, size, life stage, and life history strategy affects residence time in the Delta—actively migrating smolts (juvenile fish undergoing physiological transformation for entry into sea water) can travel through the Delta within days (Perry et al. 2010; Buchanan et al. 2013), whereas smaller rearing juveniles may reside in the Delta from weeks to months (Kjelson et al. 1982; del Rosario et al. 2013). These life stages have different rearing strategies and consequently make use of habitat in the Delta in very different ways.

Anadromous salmonids in the Central Valley display a wide range of alternative life history strategies (Healey 1991). Central Valley rivers harbor populations of Chinook Salmon (*Oncorhynchus tshawytscha*) and Steelhead, the anadromous form of Rainbow Trout (*O. mykiss*). The Delta is unique among large estuaries on the West Coast of North America in that four distinct runs of Chinook Salmon traverse the Delta: the fall, late-fall, winter, and spring runs, named for the timing of the adult upstream migration. The winter and spring runs are listed as endangered and threatened, respectively, under the federal Endangered Species Act, which affects water management in the Delta; Central Valley Steelhead are also listed as threatened (SWRCB 1999; NMFS 2009).

Although the four runs of Chinook Salmon express a range of life history strategies, adult upstream migration timing is just one dimension of a much wider array of life history strategies that may also be characterized by variation in juvenile rearing tactics. Classic examples include stream-type versus ocean-type Chinook Salmon where juveniles of stream-type Chinook Salmon spend 1 to 2 years in fresh water and juveniles of ocean-type Chinook Salmon spend just a few months in fresh water before they migrate to the ocean (Healey 1991). Most Chinook Salmon juveniles in the Central Valley express an ocean-type rearing life-history strategy, but vary in how much time they spend in the different habitats between runs and life-stages before ocean residence. Williams (2012) identified at least six alternative life history strategies used by juvenile Chinook Salmon from the four runs. For example, juvenile salmon may rear entirely within their natal tributary and then migrate quickly downstream as smolts through mainstem rivers, the Delta, and San Francisco Bay (Figure 1). Alternatively, some juveniles leave their natal tributaries as fry or parr and spend considerable time rearing in mainstem rivers or the Delta before they enter the ocean (Figure 1). Thus, a specific life history trajectory can be defined as a unique spatial pattern of habitat use over time (Figure 1; Mobrand et al. 1997). However, life-history trajectories should be thought of as a continuous spatio-temporal distribution of habitat use, with dominant modes being classified as a particular trajectory.

![Figure 1](image-url)  
**Figure 1** Conceptual model showing three alternative life history trajectories of juvenile Chinook Salmon illustrating different patterns of habitat use in the Sacramento–San Joaquin River Delta. Juvenile salmon may (A) reside in natal tributaries until migrating quickly to the ocean as smolts, (B) rear upstream of the Delta but also spend considerable time growing in the Delta before seaward migration, or (C) rear wholly in the Delta before emigrating. Many other life history trajectories are possible, each forming a unique pattern of habitat use over time or developmental stage. Adapted from Mobrand et al. (1997) and Williams et al. (2012).
Diversity in life history trajectories within and among salmon populations buffers them against spatiotemporal variations in the environment. Termed the “portfolio effect” by analogy with stable returns from a diversified stock portfolio, a diversity of life history strategies has been shown to support more stable population trajectories over the long run (Schindler et al. 2010; Carlson and Satterthwaite 2011). In contrast, lack of diverse life history strategies results in populations that exhibit synchronous spatio-temporal patterns in habitat use, which can lead to boom-and-bust cycles when most individuals simultaneously experience favorable or unfavorable environmental conditions (Moore et al. 2010; Thorson et al. 2014). Lack of juvenile life-history diversity in Central Valley fall-run Chinook Salmon has been implicated as one of the primary causes of the collapse of this population in 2008 (Lindley et al. 2009).

Observed variation in life history strategies of salmonids in the Central Valley likely evolved from the diverse array of habitats historically present within the Central Valley—from seasonally inundated flood plains to the diurnally fluctuating tidal channels of the Delta to San Francisco Bay. Such diverse environments allowed a wide array of alternative strategies to be successful, i.e., for fish adopting a particular strategy to survive to eventually contribute to spawning. While the Delta historically played a critical role in supporting all life stages of juvenile salmon—fry, parr, and smolts—the contemporary Delta has been homogenized to such an extent that salmon populations must now contend with an alien environment, compete with alien species, and evade alien predators (Luoma et al. 2015). Thus, managing the Delta with the aim of recovering salmon populations rests on understanding how habitat and flow complexity affects the expression and maintenance of alternative life history strategies, and in turn, how each life history strategy contributes to the composition of Chinook Salmon populations in the Central Valley.

Understanding how different juvenile life stages of anadromous salmonids from the different runs use and survive in the Delta poses significant challenges. As juvenile salmon populations enter the Delta from upstream tributaries, populations disperse among the Delta’s complex channel network (Figure 2). This dispersal process is driven by the relative quantities of discharge that enters each of the Delta’s channels, the horizontal distribution of fish (which likely varies by life stage) as they pass a channel junction, and tidal cycles that alter flow patterns at channel junctions (Perry et al. 2015; SJRGA 2013). Once fish enter a given channel, they are subject to channel-specific processes that affect their rate of migration, vulnerability to predation (Grossman 2016), feeding success, growth rates, and ultimately, survival. Water management actions alter the distribution and quantity of flow through the Delta’s channel network, and therefore influence the spatial distribution, habitat use, and route-specific survival of juvenile salmon on their seaward migration. Eventually, alternative migration routes converge at Chipps Island and the population once again comes together to migrate through San Francisco Bay.

In this paper, we review new scientific insights about juvenile salmon in the Delta over the last decade. Our review focuses on new research that sheds light on (1) how juvenile salmon distribute and survive within the Delta’s channel network, and (2) how alternative life history strategies from the various runs use the Delta. The Delta must support a diversity of life history strategies if salmon populations are to persist in the face of climate change and other anthropogenic factors (Healey 1994; Healey and Prince 1995). Understanding how juvenile salmon that express alternative life history strategies distribute, rear, and survive in the Delta forms the knowledge base for understanding the effect of habitat and management actions aimed at protecting salmon populations.

**GROWTH OF JUVENILE SALMONIDS IN THE DELTA**

Growth rate is a critical metric for understanding how alternative life-history strategies perform because it integrates fish response to both biotic (e.g., competition, food quality and quantity) and abiotic (e.g., temperature) conditions over time, and because growth rate responds more quickly to changing conditions than metrics such as population size (Delta ISB 2015). However, growth of fish in their natural environment is difficult to measure without a large-scale mark-and-recapture study with individually
Figure 2  Map of the Sacramento–San Joaquin Delta showing important river channels and locations discussed in this review.
tagged fish. Growth is often estimated as a change in mean size of sampled fish between location and sampling times or a change in mean size of batch-marked fish (e.g., coded-wire-tagged [CWT] fish) between release and recapture. Unfortunately, these measures of growth can be biased by size-selective sampling and mortality and mask variability in individual growth.

New findings based on structural and chemical otolith analysis (see **Box 1**) provides some insight about growth in the Delta and San Francisco Bay. For example, Miller et al. (2010) examined ratios of the concentrations of metals to calcium in the otoliths of harvested adult fall-run Chinook Salmon, which follow an ocean-type life history, and compared them to juvenile growth patterns determined from the otolith structure. They found that in 2003 and 2004 nearly 70% of returning adults had entered brackish waters as fry (typically ≤ 55 mm in fork length) or parr (56–75 mm) rather than as smolts (>75 mm). Some adults that had exited freshwater as fry or parr passed into marine waters quickly, but approximately 25% displayed noticeable growth within brackish waters. Although previous research reported little growth among juvenile Chinook Salmon during estuarine residence in San Francisco Bay (MacFarlane and Norton 2002), the findings of Miller et al. (2010) suggest that estuarine rearing was a life-history strategy that contributed to adult returns. Other river deltas and estuaries on the West Coast also serve as important rearing areas for fry or parr, including the Columbia, Skagit, and Fraser deltas, as well as those of river systems on Vancouver Island in British Columbia (Healey 1991; Greene et al. 2005; Bottom et al. 2005).

**SUCCESSFUL JUVENILE LIFE-HISTORY STRATEGIES IN THE DELTA**

A wide array of life-history strategies is observed within the Delta, and maintaining habitat to support all life-history strategies is important for population resilience. Understanding which strategies are actually successful—i.e., strategies in which juveniles ultimately survive to return as adults—can provide critical insights into the role of the Delta in salmonid population dynamics. In the past decade, chemical and structural otolith analyses have led to

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**BOX 1**

**Using Chemical and Structural Otolith Analysis to Reconstruct Juvenile Rearing Strategies**

Otoliths are small bones found in the inner ear of vertebrates, including fish, and form part of the vestibular system. As the fish grows, the otolith accrues daily growth rings, similar to a tree laying down annual growth rings. The width of the otolith ring reflects the growth rate: wide rings indicate fast growth, and narrow rings indicate slow growth. Just as tree rings from a mature tree provide a history of the tree’s growth through its life span, an otolith removed from an adult fish provides information on the juvenile growth of the fish, starting from emergence from the gravel.

Statistical models have been developed that relate otolith size to fish size (body length, commonly measured as fork length). Thus, via “structural analysis” of the otolith, it is possible to reconstruct the fish’s juvenile growth patterns. Structural analysis has also been used to distinguish between naturally produced and hatchery reared adults (Barnett–Johnson et al. 2007).

Recent analytical methods have focused on chemical analysis of the otolith. The otolith is made of calcium carbonate taken from the water, but also stores trace elements reflective of environmental conditions in the rearing areas. In particular, strontium (Sr) and barium (Ba) isotopes are both found in otoliths. Both metals may vary in their environmental concentration along a headwater-to-ocean gradient, allowing researchers to make inferences about the rearing environment of juvenile fish from the concentrations of these elements, relative to calcium, in the otolith taken from the adult fish. The ratio of strontium to calcium (Sr:Ca) is found in higher concentrations in marine water, while the ratio of barium to calcium (Ba:Ca) is in higher concentrations in freshwater. By examining ratios of these metals to calcium in different parts of the otolith, it is possible to identify which part of the otolith was formed during the juvenile transition from freshwater to brackish or marine environments. Combining this result with the statistical relationship between otolith size and fish length, it is possible to estimate the size of the fish when it left freshwater (Miller et al. 2010).

A related approach tracks strontium isotope ratios (87Sr/86Sr) in the otolith to reconstruct juvenile rearing and migration history. The 87Sr/86Sr ratio varies in the water of different freshwater streams in the Central Valley, and analysis of the isotope ratio from returning adults (i.e., harvest or spawners) can yield information on rearing type (natural vs. hatchery), natal stream, and size at exit from the natal stream (Sturrock et al. 2015).
important new insights about variation in the success of different juvenile life history strategies (Box 1). In their otolith analysis of adult Central Valley Chinook Salmon, Miller et al. (2010) found evidence of multiple juvenile life history strategies: among 99 adult fall-run Chinook Salmon from the Central Valley that were harvested in an ocean fishery and that had entered the ocean as juveniles in 2003 and 2004, 48% had left freshwater as parr, 32% as smolts, and 20% as fry. They were also able to detect evidence of prolonged rearing in brackish waters among approximately 25% of the parr migrants, 55% of fry migrants, and 3% of smolt migrants (total of 18 individuals), suggesting that estuary rearing (the Delta and San Francisco Bay) was more important to overall success than previously thought.

Sturrock et al. (2015) used otolith isotope analysis and structural analysis to reconstruct fish size and life stage at juvenile emigration for returning spawners that had emigrated from the lower Stanislaus River in 2000 and 2003. They compared the relative frequency of various juvenile life stages (fry, parr, and smolt-sized fish), that left the Stanislaus River as inferred from the otolith analysis to the relative frequencies observed from juvenile sampling at rotary screw traps during those years. This approach allowed them to estimate the long-term probability of surviving to adult spawning for different life stages at emigration from the Stanislaus River. Although fry and smolts collectively formed the majority of the juvenile emigrants from the Stanislaus River, a higher proportion of surviving adults had emigrated from the Stanislaus River as parr than as either fry or smolts in both years. Survival estimates from juvenile emigration to adult spawning were 0.0178 to 0.0274 for parr, compared to 0.0007 to 0.0019 for fry, and 0.0077 to 0.0120 for smolts, but 95% confidence intervals for parr and smolts overlapped (Sturrock et al. 2015). They further found that the fry outmigration strategy contributed little to adult returns in the dry year but up to 20% of the adult returns in the wet year.

Both Miller et al. (2010) and Sturrock et al. (2015) found that a high proportion of the adult catch was composed of parr emigrants and that the contribution of fry emigrants varied among years. These findings suggest that management that promotes a diversity of life-history strategies in the Delta is likely to be more effective at improving population resiliency than that which focuses on one life-stage or one habitat type. Sturrock et al. (2015) also concluded that improvements in the estimates of the emigrating juvenile population size would facilitate efforts to understand the role of fry and parr in salmon population dynamics.

**RUN TIMING, COMPOSITION, AND ABUNDANCE**

Quantifying the contribution of different life history strategies to each race requires knowledge of when juveniles of specific life stages from specific runs occupy the Delta. Run timing is also particularly important for understanding when threatened and endangered runs are present in the Delta so protective water management actions can be implemented. Although juvenile salmon are present in the Delta in all months of the year (with peaks in winter and spring, Erkkila et al. 1950), understanding variation in juvenile migration timing and abundance among runs is complicated by overlapping size distributions among the runs during much of the year (Fisher 1994; Yoshiyama et al. 1998). In addition, identifying basin of origin is difficult because spatial distributions of juvenile salmon from the Sacramento and San Joaquin basins overlap in the much of the Delta.

Two separate length-at-date models have been widely applied to quantify the race composition of fish sampled at different monitoring locations in the Delta based on their length on a specific date: Fisher (1992) (“river model”) and Harvey et al. (2014) (“Delta model”). However, because fish of a given size from different runs may occupy the Delta at the same time, genetic analysis has revealed that these models are inaccurate, with the magnitude of bias depending on run, sampling location, and time of year (Hedgecock 2002; Pyper et al. 2013a; Harvey et al. 2014). Recent work from genetic analysis of tissue samples from fish collected in the Chipps Island trawl and at the pumping station fish facilities found that the river and Delta length-at-date models overestimated the proportion of winter and spring runs and underestimated the proportion of fall and late-fall runs (Figure 3). Bias in the estimates of race composition in fish sampling programs hampers our
understanding of run-specific life-history attributes (such as size, timing, and relative abundance) in the Delta. In the future, implementation of a genetic sampling program at long-term fish monitoring stations (e.g., Chipps Island) would provide unbiased estimates of run composition and timing of juveniles from the different runs that migrate through the Delta (IEP–SAG 2013).

Despite their biases, the length-at-date models are used for to classify runs and understand the potential life-history strategies of specific runs. For example, del Rosario et al. (2013) analyzed migration patterns of winter-run-sized Chinook Salmon (using the river length-at-date criteria) by examining differences in cumulative catch curves at two sampling locations: Knights Landing (51 km upstream of Sacramento), and Chipps Island (at the terminus of the Delta). Average residence time of winter-run-sized fish in the Delta was nearly 3 months. Median entry time into the Delta varied among years from mid-November to late January, and was strongly linked to the first major freshet of the winter. Yet among years, winter-run-sized fish exited the Delta over a narrow window in March. Consequently, between 1999 and 2007, median residence times ranged widely from 41 to 117 d. Although uncertainties remain regarding the length-at-date models, this research

Figure 3  Comparison of run assignments based on length-at-date criteria versus DNA (observed and corrected) for juvenile Chinook Salmon caught in Chipps Island trawl and DNA assigned to run (Pyper et al. 2013a). For each sample year and assignment method, the percentage of total juveniles assigned to each run is shown. Sample year 2008 is defined as August 1, 2007 through July 31, 2008, and similarly for 2009, 2010 and 2011.
sheds light on how migration strategies are linked to environmental cues that affect entry and residence times of winter-run-sized juvenile Chinook Salmon in the Delta.

Estimating absolute abundance of juvenile Chinook Salmon in the Delta has long been a goal of juvenile fish monitoring programs but has been difficult to achieve in practice (Dekar et al. 2013; IEP–SAG 2013). Juvenile abundance is an important parameter, particularly when viewed in a life-cycle context relative to the number of spawners or to abundance at some other point in time or space. “Abundance” may be estimated in two distinct ways: (1) by estimating the number of fish that pass a fixed sampling location over time (e.g., using trawls or screw traps), or (2) by estimating abundance over some spatial area at a particular point in time (e.g., using beach seines). However, in both estimation methods, the sampling gear captures only a fraction of the fish present. Consequently, estimating abundance requires expanding catch by an estimate of capture probability. Estimating capture probabilities for all gears, within and between years, for the size range of juvenile salmon caught has been difficult to achieve (Pyper et al. 2013b). Catch is often used as an index of abundance (i.e., relative abundance) under the assumption of constant capture probability. Inferences about trends in abundance drawn from catch data may be seriously biased if capture probability varies over time or with environmental variables such as flow or temperature.

To estimate abundance of juvenile salmon, monitoring programs have recognized the need to explicitly estimate capture probability and factors that affect capture probability (Dekar et al. 2013; IEP–SAG 2013; Pyper et al. 2013b). To estimate abundance of juvenile salmon passing Chipps Island, Kimmerer (2008) used the “fish flux method,” which assumed that capture probability was proportional to the fraction of the water volume sampled multiplied by the migration speed past the trawl. Subsequently, Pyper et al. (2013b) conducted an extensive analysis using three different data sets and analytical techniques to estimate capture probability from releases of CWT fish. Capture probability varied considerably from year to year, and this variation was not explained by covariates that would be expected to influence capture probability. Mean capture probability estimates ranged from 0.006 to 0.012 compared to 0.04 from the fish flux method. Pyper et al. (2013b) cautioned against using the fish flux method because it considerably overestimated capture probability relative to empirical estimates obtained from CWT fish and could, therefore, seriously under-estimate true abundance.

To estimate the absolute abundance of juvenile salmon in the Delta from catch at monitoring sites, much work remains to develop sampling designs that account for capture probability. Ongoing efforts to develop such methods stand to considerably improve our understanding of abundance and its relationship to population dynamics and life history strategies employed by different runs of salmon in the Central Valley.

**SURVIVAL OF JUVENILE SALMONIDS IN THE DELTA**

Understanding how juvenile salmonids of different life stages and runs survive in the Delta is critical for devising restoration and management actions. Between the 1970s and 2006, mark–recapture studies using CWTs formed the basis of research to estimate survival of subyearling fry and smolts of Chinook Salmon in the Delta (Kjelson et al. 1982; Brandes and McLain 2001; Newman and Rice 2002; Newman 2003). In general, this methodology involved marking tens of thousands of fish, releasing them at various locations in the Delta, and then recapturing them using a mid-water trawl at the outlet of the Delta at Chipps Island and in the commercial ocean fishery. Different release locations served as reference points to compare the relative probability of surviving through the Delta via different migration routes. Various statistical methods have been used to analyze the CWT studies, ranging from simple regression analysis of survival “indices” based on expansion of trawl counts (Kjelson et al. 1981; Kjelson and Brandes 1989; Brandes and McLain 2001) to sophisticated Bayesian hierarchical models that account for the multinomial structure of recapture data and for multiple sources of variation (Newman and Rice 2002; Newman 2003).

Recent analyses summarized the major CWT experiments occurring in the Delta through 2006, including both Sacramento River and San Joaquin
River studies (Newman 2008; Newman and Brandes 2010). These studies found modest evidence that closure of the Delta Cross Channel (DCC) gate improved survival of fish released at Sacramento. In addition, mean survival of fish released into Georgiana Slough, who then migrate through the interior Delta (the region to the south of the Sacramento River; Figure 2), was only 35% to 44% of the mean survival of fish that remained within the Sacramento River. Newman (2008) also found that for San Joaquin River fish, survival through the Delta was lower for fish that entered the Old River compared to those that continued to migrate down the San Joaquin. Newman (2008) also found weak (non-significant) evidence of a positive effect of export rate (the amount of water pumped out of the Delta) on survival of San Joaquin River fish, but noted that the tendency of high exports to occur with high inflow made it difficult to draw firm conclusions.

Although the CWT studies provided important information that helped us understand how water management actions affect the survival of juvenile salmon, major knowledge gaps remained. For example, until recently there remained a major lack of information about how juvenile salmon were distributed among the Delta’s channel network once they entered the Delta. This understanding is critical, because even though survival may differ drastically among migration routes, the effect of each route on total survival in the Delta depends on the fraction of the population that uses each route. Because water management actions may differentially affect population components in different regions of the Delta, understanding both how survival varies among routes, and how fish distribute among routes, is critical for understanding how management actions at local scales affect total survival.

Starting in 2006, biotelemetry techniques began to replace CWT studies as a way to quantify migration behavior and survival of juvenile salmon in the Delta. Application of biotelemetry techniques entails deployment of telemetry monitoring stations at key locations throughout the Delta, implanting small transmitters into juvenile salmon, and then tracking their migration through the Delta (Box 2). In spatially complex settings such as the Delta, biotelemetry has a number of advantages over traditional mark–recapture techniques that rely on the physical recapture of fish (e.g., CWTs). First, uniquely identifiable transmitters provide detailed information about the temporal and spatial movements of individuals migrating through a network of telemetry stations in the Delta. Second, the “capture” process is passive, so that an individual may be “captured” numerous times as it migrates unimpeded through the Delta. Third, the spatial arrangement of telemetry stations in the Delta can be tailored to the Delta’s complex channel structure to quantify both movement among and survival within given migration routes. Last, because telemetry receivers are able to monitor for tagged fish continuously, detection probabilities are typically high (>0.75), and small sample sizes can yield high precision of parameter estimates.

Although mark–recapture statistical models have long been applied to telemetry data on migrating fish populations (Skalski et al. 1998, 2002), these statistical models had to be adapted to accommodate the spatial complexity of the Delta. Perry et al. (2010) developed a multistate mark–recapture model that was tailored to the hierarchical branching structure of the Delta (Box 2). This research represented a landmark advance in estimating the underlying components of survival of juvenile salmonids that migrate through the Delta.

For hatchery late–fall–run Chinook Salmon that emigrate from the Sacramento River, analysis of telemetry data has led to new insights about survival through the Delta and has supported findings of previous CWT analyses. Overall, survival through the Delta for hatchery-origin late–fall–run Chinook Salmon ranged from 0.17 to 0.54 for migration years 2006–2010 (Perry et al. 2010, 2013). This range of survival appears to be lower than observed in other large West Coast watersheds. For example, when survival rates of yearling Chinook Salmon in other systems are scaled to the same migration distance of the Delta (81 km via the Sacramento River), mean survival was 0.92 in the lower Columbia River (McMichael et al. 2010) and 0.67 in the Fraser River (Welch et al. 2008). Among migration routes, survival of juvenile salmon emigrating within the Sacramento River was always greater than twice that of fish entering the interior Delta via the DCC and Georgiana Slough, similar to findings of Newman (2008).
Translating Telemetry Data into Routing and Survival Estimates

One of the most important advances in juvenile salmon research over the past decade has been the use of telemetry techniques combined with multistate mark–recapture models to estimate survival and the proportion of fish using different migration routes. Here we take the opportunity to describe in a bit more detail how researchers design, implement, and estimate survival and movement parameters for these studies. For more detail, interested readers should consult Perry (2010), Perry et al. (2010), and Buchanan et al. (2013), and for a more general treatment of multistate mark–recapture models, see Nichols and Kendall (1995) and Lebreton and Pradel (2002).

A major challenge with quantifying survival of juvenile salmon in the Delta is the extreme complexity of the Delta’s channel network. Historically, survival was estimated between a release and recapture point using coded-wire tags but there was little understanding of how fish used alternative migration routes between release and recapture points. Although telemetry techniques allow researchers to track migration pathways used by individual fish, translating these detections into robust survival and routing estimates requires design of a statistical mark–recapture model. Telemetry system design (i.e., the spatial arrangement of antennas or hydrophones in the Delta) dictates the structure of the mark–recapture model and, hence, the survival and movement parameters that can be estimated from telemetered fish (see Figures B2.1A and B2.1B). Therefore, careful study design is critical to successfully applying mark-recapture models to a spatially complex system like the Delta. Study design involves (1) identifying parameters of interest, (2) designing the mark-recapture model to estimate these parameters, and (3) implementing the telemetry system required by the mark–recapture model. In our experience, many telemetry studies fail to follow these important steps, resulting in inability to estimate important survival and routing parameters.

Demographic parameters of interest are \( S_{hi} \), the probability of surviving from telemetry station \( i \) to \( i+1 \) within route \( h \), and \( \psi_{hi} \), the probability of a fish entering route \( h \) at river junction \( i \), conditional on fish surviving to junction \( i \) (Figure B2.1B). To estimate survival within a specific migration route, telemetry receivers must be deployed just downstream of the entrance to each migration route (channel) at a junction where the river divides into two or more channels. This arrangement of receivers also allows for estimation of routing probabilities, \( \psi_{hi} \). For example, see sites A1 and B3 in Figure B2.1A at the junction of Sutter and Steamboat sloughs with the Sacramento River. Once again, the structure of the statistical model and estimated parameters follow directly from the spatial arrangement of telemetry receivers in the Delta (Figures B2.1A and B2.1B). The other important function of a mark–recapture model is to statistically distinguish between those individuals that have died in a given reach, and those that have survived but may not have been detected at downstream locations. Detection probabilities (\( P_{hl} \)) estimate the probability of detecting a transmitter given a fish is alive and the transmitter operational at telemetry station \( i \) within route \( h \). Although high detection probabilities are one of the strengths of using telemetry techniques, seldom are telemetry systems able to perfectly detect every individual migrating past a site. Thus, jointly estimating survival and detection probabilities prevents negative bias in survival estimates. These individual reach-specific survival and routing probabilities provide researchers a wealth of information about how juvenile salmon survive in specific reaches, and the proportion of fish entering a given migration route. However, these parameters may also be combined to estimate survival for an entire migration pathway through the Delta and the probability of fish using that pathway. The product of reach-specific survival probabilities that trace a specific pathway through the Delta estimates the total survival between beginning and ending points of the Delta (e.g., between Sacramento and Chipps Island) for fish that used that specific migration pathway (see bottom panel). These summaries provide a powerful way to compare alternative migration routes because they estimate survival between the same beginning and ending points of the Delta but for fish that used alternative migration pathways. Likewise, by multiplying the routing probabilities along a given migration pathway, we obtain the expected proportion of fish that used that migration route (see bottom panel). These provide the fundamental components that allow researchers to understand how survival within a migration route and the proportion of fish using that route influence total survival of fish migrating through the Delta.

(continued on page 11)
Figure B2.1  Approach for translating detections from a system of telemetry stations in the Delta into estimates of survival for each migration route. (A) shows the system of telemetry receivers that uniquely defines each migration route, (B) shows a schematic of the mark–recapture model and associated survival, detection, and routing parameters, and (C) shows how reach-specific survival parameters are summarized into survival for four different migration routes. Adapted from Perry et al. (2010).
Estimates of survival through the Delta for hatchery juvenile fall-run Chinook Salmon that emigrate from the San Joaquin River have been markedly lower than those from the Sacramento River. In addition, survival in the San Joaquin River has declined considerably since the 1990s and remained very low over the past decade (Figure 4). From 2003 to 2012, survival through the Delta has ranged from 0 to 0.11 among release groups, and has been ≤0.05 for 15 of 22 observations (Figure 4). Survival of San Joaquin River fall-run juveniles, estimated from telemetry tags, is considerably lower than survival of juvenile Chinook Salmon estimated from other large estuaries along the West Coast. When adjusted for different migration distances, juvenile Chinook Salmon survival has been estimated at more than 20 times greater in the Columbia River, and more than 15 times greater in the Fraser River, compared to San Joaquin River salmon (Buchanan et al. 2013).

Telemetry data provide more detailed spatial information on survival through the Delta than CWT data. In particular, unlike previous CWT studies, telemetry data have provided little evidence for the hypothesis that survival of Chinook Salmon is consistently higher for fish that remain in the San Joaquin River compared to those that enter the Old River (SJGRA 2013; Buchanan et al. 2013, 2015). It is uncertain how much the relative survival in these two routes may depend on river conditions and the presence of either a physical rock barrier or a non-physical barrier (e.g., a Bio-Acoustic Fish Fence, Bowen et al. 2012; Bowen and Bark 2012) installed at the head of Old River to prevent fish from entering Old River. The goal of these barriers is to divert fish away from the Old River and into the San Joaquin River because the Old River leads fish towards the State Water Project (SWP) and federal Central Valley Project (CVP) where they may be entrained into water pumping stations. Unlike a physical barrier, the non-physical barrier does not divert flow away from the Old River into the San Joaquin River; it is possible that this additional flow is needed to boost survival of fish that remain in the San Joaquin River.

![Figure 4](image_url)  
**Figure 4** Estimated survival of fall-run Chinook Salmon from either Durham Ferry, Mossdale, or Dos Reis to either Jersey Point (coded-wire tags, CWT) or Chippis Island (acoustic telemetry tags, AT). Intervals are 95% confidence intervals (truncated to 0 if necessary). Data sources: Holbrook et al. (2009), SJGRA (2013), Buchanan et al. (2015).
Telemetry data from Delta survival studies have shown that survival tends to be higher in the upper reaches of the Delta compared to lower reaches. In the Sacramento River, survival rate per kilometer generally declined along a downstream gradient, with lowest survival rates occurring in the interior Delta and the region around Cache Slough (Perry 2010). In the San Joaquin system, survival estimates of juvenile fall-run Chinook Salmon from the region near the Mossdale Bridge to Turner Cut averaged 0.30 for 2008–2012, while survival in all possible routes downstream of the Turner Cut junction to Chipps Island averaged only 0.11 in 2008 and 2010–2012 (Holbrook et al. 2009; Buchanan et al. 2013, 2015; SJRGA 2013).

Low survival of both Chinook Salmon and Steelhead entrained into the SWP and CVP has been observed from numerous telemetry studies (Clark et al. 2009; SJRGA 2011, 2013). Nevertheless, in 2010 and 2011, when overall Delta survival was less than 0.10 for San Joaquin River fall-run Chinook Salmon, the majority (19 of 26 tagged fish) that reached Chipps Island were observed passing through the salvage facility at the CVP, where they were subsequently transported and released just upstream of the terminus of the Delta at Chipps Island (SJRGA 2011, 2013). Thus, this route may be an important contributor to over survival during periods when survival through in-river migration routes is very low.

Overall, the past decade has provided a great deal of information on survival in the two major migration routes defined at the head of Old River. Patterns of survival in these routes have not been consistent, however, and it is not yet understood which factors (e.g., river flow, barrier presence and type) determine survival in either route. Throughout the past decade, survival of fall-run Chinook Salmon smolts has been low in both routes.

A considerable amount of new information has been gained in the past decade on survival in the upper portions of the south Delta. Less is understood about survival in the lower reaches of the San Joaquin River and in regions such as Frank’s Tract and the Old River and Middle River corridors. Studying survival in these regions is complicated by large river channels, strong reverse flows from tide, and attrition of the tagged population from mortality and entry to other migration routes before they reach downstream regions.

**MIGRATION ROUTING**

Understanding entrainment rates at river junctions (the proportion of fish that enter each channel) at the upstream periphery of the Delta—Sutter Slough, Steamboat Slough, the DCC, and Georgiana Slough on the Sacramento River; and the Old River, Turner, and Columbia cuts on the San Joaquin River (Figure 2)—is critically important because entrainment rates control the proportion of the juvenile salmon population that take a particular migration route, and thereby affect how the population is distributed among channels that have spatially variable transit times and survival rates. The Delta Cross Channel and Georgiana Slough branch off the Sacramento River and divert fish into the interior Delta, where survival probabilities are lower (Perry et al. 2010; 2013) and fish have a higher probability of being drawn towards water pumping stations in the southern Delta (Newman and Brandes 2010) than fish that remain in the Sacramento River. Sutter and Steamboat sloughs branch off the Sacramento upstream of Georgiana Slough and the DCC; thus, fish taking these routes are not subject to entering the interior Delta. On the San Joaquin River, fish first encounter the junction of the San Joaquin River and Old River. Fish that remain in the San Joaquin River may subsequently enter Turner and Columbia cuts, which lead toward the Old and Middle rivers, where net flows may draw fish toward the pumping stations.

Before the telemetry studies that began in 2006, estimates of fish routing at river junctions were few (Schaffer 1980; Kjelson et al. 1989), and the hypothesis was that juvenile salmon distributed among river channels in direct proportionality to the fraction of mean river discharge that entered each channel. However, since 2006, the use of telemetry techniques has markedly improved our understanding of how river flow, tidal dynamics, and barrier installation or gate operation affect the migration routes of juvenile salmon (Box 3). Given new tools to understand migration routing at river junctions, research increasingly has been focused on critical river junctions that may determine the ultimate
fate of juvenile salmon. Initial studies first provided point estimates of the mean proportion of fish using different migration routes over an entire release group (Perry et al. 2010; Buchanan et al. 2013). These studies were followed by research to understand how tidally-varying river flows affect the probability of an individual entering different river channels (Perry 2010; Perry et al. 2015; SJRGA 2013). More recent research has focused on evaluating behavioral guidance structures (e.g., non-physical barriers) as management tools to divert fish away from low-survival migration routes and toward high-survival migration routes (Perry et al. 2014; Bowen et al. 2012; Bowen and Bark 2012).

In addition to survival, the mark–recapture models developed by Perry et al. (2010) and Buchanan et al. (2013) provided the first estimates of the proportion of fish that enter different channels at key river junctions, parameters critical for understanding the fraction of fish that were subject to the survival rates of a given migration route (Perry et al. 2013, see also Box 2). For example, Perry et al. (2010) found that 16% to 20% of the fish arriving at the DCC/Georgiana Slough junction entered Georgiana Slough and 38% of the fish entered the DCC. These findings illustrate why accounting for routing at multiple river junctions is important for understanding how different fractions of the migrating population are subject to survival rates associated with different migration routes. Perry et al. (2013) further found that eliminating entrainment into the interior Delta (via Georgiana Slough and the DCC) could increase overall survival by 2% to 7%, given the route-specific survival probabilities estimated for six release groups between 2007 and 2010.

Research between 2007 and 2010 revealed that survival through the interior Delta was consistently lower than other migration routes for Sacramento River fish, which prompted managers to investigate use of non-physical barriers to alter migration routing at river junctions. On the San Joaquin River, there was also interest in keeping fish from entering the Old River, which directed fish toward the SWP and CVP. Therefore, a non-physical barrier known as a Bio-Acoustic Fish Fence (BAFF) was installed and tested at the entrance of Georgiana Slough on the Sacramento River and at the entrance to Old River on the San Joaquin River. The BAFF consisted of a curtain of air bubbles, sound, and flashing lights intended to elicit a behavioral avoidance response that would keep fish from entering these migration routes. On the Sacramento River, the BAFF was shown to reduce the proportion of fish entering Georgiana Slough from 22.3% to 7.7% (Perry et al. 2014). However, the BAFF’s effectiveness was shown to decrease with increasing discharge, likely because the fish were unable to avoid being swept into Georgiana Slough as water velocities increased. The physical barrier at the head of Old River was shown to better deter fish than the non-physical barrier at the head of Old River, although predation around both barriers was high (CDWR 2015).

On the San Joaquin River, the BAFF was tested in 2009 and 2010 to determine if it could be used to prevent fish from entering Old River. Both the BAFF configuration and placement in the river—as well as river flow and water velocity—were considered to be important factors in determining its effectiveness. However, during 2010 when the BAFF was present, survival was low in both the San Joaquin and Old River routes to Chipps Island, and there was no consistent survival benefit to remaining in the San Joaquin River. High flows prevented a third year of investigation into the BAFF in 2011, and a physical barrier was installed during the spring outmigration in 2012. From the telemetry studies in 2008–2011, the proportion of Chinook Salmon that entered the head of Old River ranged from 37% to 68%, and averaged 52% (Holbrook et al. 2009; SJRGA 2010, 2011, 2013). Tagged Chinook Salmon were less likely to enter Turner Cut, a tidally influenced junction that has no barrier, with estimates ranging from 0% to 32% (average = 14% in 2008–2012).

Many river junctions in the Delta are highly dynamic, with the direction and magnitude of flow that enters each channel varying on hourly time-scales with the tides. Consequently, the probability of a fish entering a given river channel will depend considerably on the time-specific hydraulic conditions it encounters upon arrival at a river junction. Furthermore, because fish behavior may determine both when fish arrive at a river junction and where fish are located in the channel’s cross-sectional profile, physical and behavioral processes may interact such that the mean
fraction of fish that enter a river channel deviates from the mean proportion of discharge that enters a river channel (Box 3). For example, Perry et al. (2015) showed both that the probability of a fish remaining in the Sacramento River ranged from near zero during reverse-flow flood tides to near one during ebb tides, and that the probability of a fish remaining in the Sacramento River was higher than the proportion of discharge remaining in the Sacramento River. This finding was supported by Cavallo et al. (2015) who compiled empirical routing estimates from multiple telemetry studies. They found that the proportion of fish that entered distributaries (secondary channels that branch off a main channel) was consistently lower than the fraction of discharge that entered the distributaries.

**Box 3**

**Entrainment Zones and the Critical Streakline**

The conceptual model we use to study entrainment rates of juvenile salmon at junctions is based on the entrainment zone and critical streakline concepts. This conceptual model illustrates why we should not expect fish to distribute among channels in junctions in direct proportion to discharge. Just upstream of a river junction, passive particles within the parcel of water entering a side channel (the entrainment zone) have a high probability of entering the side channel and a low probability of remaining in the main channel (Figure B3.1A). The extent of each entrainment zone is determined by the location of the critical streakline (the red lines in Figure B3.1 on page 16), defined as the spatial divide between parcels of water that enter a side channel or remain in the main channel. The location of the critical streakline can be found by integrating velocity vectors over the channel cross-section until the accumulated discharge just equals the discharge entering the side channel. Recent research supports the entrainment zone concept by showing that fish located on either side of the critical streakline have a higher probability of entering their respective channels (Perry et al. 2014). Based on this conceptual model, the only condition under which fish enter river channels in direct proportion to flow is when their cross-sectional distribution is uniform and constant over time (Figure B3.1A). Yet migrating and rearing juvenile salmon are seldom, if ever, uniformly distributed within a river channel (Horn and Blake 2004; Perry et al. 2014), leading to entrainment rates that deviate from the proportion of flow entering each channel (Figure B3.1A; Cavallo et al. 2014; Perry et al. 2015). However, entrainment rates can be predicted by understanding how the combination of critical streakline position and cross-sectional distribution of fish co-vary as a function of environmental variables such as tidal forcing, because the point at which the streakline bisects the fish’s spatial distribution determines the entrainment probabilities. Critical streakline positions can be estimated on a 15-minute interval using existing long-term flow monitoring data.

The velocity distributions in junctions within the tidally forced regions of the Delta are complex in both space and time (Figure B3.2) because ocean tides propagating into the Delta influence water levels, discharges, and velocity structures well into the upland fringes of the Delta. For example, the tidal currents reverse in all of the river junctions in the Delta during low flow conditions. The critical streakline concept is therefore a way of collapsing the complexity of the tidally forced flow fields to their essence with regard to fish fates. For example, instead of having to map every single velocity in the entire flow field and compute Lagrangian trajectories within an entire junction we need only compute a single path — the critical streakline. To deduce those behaviors that lead to a change of fate, we need only determine whether a fish crosses the critical streakline. Behaviors that keep fish within each entrainment zone do not ultimately change the fish’s fate. Therefore, the critical streakline concept has been used to good effect both in understanding why fish go where they go at junctions and in evaluating and optimizing the design of non-physical barriers. Thus, non-physical barriers that focus on moving fish from one side of the streakline to the other will effectively alter migration routing at a river junction.

(continued on pages 16–17)

**FLOW, EXPORTS, AND OTHER FACTORS THAT AFFECT SURVIVAL**

Nearly 30 years of CWT studies have formed the basis of understanding of how factors such as river discharge, gate operations, temperature, and turbidity affect the survival of juvenile salmon in the Delta. Analysis of CWT studies conducted in the Sacramento River demonstrated a positive effect of river flow on survival, a negative effect of water temperature, a negative effect of an open DCC gate, and a negative but sometimes non-significant effect of exports (Kjelson et al. 1981, 1989; Baker et al. 1995; Brandes and McLain 2001; Newman and Rice 2002; Newman 2003, 2008; Newman and Brandes 2010). For San Joaquin River fish, Newman (2008) found evidence of
Figure B3.1  Red regions denote the entrainment zone for the side channel, whereas the green regions show the region where fish continue along the main channel. The red line between these regions is the critical streakline. (A) shows the required conditions for fish to “go with the flow”—in this case, the bulk as-measured discharge in each channel. These conditions include the spatially uniform fish-entrance distribution that is shown and behaviors that do not result in fish crossing the critical streakline. (B) shows the conditions that create situations where fish are not distributed in proportion to the flows in each channel. These conditions include a non-uniform fish-entrance distribution, variable entrance timing, and behaviors that cause fish to transit the critical streakline.
Figure B3.2  Critical streakline dynamics at a typical river junction in the Delta as the tide changes from flood to ebb twice a day. Almost all channel junctions in the Delta exhibit this sequence of changing flow patterns during periods of low river inflow. The exceptions are junctions at the Delta Cross Channel and Georgiana Slough: in these junctions, reversing flows rarely occur and when they do it is weak and short-lived. Finally, this sequence, including the direction of movement of the streakline, can be reversed depending on the phase relation between the main and side channels. The white (unshaded) regions represent “slack water” or negligible velocity regions.
a positive effect of inflow on survival; he also found weak (non-significant) evidence of a positive effect of export rate on survival, but noted that the tendency of high exports to occur with high San Joaquin River inflow made it difficult to draw firm conclusions.

Although telemetry studies have been relatively small-scale to date, with only a few years of replication, analyses are now beginning to paint a fuller picture of how survival in the Delta varies with environmental variables. For example, Perry et al. (2010) found that survival of juvenile late-fall-run Chinook Salmon was positively related to discharge and fish size in the Sacramento River and in Sutter and Steamboat sloughs. Perry’s flow–survival relationship was very similar to that found by Newman (2003) when compared for a common reach and fish size, despite the different time periods, methods (coded-wire tag vs. acoustic tag), and fish sizes used in the studies (Figure 5).

One of the primary benefits of using acoustic telemetry rather than CWTs is the ability to estimate survival on smaller reaches throughout the Delta. Now that multiple years of telemetry data are becoming available, more work is needed to relate survival to environmental variables at finer spatial scales within the Delta to understand how these factors affect survival in different regions. For example, survival may be related to river flows upstream, but not in lower reaches of the Delta where the magnitude of tidal flows swamps net flows. Such hypotheses are difficult to test with CWT studies, but are feasible with telemetry. Moreover, river discharge, in and of itself, does not influence survival but rather is the master variable in the Delta (sensu Mount et al. 2012) that affects the underlying mechanisms that influence survival. For example, discharge affects turbidity and fish migration rates, both of which affect predator encounter rates and, ultimately, survival. Survival models such as the XT model—a predator–prey model that expresses survival as a function of travel time, travel distance, and predator densities (Anderson et al. 2005)—hold promise for helping us to better understand the mechanisms that underlie correlative relationships such as those in Newman (2003) and Perry (2010).

Inference about the direct effect of water exports on survival of juvenile salmonids in the south Delta has been based on salvage rates at the water export facilities, and the connection between salvage and entrainment loss. Salmonids that enter the water export facilities (i.e., either pass the trash racks at the CVP or enter the Clifton Court Forebay outside the SWP) are said to be “entrained.” Entrained fish may either enter the water conveyance canals, be diverted via louvers or screens away from the canals to a holding tank (“salvaged”) and transported by truck around the rest of the Delta, be preyed upon before reaching the fish guidance structures to the holding tanks (“pre-screen mortality” or “pre-screen loss”), or die during the salvage and transport process.

Salvage of CWT fish provides indirect evidence of mortality from exports under the assumption that a constant fraction of fish that enter the facilities is salvaged. Recent analysis of historical coded wire tag data from fall Chinook Salmon released in the San Joaquin River has found higher rates of salvage.
during periods of higher exports (Zeug and Cavallo 2014). Nevertheless, the overall proportion of tagged release groups recovered from salvage has been low (average <1% for both Sacramento River and San Joaquin River release groups from 1993 through 2007; Zeug and Cavallo 2014); however, the proportion salvaged does not account for the fish that die from being diverted off their migration routes before they arrive at the fish facilities, or how well salvage counts may reflect total entrainment in the facilities. Kimmerer (2008) estimated that at 10% pre-salvage survival (i.e., 90% combined loss from initial entrainment from both pre-screen loss and imperfect fish guidance efficiency at louvers or screens), the proportion of winter-run Chinook Salmon released in the Sacramento River that die in the fish facilities (“proportional loss”) could be as high as 30% at combined exports of 300 m³s⁻¹. When Zeug and Cavallo (2014) estimated the combined direct loss at the CVP and SWP relative to total migration mortality, the relative loss from entrainment from exports was as high as 17.5% for San Joaquin River releases.

Analyses of CWT data have compared recovery rates of fall-run Chinook Salmon released at either Durham Ferry or Mossdale on the San Joaquin River to those released at Jersey Point, using tags recovered from the trawls at Chipps Island and Antioch, and tags captured in the ocean fishery. Comparing recovery rates from upstream and downstream groups of fish in this way is an attempt to isolate survival through the Delta to Jersey Point from survival in the ocean. These analyses show positive relationships between survival to Jersey Point and flow when the barrier is installed at the head of Old River (SJRGA 2007, 2013). Another way of looking for an effect of flow is by comparing counts of adults that return to hatcheries and spawning grounds or migrate upstream past dams (“adult escapement”) with conditions during the juvenile outmigration 2.5 years earlier. Such comparisons have been made using both San Joaquin River flow at Vernalis and the ratio of Vernalis flow to export rates. Statistically significant associations were found between adult escapement and both Vernalis flow and the ratio of flow to exports: adult returns tended to be higher when Vernalis flows were higher, and also when the ratio of flow to exports was higher and the barrier had not been installed at the head of Old River (SJRGA 2007).

The effect of exports on survival in the immediate vicinity of the water export facilities is understood better than effects farther away. At the CVP, a fish insertion experiment found higher efficiency of the fish guidance structures (“louver efficiency”) for juvenile Chinook Salmon when water velocities in the intake canals were higher (Surpphin and Bridges 2008). Because export rate determines water velocity in the CVP, higher CVP export rates translates into potentially higher survival to salvage, at least for fish in the immediate vicinity of the CVP. Furthermore, an acoustic telemetry study at the CVP found a higher probability of juvenile Chinook Salmon entering the facility at higher water pumping rates, although Steelhead behavior was more variable (Karp et al. 2014, unreferenced, see “Notes”). At the SWP, the water pumping plant and fish collection facility are accessed through the Clifton Court Forebay (CCF), which is isolated from the Delta by radial gates that are opened several hours each day to allow freshwater to enter the reservoir; fish may also enter the CCF when the gates are open. Tagging studies using passive integrated transponder tags and acoustic telemetry tags in the CCF have estimated high pre-screen loss for Steelhead (0.78 to 0.82; Clark et al. 2009), which is similar to estimates for Chinook Salmon reported by Gingras (1997). Gingras (1997) also reported that pre-screen loss in the CCF declined as exports increased for Chinook Salmon. The Steelhead tagging studies in the CCF did not examine a relationship between pre-screen loss and export rate, but compared movement rate (i.e., time to salvage) within the CCF to export rate, and found no statistically significant relationship (Clark et al. 2009).

UNCERTAINTIES FROM TELEMETRY DATA

Although use of telemetry techniques has vastly improved our level of understanding about migration and survival dynamics of juvenile salmon in the Delta, these studies have important limitations. For example, most published studies to date have used large, actively migrating salmon smolts (e.g., late-fall-run Chinook Salmon of hatchery origin) because transmitter size restricts the minimum size of fish that may be tagged. Thus, fish used in telemetry studies may be larger than their naturally produced counterparts, larger (on average) than the hatchery population from which they are sampled,
and larger than most of the juvenile salmon that migrate through the Delta. Ongoing technological advancements continue to miniaturize transmitters. Thus, recent studies using the latest transmitter technology are able to tag fish as small as 90 mm, which allows for studying a wider array of sizes and run types of salmonids. On the Sacramento River, ongoing telemetry studies are now investigating migration and survival of winter-run and spring-run Chinook Salmon, runs of specific management interest because of their status as listed under the federal Endangered Species Act. In the future, we expect that these studies will provide important insights into run-specific survival in the Delta. Even with these new technologies that allow smaller fish to be tagged, understanding survival and rearing tactics of fry and parr will remain a significant knowledge gap, at least for the foreseeable future.

Most studies, even those now tagging smaller fish, still rely on hatchery-origin fish as their study subject. This situation will likely not change in the near future because it is difficult to catch enough naturally produced fish to tag at most monitoring locations in the Delta and because the origin of captured fish is often unknown. Because behavior and survival of hatchery-origin fish may differ from that of wild fish (Reisenbichler and McIntyre 1977; Kostow 2004), care must be taken when inferences are drawn from hatchery fish about survival of wild fish. For example, inferences from hatchery fish about absolute survival of wild fish may not hold, but factors that influence relative differences in survival among migration routes (e.g., interior Delta relative to Sacramento River) may act similarly on both wild and hatchery populations that migrate through the Delta during the same time period.

An additional complication with interpreting telemetry data is distinguishing between detections of surviving study fish and detections of predatory fish that have eaten the study fish and still have the telemetry tag in their gut. Depending on the spatio-temporal patterns of detection of salmon and predators of tagged salmon, detections from predators may bias survival estimates. This is particularly problematic for survival and behavioral studies on smaller spatial scales and near the water export facilities, where large populations of predators congregate and complex hydrodynamics may influence salmonid behavior in unknown ways. Telemetry studies in the Delta have made efforts to identify and remove detections suspected of coming from predators (SJRG 2010, 2011 and 2013; Buchanan et al. 2013; Romine et al. 2014). New statistical techniques to distinguish movement patterns of smolts from those of predators have recently been developed for spatially explicit two-dimensional (2-D) telemetry data (Romine et al. 2014) and for presence–absence detection data (Gibson et al. 2015). New telemetry tags that alert researchers to predation events are undergoing testing and may reduce uncertainty in interpreting telemetry data.

**SPATIALLY EXPLICIT MODELS OF SALMON IN THE DELTA**

Management actions that influence the quantity and distribution of water in the Delta affect how juvenile salmon populations distribute among and survive within those channels. Consequently, spatially explicit models are needed to understand how management actions at specific locations affect juvenile salmon survival within the Delta’s complex channel network (Rose et al. 2011; Delta ISB 2015). These models explicitly represent the Delta as a hierarchical channel network to simulate how fish move among and survive within different channels. Over the past decade, salmon simulation models have begun to explicitly represent the Delta’s channel structure to varying degrees of resolution and complexity.

Spurred by recommendations from a workshop on Central Valley salmonid life cycle models (Rose et al. 2011), NOAA is currently developing a stage-structured life cycle model that explicitly includes the Delta (Hendrix et al. 2014). This model has two key features critical to understanding the Delta’s role in the salmon life cycle. First, alternative life-history strategies of fry and smolts in the Delta are explicitly represented in this model. Entry timing and residence time of fry in the Delta is driven by a density-dependent response to habitat capacity in upstream habitats that vary as a function of discharge. Second, an agent-based model for the Delta is being developed based on DSM2-PTM, a particle tracking module to the widely used DSM2 hydrographic simulation model (Kimmerer and Nobriga 2008). Dubbed the ePTM (enhanced Particle
Tracking Model), this model adds salmon-like behaviors to the particles such as active swimming, nocturnal or diurnal migration, and selective tidal stream transport.

Another life cycle modeling effort that explicitly represents the channel structure of the Delta is the IOS model (Zeug et al. 2012), which uses the Delta Passage Model (Cavallo et al. 2011) to represent the Delta’s channel structure. The Delta Passage Model represents the Delta as a coarse network of reaches and channel junctions. This model simulates the travel times of fish through each reach; routing at critical channel junctions (e.g., the DCC and Georgiana Slough), which may depend on discharge when fish arrive at a junction, and survival within each reach, which may also be driven by discharge when fish enter the reach.

The current efforts to develop spatially explicit models for the Delta were made possible by the availability of spatially explicit data from acoustic telemetry studies. Because the acoustic telemetry data provide information about migration rates, survival, and routing of juvenile salmon in different regions of the Delta, this data is proving critical to setting parameters for spatially explicit models. For example, parameters were set for migration routing and survival relationships in the Delta Passage Model based on findings of telemetry survival studies. In addition, behavioral parameters in the ePTM are being estimated by calibrating the ePTM to acoustic telemetry data.

Spatially explicit models of juvenile salmon in the Delta are in their infancy and have followed on the heels of acoustic telemetry studies that provide the data to inform these models. An agent-based model of juvenile salmon coupled to a 3-D hydrodynamic model of the Delta does not yet exist, but has been recognized as a critical need to understand key drivers, identify information gaps, and support management of water and fishery resources (Delta ISB 2015). Over the next decade, we expect these modeling approaches to mature as hydrodynamic models to drive salmon models, analytical techniques for fitting models to data, and computing resources to support model runs are developed.

**DROUGHT-RELATED EFFECTS**

Significantly less precipitation and warmer temperatures since the spring of 2012 have led to statewide drought conditions. These conditions resulted in record low flows and high water temperatures and likely caused substantial negative population-level effects on salmon populations. For instance, during 2014, estimated egg-to-fry mortality of naturally spawned winter-run juveniles was 95% because of water temperature during egg incubation. The full ramifications of the recent drought on salmon populations have yet to be realized until adults return to spawn. How juvenile salmonids in the Delta have been affected by drought is unknown.

Recent drought conditions led to many emergency water management strategies that are typically governed by the federal Endangered Species Act (NMFS 2009) and state water quality control plans. Emergency actions included changes in winter and spring reservoir release schedules, DCC gate and CVP/SWP export facility actions, and temperature control device operations at Shasta Dam. The drought and subsequent emergency actions have spurred a number of interagency teams to evaluate the likely effects of drought on salmon populations and to design monitoring frameworks that are better able to quantify population responses to drought at key “checkpoints” in the freshwater environment, including the Delta.

An interagency team developed a conceptual model to evaluate how reduced flows, increased DCC gate openings and exports, and additional constraints on flexibility of operating temperature control devices affected a number of physical and biological metrics (USBR 2015). Observations from fish and ecosystem monitoring for the 2013 cohort of winter-run Chinook Salmon were compared to outcomes from a recent comparative period (2007–2012). Based on this information, the drought affected multiple stages of winter-run Chinook Salmon through an extended period of the cohort’s freshwater residence during 2014 (Table 1), and will likely have consequences as this cohort returns to the river as adult spawners.

A larger analysis of environmental conditions and consequences for salmon during 2015 and 2016 is being pursued as part of the Interagency Ecological Program’s Management, Analysis and Synthesis
(MAST) project known as SAIL (Salmon Assessment of Indicators by Life stage). An important finding thus far is that finer temporal and spatial monitoring of each population cohort is necessary to understand spatial and population level responses to drought and other environmental factors. In addition, early life stage transitions may be affected more by management, physical, and biological mechanisms (e.g., rearing habitat and disease) than we can currently quantify with available monitoring data. This synthesis work is essential to understanding mechanisms and consequences of drought, and may yield insight into how to better cope with climate change.

**CONCLUSIONS**

Because of the complexity of the Delta’s channel network and its complicated sampling environment, answering questions about how juvenile salmon use, grow, and survive in and through the Delta has been a major challenge. Fortunately, during the past decade, technological advancements that have miniaturized acoustic transmitters and novel application of otolith microchemistry and genetic methodologies have resulted in the ability to obtain more detailed information about individual fish. These new technologies have advanced our knowledge of how different life stages and runs of juvenile salmon move, rear, and survive in the Delta. Knowledge gaps remain about the dynamics of naturally produced juveniles and of fry and parr life stages in the Delta. These life stages are expected to rear for some time in the Delta rather than migrating quickly to the ocean. Given additional focus on estimating capture efficiency and absolute abundance of juveniles in the Delta, researchers can begin to quantify how abundance varies with habitat characteristics to better understand the habitat needs of the juvenile salmonids that rear in the Delta for a considerable length of time. Improvements in juvenile fish sampling methods and juvenile population monitoring, combined with expanded chemical otolith analysis and genetic sampling, hold the potential to shed further light on these questions. By coupling fine-scale hydrodynamic models of the Delta with agent-based models of juvenile salmon, researchers are beginning to use models to understand how water management actions and climate change might influence movement and survival of juvenile salmon in the Delta.

Much has been learned about salmon in the Delta over the past decade, yet much remains to be learned. With continued drought and ongoing climate change, maintaining viable salmon populations will become even more challenging. Better understanding of how different life stages use the Delta will help inform management actions to ensure that the Delta is capable of supporting the diversity of life-history strategies expressed by Central Valley Chinook Salmon and Steelhead populations.

**ACKNOWLEDGEMENTS**

This work was supported by an honorarium from the USGS (for RWP) and the Delta Stewardship Council (for PLB and RAB). Many thanks to J.G. Romine for providing the map in Figure 2. We thank two anonymous reviewers for helpful comments that improved the manuscript.

Table 1  Effects of drought conditions and water operations in 2014 on the 2013 brood year of winter run Chinook Salmon. Adapted from USBR (2015).

<table>
<thead>
<tr>
<th>Metric</th>
<th>Brood years 2007–2012</th>
<th>Brood year 2013</th>
<th>Interpretation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estimated egg to fry survival above Red Bluff Diversion Dam</td>
<td>17.5–48.6%</td>
<td>15.1%</td>
<td>Lowest survival in natal rearing areas during drought conditions</td>
</tr>
<tr>
<td>Residence time above Knights Landing</td>
<td>65–164 d</td>
<td>133 d</td>
<td>Resided in river for similar period compared to recent years</td>
</tr>
<tr>
<td>Residence time in Delta Cross Channel</td>
<td>21–116 d</td>
<td>3 d</td>
<td>Resided in Delta for shorter period compared to recent years</td>
</tr>
<tr>
<td>Duration of salvage period</td>
<td>109–216 d</td>
<td>42 d</td>
<td>Resided in South Delta and present in Delta for shorter period compared to recent period</td>
</tr>
<tr>
<td>Duration of connectivity with Yolo Bypass</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
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NOTES

ABSTRACT

The Sacramento–San Joaquin Delta (Delta) is a heterogeneous, highly modified aquatic system. I reviewed relevant predator–prey theory, and described extant data on predator–prey relationships of Delta fishes. I ranked predator consumption rates as occasional, moderate, and common, based on frequency-of-occurrence data, and evaluated the frequency, and hypothesized the effects of predation on native and invasive species. I identified 32 different predator categories and 41 different prey categories. Most predators were occasional consumers of individual prey species, although I also observed moderate and common consumption of some prey types. My analysis yielded few generalizations regarding predator–prey interactions for Delta fishes; most predators consumed a variety of both native and invasive fishes. The only evidence for predator specialization on either native or invasive fishes occurred in Prickly Sculpin which, when it consumed fishes, ate mostly native species. Both Striped and Largemouth Bass exhibited wide dietary breadth, preying upon 32 and 28 categories of fish prey respectively. Sacramento Pikeminnow, a native predator, also displayed wide dietary breadth of piscine prey, with 14 different prey categories consumed. Data for reptilian, avian, and mammalian predators were sparse; however, these predators may be significant fish predators in altered habitats or when hatchery salmonids are released. The database for predators and their fish prey was not strong, and I recommend long-term dietary studies combined with prey availability and behavioral and experimental studies to establish predator preferences and anti-predator behaviors, rather than just consumption. The behavioral effects of contaminants on prey species also warrant further examination. Although it has been suggested that a reduction in the Striped Bass population be implemented to reduce predation mortality of Chinook Salmon, the large number of salmon predators in the Delta make it unlikely that this effort will significantly affect salmon mortality.

KEY WORDS

Delta Smelt, Longfin Smelt, Chinook Salmon, Rainbow Trout, Striped Bass, Largemouth Bass, Pikeminnow, predator–prey interactions
INTRODUCTION

The Sacramento–San Joaquin Delta (the Delta) is the eastern portion of the largest estuarine system on the West Coast of the Americas (San Francisco Estuary), encompassing an area of 3,238 km$^2$ (Whipple et al. 2012). This system provides critical habitat for plants and animals with over 700 species recorded, as well as providing irrigation water to farms in the highly productive Sacramento and San Joaquin valleys. In the last 150 years, human effects on the Delta include a variety of environmental changes (DSC 2013; Wiens et al. 2016) including:

1. physical (channelization, flow control, water export),
2. chemical (discharge of toxins, nutrients), and
3. biological (species invasions, conversion of floodplain to agriculture) characteristics, most of which have produced declines in habitat quality for native species.

These decreases in habitat quality in concert with the effects of introduced species, are linked to major declines in formerly abundant native species such as Chinook Salmon (Oncorhynchus tshawytscha), Steelhead (Oncorhynchus mykiss), Delta Smelt and Longfin Smelt (Hypomesus transpacificus and Spirinchus thaleichthys, respectively), the extinction of the Thicktail Chub (Gila crassicauda), and the local extinction of the Sacramento Perch (Archoplites interruptus). Population declines have not just occurred in native species; both Striped Bass (Morone saxatilis) and Threadfin Shad (Dorosoma petenense) display recent decreases in abundance (Sommer et al. 2007).

There are few data on either the historic or current effects of predators on the abundance, size, structure, and dynamics of Delta fish populations. Nonetheless, it is clear that native and invasive (I use the term invasive because it accurately represents the fact that these species have invaded a habitat) prey species run a gauntlet of potential predators including invasive predatory fishes (Striped Bass, centrarchids, percids and ictalurids), native and introduced snakes and amphibians (garter snakes and bullfrogs), mammals (North American river otter Lontra canadensis), and birds (herons, egrets, terns and gulls) (Draulans 1988). Because the Delta is a heterogeneous, highly modified aquatic system (Kimmerer et al. 2008; Dettinger et al., submitted; Luoma et al. 2015), it is a challenge to quantify the true effect of predation on fishes, given the ubiquity of co-varying factors (e.g., habitat degradation) capable of negatively affecting fish populations (Grossman et al. 2013; Wiens et al 2016). In this paper, I describe relevant predator–prey theory, and potential predator–prey interactions among resident predators and fish prey. Unfortunately, there is scant information on the interactions between predators and prey in the Delta; consequently, my main source of information comes from dietary studies of predatory fishes. This information includes the published scientific and selected gray literature references, as well as presentations at the 2013 Fish Predation Workshop (http://www.dfg.ca.gov/erp/predation.asp) and 2015 Interagency Ecological Program Workshop (http://www.water.ca.gov/iep/activities/workshop_events.cfm).

THE DYNAMICS OF PREDATION ON FISH POPULATIONS

How predation effects a fish population may be examined at varying levels of resolution, ranging from quantification of the frequencies, numbers, masses or volumes of prey species in gut contents, to estimates of the number of prey consumed per predator per day, to estimates of the proportion of the prey population consumed by each predator over a given time interval (day, month, season). The level of precision required to quantify predator effects depends on the question being posed. At the most basic level, gut content information tells us that a given predator consumes a given prey species, but yields limited information on the effects of predation on the prey species. Alternatively, lab experiments with a single predator and prey species yield information on the behavioral interactions of the species pair, but cannot tell us how the predator will behave in the presence of alternate prey or in “natural habitats.” The most detailed level of predator–prey studies involves quantification of the effects of a given predator on prey abundance and survivorship and requires measurement of multiple factors including: (1) the amount of predation mortality experienced by a prey population over time, (2) ultimate and proximate causes of mortality,
and (3) assessment of the role of predation in the regulation of prey abundance. Grossman et al. (2013) discuss the potential methods for approaching predation-effect studies for fish predators and salmonid prey that inhabit the Delta.

Despite the difficulty of the task, scientists and managers may require information on how predators affect fish populations, which, in addition to the factors listed above, requires: (1) accurate and precise estimates of predator and prey abundance, and (2) numerical and proportional estimates of predation mortality on the prey population or for a particular life-history stage (e.g., eggs, fry, and juveniles). Quantifying these parameters is both logistically and methodologically difficult, and obtaining estimates with reasonable accuracy and precision requires replication in space and time. For example, providing accurate data for step two requires experiments quantifying the functional and numerical responses of important predators on fish, coupled with annual and seasonal estimates of prey abundance and productivity (Grossman et al. 2013). Although this will produce an estimate of mortality attributable to predation, the estimate then must be compared to other sources of mortality (e.g., disease) to determine whether predators contribute significantly to population mortality through time. In the Delta, mortality from predation is just one of many potential sources of mortality, including: entrainment by water abstraction facilities, physiological stress, contaminants, food limitation, disease, and parasites, all of which may interact. Finally, mortality may be compensatory such that an increase in predation mortality may be offset by a decrease in disease mortality, with the overall mortality rate remaining constant. Because of the presence of compensatory processes in many fish populations (Grossman et al. 2006, 2012), quantifying the direct effects of predation on the total mortality rate of a population is difficult, at best (Kerfoot and Sih 1987).

The ultimate goal of predation studies is to quantify the importance of predator–prey relationships on both the dynamics of populations and the behavior of individuals. However, predation may be either the proximate or the ultimate cause of individual mortality, and identifying which of these two processes is operating may be problematical. In the Delta, decreased and altered timing of freshwater inputs, combined with water abstraction may alter flow patterns, especially in the east and south Delta (Kimmerer et al. 2008; Dettinger et al., submitted; Luoma et al. 2015). This may confuse prey species and shift their movements and migratory pathways in a manner that keeps them in greater contact with potential predators (Winder et al. 2011; Cloern and Jassby 2012). In this case, predation may be the proximate cause of mortality, whereas altered flow regimes are the ultimate cause. Clearly, the classification of mechanisms as either ultimate or proximate causes of mortality is an oversimplification, and the point at which a process moves from merely being an “influence” to a proximate cause, and, finally, an ultimate cause of mortality is typically unknown and, even when known, frequently subject to debate.

It is reasonable to assume, however, that some anthropogenic factors, especially contaminants known to affect fish behavior, are likely to be the ultimate cause of a significant amount of predation mortality for Delta fishes. Much more research is needed on the behavioral effects of contaminants on Delta fishes, because little is known about these effects (but see Connon et al. 2009, 2011; McIntyre et al. 2008, 2012). There are some extant data for other related species and these likely can be use to gain insights on predator–prey interactions of Delta fishes. The contaminant load of the Delta is likely substantial and includes the following potentially bioactive agents: pesticides, pharmaceuticals and personal care products (e.g., psychoactive drugs and triclosan), estrogen disruptors, ammonia, metalloids such as selenium, and heavy metals such as mercury, copper, and aluminum. These contaminants are all present in concentrations that could affect fish behavior (Sloman and Wilcox 2006; Connon et al. 2011; Brooks et al 2012; Fong et al., submitted). Aluminum, in particular, affects physiological homeostasis as well as learning and behavioral performance in Atlantic Salmon (Salmo salar, Grassie et al. 2013). Similarly, Sandahl et al. (2007) demonstrates that copper concentrations commonly found in Delta waters can produce abnormal anti-predator behaviors in Coho Salmon (Oncorhynchus kisutch). Their video (http://pubs.acs.org/doi/suppl/10.1021/es062287r) shows control salmon ceasing movement and dropping to the bottom of the
and Cech 2004). All else being equal, foraging theory predicts that predators will choose prey that maximize their net energy gain (Grossman 2014), although this can be achieved through a variety of methods. It is likely that Delta predators prefer fish prey that are energy-rich (Hartman and Brandt 1995), easily handled and consumed (i.e., small, soft-rayed and fusiform) and naïve to invasive predators (Kuehne and Olden 2012; Carthey and Banks 2014). Several authors document the importance of behavioral naiveté to predation mortality by demonstrating that invasive Lake Trout and Northern Pike feed disproportionately on salmonids despite the apparently higher abundance of native catostomid prey (Johnson and Martinez 2000; Johnson et al. 2002; Lepak et al. 2012). Given that the majority of fish predators in the Delta are invasives (Table 1), native fishes in the Delta may suffer increased predation because a lack of shared evolutionary history could produce predator naiveté in prey species (reviewed by Carthy and Banks 2014). Nonetheless, it is also true that some invasive predators have

Figure 1  A schematic depicting the components of predation and factors that affect predation in the Delta. “Search and Encounter” refers to the activities involved in locating prey, “Pursuit and Attack” to drawing close to the prey and initiating capture, “Capture” to the physical capture of the prey, and “Handling” to the physical act of manipulating the prey for consumption. The factors in the black boxes represent factors in unmodified environments that affect the predation process. The factors in underlined gray italics represent anthropogenic environmental changes in the Delta that increase predation rates on fishes.

Components of the Predation Process

The act of predation may be broken into several component rates, including search and encounter, pursuit and attack, capture and handling, and consumption, and these components are affected by a variety of changes that have occurred in the Delta (Figure 1). In unmodified environments, these components are affected by factors such as prey abundance and availability, spatial and temporal overlap of predator and prey, habitat complexity, turbidity, behavior, physiology, and morphological adaptations that facilitate (predator) or inhibit (prey) the predation process. Although most fish predators are opportunistic feeders (Gerking 1994), differences in prey characteristics (e.g., morphology, behavior, and energy content) also affect prey choice (Moyle and Cech 2004). All else being equal, foraging theory predicts that predators will choose prey that maximize their net energy gain (Grossman 2014), although this can be achieved through a variety of methods. It is likely that Delta predators prefer fish prey that are energy-rich (Hartman and Brandt 1995), easily handled and consumed (i.e., small, soft-rayed and fusiform) and naïve to invasive predators (Kuehne and Olden 2012; Carthey and Banks 2014). Several authors document the importance of behavioral naiveté to predation mortality by demonstrating that invasive Lake Trout and Northern Pike feed disproportionately on salmonids despite the apparently higher abundance of native catostomid prey (Johnson and Martinez 2000; Johnson et al. 2002; Lepak et al. 2012). Given that the majority of fish predators in the Delta are invasives (Table 1), native fishes in the Delta may suffer increased predation because a lack of shared evolutionary history could produce predator naiveté in prey species (reviewed by Carthy and Banks 2014). Nonetheless, it is also true that some invasive predators have
Table 1  Native and invasive fishes in the Sacramento–San Joaquin Delta system. Uncommon species are included, although primary marine and estuarine fishes that occasionally enter the Delta are not. Piscivores are marked in bold and the names of predators that may prey on fishes for some portion of their lifespan or when periodically available are underlined.

<table>
<thead>
<tr>
<th>Native species common name</th>
<th>Scientific name</th>
<th>Invasive species common name</th>
<th>Scientific name</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chinook Salmon</td>
<td><em>Oncorhynchus tsawytsha</em></td>
<td>American Shad</td>
<td><em>Alosa sapidissima</em></td>
</tr>
<tr>
<td>Delta Smelt</td>
<td></td>
<td>Bighorn Bullhead</td>
<td><em>Percina macrocephalus</em></td>
</tr>
<tr>
<td>Green Sturgeon</td>
<td><em>Acipenser medirostris</em></td>
<td>Black Bullhead</td>
<td><em>Ameiurus melas</em></td>
</tr>
<tr>
<td>Hardhead</td>
<td><em>Mylopharodon conopephalus</em></td>
<td>Black Crappie</td>
<td><em>Pomoxis nigromaculatus</em></td>
</tr>
<tr>
<td>Longfin Smelt</td>
<td><em>Spirinchus thaleichthys</em></td>
<td>Blue Catfish</td>
<td><em>Ictalurus furcatus</em></td>
</tr>
<tr>
<td>Pacific Lamprey</td>
<td><em>Entosphenus tridentatus</em></td>
<td>Bluegill</td>
<td><em>Lepomis macrochirus</em></td>
</tr>
<tr>
<td>Prickly Sculpin</td>
<td><em>Cottus asper</em></td>
<td>Brown Bullhead</td>
<td><em>Ameiurus nebulosus</em></td>
</tr>
<tr>
<td>River Lamprey</td>
<td><em>Lampetra ayresi</em></td>
<td>Common Carp</td>
<td><em>Cyprinus carpio</em></td>
</tr>
<tr>
<td>Steelhead</td>
<td><em>Onchorhyncus mykiss</em></td>
<td>Channel Catfish</td>
<td><em>Ictalurus punctatus</em></td>
</tr>
<tr>
<td>Sacramento Hitch</td>
<td><em>Lavinia exilicauda</em></td>
<td>Fathead Minnow</td>
<td><em>Pimephales promelas</em></td>
</tr>
<tr>
<td>Sacramento Splittail</td>
<td><em>Pogonichthys macrolepidotus</em></td>
<td>Green Sunfish</td>
<td><em>Lepomis cyanellus</em></td>
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<tr>
<td>Sacramento Pikeminnow</td>
<td><em>Ptychocheilus grandis</em></td>
<td>Goldfish</td>
<td><em>Carassius aurata</em></td>
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<tr>
<td>Sacramento Blackfish</td>
<td><em>Orthodon microlepidotus</em></td>
<td>Golden Shiner</td>
<td><em>Notemognus chrysoleucus</em></td>
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<tr>
<td>Sacramento Perch³</td>
<td><em>Archoplites interruptus</em></td>
<td>Largemouth Bass</td>
<td><em>Micropterus salmoides</em></td>
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<tr>
<td>Threespine Stickleback</td>
<td><em>Gasterosteus aculeatus</em></td>
<td>Mississippi Silverside</td>
<td><em>Menidia beryllina</em></td>
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<tr>
<td>Tule Perch</td>
<td><em>Hysterocarpus traskii</em></td>
<td>Rainwater Killifish</td>
<td><em>Lucania parva</em></td>
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<tr>
<td>White Sturgeon</td>
<td><em>Acipenser transmontanus</em></td>
<td>Red Shiner</td>
<td><em>Cyprinella lutrensis</em></td>
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<td></td>
<td></td>
<td>Redear Sunfish</td>
<td><em>Lepomis microlophus</em></td>
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<td></td>
<td>Redeye Bass</td>
<td><em>Micropterus coosae</em></td>
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<td></td>
<td></td>
<td>Smallmouth Bass</td>
<td><em>Micropterus dolomieu</em></td>
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<td></td>
<td></td>
<td>Shimofuri Goby</td>
<td><em>Tridentiger bifasciatus</em></td>
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<td></td>
<td></td>
<td>Shokihaze Goby</td>
<td><em>Tridentiger barbatus</em></td>
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<td></td>
<td></td>
<td>Spotted Bass</td>
<td><em>Micropterus punctulatus</em></td>
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<td></td>
<td></td>
<td>Striped Bass</td>
<td><em>Morone saxatilis</em></td>
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<td></td>
<td></td>
<td>Threadfin Shad</td>
<td><em>Dorosoma petense</em></td>
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<td></td>
<td></td>
<td>Wakasagi</td>
<td><em>Hypomesus nipponensis</em></td>
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<tr>
<td></td>
<td></td>
<td>Warmouth</td>
<td><em>Lepomis gulosus</em></td>
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<td></td>
<td></td>
<td>Western Mosquitofish</td>
<td><em>Gambusia affinis</em></td>
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<tr>
<td></td>
<td></td>
<td>White Catfish</td>
<td><em>Ameiurus catus</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td>White Crappie</td>
<td><em>Pomoxis annularis</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Yellowfin Goby</td>
<td><em>Acanthogobius flavimanus</em></td>
</tr>
</tbody>
</table>

a. Effectively extinct from the Delta.
been established in the Delta for over 100 years (e.g., Striped Bass) and it is possible that prey species have had sufficient time to develop behavioral adaptations to these predators.

The effects of both contaminants and invasive species may be magnified by environmental changes that have occurred in the Delta over the last 100 years including: (1) species invasions that alter physical habitat structure, (2) alterations of hydrologic regimes, temperature regimes and turbidity levels, (3) wetland loss, and (4) anthropogenic changes in physical structure (levees, canals, and abstraction facilities), coupled with changes in climate (Cloern and Jassby 2012; Dettinger et al., submitted). Predation on Delta fishes also is affected by ecosystem-level effects of invasives, such as the shift in the food web produced by the non-predatory overbite clam (*Corbula amurensis*, Cloern and Jassby 2012, Brown et al., forthcoming) or changes in the structural complexity of littoral habitats produced by invasive submerged aquatic macrophytes such as *Egeria* (Brown et al. 2016, forthcoming; Conrad et al. 2016). The increase in littoral structural complexity favors invasive species that currently inhabit the Delta, although it may also provide refuge for some prey species. Figure 2 illustrates the complexity of predator-fish prey interactions as potentially affected by invasions from introduced macrophytes where increases in vegetation have different effects on prey and predator, as well as different effects on roving predators (i.e., Striped Bass) versus ambush predators (i.e., Largemouth Bass).

The study of predator–prey dynamics in fishes is complicated by the fact that almost all fishes display ontogenetic dietary niches. The majority of fishes are micro-carnivores when young and then may transition through the macro-invertivore and finally to the piscivore stage. In addition, even herbivores may be sufficiently opportunistic to take advantage of periodically abundant high-energy prey, such as fish eggs or fry, and the population-level effects of this predation are largely unknown. If a predator is piscivorous, however, its effect on prey may be sublethal or indirect. For example, introduced Largemouth Bass almost extirpated cyprinids from Wisconsin lakes, by forcing cyprinids to shift from high-quality littoral areas to low-quality pelagic microhabitats with high temperatures, low food abundance, and abundant avian predators (Carpenter and Kitchell 1993). In other cases, behavioral avoidance of multiple predators may produce multiplicative effects, where the effect of one predator enhances the efficiency of a second predator (Hixon and Carr 1997). An hypothetical but not unlikely example for the Delta involves Striped Bass preying on native species such as hitch (*Lavinia exilicauda*) or Sacramento Blackfish (*Orthodon microlepidotus*) in open water habitats. To avoid

**Figure 2** Figure 2 illustrates the potential complexity of how predators and prey respond behaviorally to environmental changes (e.g., submerged aquatic vegetation, SAV) in the Delta. (A) illustrates potential effects on ambush (Largemouth Bass) and roving (Striped Bass) predators and their ability to capture prey. (B) displays the potential effects of how prey respond to ambush and roving predators and their mortality rate. Components in gray italics represent processes that will decrease predation in the presence of SAV, whereas those in black will increase predation in the presence of SAV.
Striped Bass, an individual of these species might find refuge in submerged aquatic vegetation (SAV), the preferred habitat of centrarchids and ictalurids, and then be eaten by one of these invasives.

Estimating the effects of predators on prey populations generally requires quantification of functional and numerical responses of predators to prey. The functional response describes the per-capita feeding rate of predators, usually as a function of prey density. An example of a functional response would be measurement of how the per capita predation rate of striped Bass on Chinook Salmon changed as the abundance of smolts changed during the outmigration. Most predatory fishes display a Type III functional response, which is “s” shaped and allows for both learning and compensatory responses to predation. However, if a functional response becomes asymptotic as prey density increases (e.g., a Type II functional response) depensatory mortality and population instability may occur (Essington and Hansson 2004). The numerical response details how changes in prey abundance affect predator abundance. Using the previous example, if we quantified the numerical response for Striped Bass and Chinook Salmon smolts, we would determine how changes in smolt abundance affected the numerical abundance of Striped Bass. When per-capita predator feeding rates are affected by predator abundance (e.g., numerical response, intra- or inter-specific interference competition) or when only a fraction of prey are vulnerable to predators, then predation rates may be relatively insensitive to fluctuations in predator abundance (DeAngelis et al. 1975; Ahrens et al. 2012). For example, the abundance of Atlantic Cod (Gadus morhua) in the Baltic Sea is not strongly related to changes in their major prey juvenile herring (Clupea harengus, Essington and Hansson 2004) and vice versa. In addition, meaningful quantification of the functional and numerical responses for abundant predators on Central Valley Chinook Salmon populations is complicated by the fact that the energetic contribution of this prey to Delta predators is so low that predator abundance is more likely influenced by the abundances of alternate prey than by Chinook Salmon abundance.

The Context-Dependency of Predation

Predation is almost always context-dependent; meaning the effects of predators on both individual prey and their populations vary based on the spatial and temporal context of the interaction. For example, predator-prey relationships in the Delta likely differ substantially in drought versus high-flow years. Linking context-dependency to spatial heterogeneity in the Delta means simply that the predation rate of Least Terns on Delta Smelt in the Sacramento River channel will likely differ from the predation rate of the same predator and prey at the same abundances in Franks Tract. An additional aspect of spatial context-dependency in the Delta involves the presence of predation hot spots where physical conditions combine to make predation much more likely than in unaltered habitats. Grossman et al. (2013) identified 13 predation hot-spots in the Delta based on data from the Bay-Delta Conservation Plan (BDCP 2013) and the California Fish and Wildlife Fish Predation Workshop (2013), (i.e., mortality data from tagged fish and flow alterations that likely would shift fish out of their normal migration path into low-quality habitat). Grossman et al. (2013) identified the following hot spots:

1. the junction of the Sacramento River and Sutter Slough,
2. Georgiana Slough,
3. Delta Cross Channel Canal,
4. Franks Tract,
5. Mildred Island,
6. Stockton Ship Channel,
7. Clifton Court Forebay,
8. Borden Highway Bridge,
9. head of Old River,
10. head of Old River Barrier (HORB),
11. Old River near Tracy,
12. pumping plant salvage release sites, and
13. the Red Bluff Diversion Dam.

In addition, a recent study (Sabal et al. 2016), demonstrated that the Woodbridge Irrigation
District Dam on the Mokelumne River also may be a predation hot spot, as well as the Tuolomne River below Don Pedro Dam (FISHBIO 2013).

Many other factors will induce variation into predator–prey relationships including: (1) the presence and type of shelter (e.g., SAV or woody debris) (Conrad 2016), (2) the ratio of prey size to predator size, (3) seasonal changes in abundance of the prey array, (4) defensive morphological (e.g., spines) or behavioral adaptations (Mittelbach and Persson 1998; Scharf et al. 2000), and (5) seasonal changes in habitat quality for prey, such as those produced by influxes of contaminants during winter–spring high flows or high water temperatures during summer and fall. Most predators are gape limited, which means that larger individuals and species are exposed to fewer predators than smaller individuals and species. Hence, environmental conditions that increase growth rates of prey such as favorable water temperatures, and increases in habitat and food quantity and quality, have a direct effect on predation rates. Predator foraging tactics also change depending on the array of prey available; hence, predator–prey relationships cannot be viewed as static entities.

The context dependency of predator–prey interactions combined with the high environmental, temporal, and spatial variability of the Delta environment undoubtedly will make quantifying the population-level effects of predation on fish populations very difficult.

**Predator Control Case Studies**

I will review the history of predator control studies because substantial concerns have been raised that predation by both native and invasive predators has contributed to declines in salmonid populations in the Sacramento–San Joaquin Delta (CDFW 2011; Grossman et al. 2013). This has led some members of the public to call for control measures on some predatory fishes, such as Striped Bass (Grossman et al. 2013, [www.youtube.com/watch?v=eEB3dnDmsBc](www.youtube.com/watch?v=eEB3dnDmsBc)). Control of invasive fishes is not an uncommon fishery management strategy (Kolar et al. 2010) but it has not had a high success rate. At present, several large-scale predator removal programs are underway in North America to aid in the recovery of native and sport fishes. The following case studies illustrate the breadth of approaches and their efficacy in controlling predatory fishes.

One of the most widespread and effective predator control programs has been directed at Sea Lamprey *Petromyzon marinus* in the Great Lakes (Smith and Tibbles 1980; Larson et al. 2003). Application of the highly selective lampricide TFM (3-trifluoromethyl-4-nitrophenol) in rearing streams has been effective at reducing Sea Lamprey populations by 90% in most areas (Adair and Sullivan 2013) at a cost of about $16 million per year (MDNR 2013). Lamprey control probably has been successful because lampreys have a vulnerable life-history stage (ammocoete) that occupies a restricted habitat (burrows in soft sediment in streams) which limits their ability to escape a toxin. Consequently, population reduction via toxicant application is logistically feasible and effective. However, because of the unusual characteristics of lampreys this success must be viewed as a special rather than a general case for predator control strategies.

The Northern Pikeminnow (*Ptychocheilus oregonensis*) Sport-Reward Program began in 1991 and is sponsored by Bonneville Power Administration. The program seeks to maintain a 10% to 20% exploitation rate on Northern Pikeminnow throughout the Columbia River by paying anglers $4 to $8 to harvest fish >228 mm TL (Porter 2010). The program removed over 2.2 million fish during 1998–2009 and is believed to have reduced predation on juvenile salmonids; nonetheless, concomitant increases in salmonid population abundance have been difficult to detect (Carey et al. 2012). The lack of positive population-level responses may be a result of compensatory responses by other predators such as Caspian terns (*Hydroprogne caspia*) and marine mammals such as California sea lions (*Zalophus californianus*, Carey et al. 2012). The possibility of compensatory responses by other predators illustrates a major shortcoming of predator control strategies; rarely is predation mortality a function of just one species. Although these programs are expensive—$14 million was paid out in bounties from 1998–2009 alone—angler reward systems may be more cost-effective than if agencies performed removals themselves (Carey et al. 2012).
Lake Trout have been widely introduced for sport fishing in western US lakes and reservoirs. In some systems, these fish threaten native and introduced salmonid populations (Dux et al. 2011). Commercial fishing and sport angling appear to have reduced Lake Trout abundance and allowed for kokanee recovery at Lake Pend Oreille, ID (Hansen et al. 2010). Angler incentives do not appear to have been effective at Flathead Lake, Montana (Federal Register 2012). Although commercial-scale netting has removed over 450,000 Lake Trout from Yellowstone Lake (Wyoming/Montana), this species continues to threaten native Cutthroat Trout (Syslo et al. 2011).

As part of the Upper Colorado River Endangered Fish Recovery Plan (USFWS 2012), invasive Channel Catfish (Ictalurus punctatus), Northern Pike (Esox lucius), Smallmouth Bass (Micropterus dolomieu) and other potential predators are being removed from critical habitat for ESA listed cyprinids and catostomids (Tyus and Sauders 2000; Johnson et al. 2008). Invasive fish control in the Upper Colorado River Basin costs over $1 million annually (Mueller 2005). Demonstrating native fish responses to removal of invasive fishes has been complicated by highly variable environmental conditions that differentially affect predators and prey. However, the available literature demonstrates that even in isolated reaches with intense removal efforts, there have been few demonstrable positive responses from native fishes (Bestgen et al. 2007; Skorupski et al. 2012).

The literature on predator removal as a management strategy for recovery of desirable species indicates there is little evidence for strong, positive, population-level responses in prey. Predator removal, even in relatively simple systems such as reservoirs and lakes, is a difficult and costly management technique, and the lack of success illustrates the challenges inherent in functionally eliminating wild fish populations in systems with complex dynamics. Eradication generally is unlikely, except in small, isolated systems where re-invasion can be prevented (Kolar et al. 2010), and even in these systems re-introductions may occur via disgruntled members of the public. Predator suppression may sometimes facilitate increases in abundances of prey fish populations, but this requires intensive and sustained efforts at removal (Beamesderfer 2000), which is expensive. In addition, because of the generalist nature of most fish predators, a decrease in the abundance of one predator may result in increases in the abundance of competing predators, or in the abundance of competitors of the prey species; the result being a net overall negative effect on the prey population of interest. Given the dynamic and highly modified environment of the Delta, if the goal is increasing population sizes of species of concern, such as Chinook Salmon or Delta Smelt, it will take careful study before any predator removal or suppression technique should be undertaken.

**PREDATION ON DELTA FISHES**

The assemblage of predatory fishes in the Delta is dominated by invasive species (Table 1) (Brown and Michniuk 2007; Nobriga and Feyrer 2007; NRC 2010; Cavallo et al. 2012; NRC 2012; Brown 2013, unreferenced, see “Notes”), although most non-piscine predators are native species. General survey data provide some estimates of abundance or relative abundance (catch-per-unit-effort) data for some predators and prey fishes at certain times and locations; however, quantitative abundance estimates for many predators and prey frequently are lacking. Nonetheless, numerous predators such as Striped Bass, Largemouth Bass, ictalurids, Mississippi Silversides, some centrarchids, and birds (e.g., ducks, herons, egrets, and terns) appear to be common in the Delta (Nobriga et al. 2002; Dege and Brown 2004; Brown and May 2006; CDFW 2013).

**Predation Studies on Delta Fishes**

A variety of approaches have been used to elucidate relationships among Delta predators and prey. The vast majority of studies are dietary investigations whose main purpose was to quantify the diets of game fishes (e.g., Thomas 1967; Stevens 1966; Bryant and Arnold 2007; Nobriga and Feyrer 2007), although several more recent studies directly address consumption of prey by predatory fishes (Lindley and Mohr 2003; Cavallo et al. 2012; Loboschefsky et al. 2012; FISHBIO 2013; Nobriga et al. 2013). The landmark dietary studies in Turner and Kelly (1966) provide important information for both game and several non-game species. Nonetheless, the design of
most dietary studies limits their use for quantifying how predators affect prey fish populations (see below). A majority of the strongest studies (Lindley and Mohr 2003; Loboschefsky et al. 2012; Nobriga et al. 2013) involve mathematical modeling that uses empirical data, which provide insights into the potential dynamics of Striped Bass predation on Delta fishes. However, the modeling studies still possesses shortcomings that limit their ability to predict future dynamics including: (1) lack of independent empirical verification of the dynamics modeled and predicted outcomes, (2) lack of empirical verification of model assumptions, and (3) limited empirical databases (e.g., limited temporal and spatial data as well as a lack of resolution in some data; e.g., prey categories such as “fish” rather than prey species). Nonetheless, all studies possess shortcomings, and in the absence of complete data these models serve as a starting point for management decisions and as hypothesis-generating tools for further studies.

Several recent studies address the predation issue directly, most notably Cavallo et al. (2012), Sabal et al. (2016) and Demetras et al. (2016). In an ambitious study, Cavallo et al. (2012), attempted a BACI (Before, After, Control, Intervention) assessment of how predatory fishes affected Chinook salmon juvenile survival, using boat electrofishing to remove predatory fishes from a section of the Mokelumne River. As might be expected with field experiments, the predator removal experiment was relatively short-term (slightly over 2 weeks long). Nonetheless, Chinook Salmon displayed higher survivorship in the removal section than in the unshocked control section after the first predator removal; however, the second removal did not result in increased Chinook Salmon survivorship. Unfortunately, predictor densities in the treatment section tripled after the first removal, which underscores the importance of potential compensatory responses by other predatory species when a single predator species is removed. This phenomenon is the biggest weakness of the “predator removal” management approach (e.g., Striped Bass predation limits Chinook Salmon populations), because in the Delta there are literally 10 to 20 other predatory species that could potentially increase in abundance (and prey consumption) if a dominant predator is removed. Indeed, this may have occurred when the Northern Pikeminnnow removal program was put into practice, given that both Caspian tern and sea lion predation apparently increased concomitantly with the removal program (Carey et al. 2012). Nonetheless, the finding of increased Chinook Salmon survivorship in the first removal of Cavallo et al. (2012) may be conservative, because fishes in the control section were not shocked and, therefore, likely in better condition.

Very recent published work by Sabal et al. (2016) and Demetras et al. (2016) indicate that Striped Bass predation upon juvenile Chinook Salmon could be substantial, although the amount of predation experienced by wild as opposed to hatchery fish—or those in habitats other than predation hot spots—remains unknown. In a multi-faceted study, Sabal et al. (2016) demonstrated that Striped Bass aggregated below the Woodbridge Irrigation District Dam on the Mokelumne River (a predation hot spot) and that Chinook Salmon survivorship increased when Striped Bass were removed. The Woodbridge Dam has a fish ladder, but Striped Bass do not typically ascend fish ladders; rather, they typically aggregate below dams and fish ladders where they prey extensively on smaller fishes (Davis et al. 2012). In predation hot spots in the Delta, predators may aggregate and consume large numbers of prey that are disoriented by: (1) unnatural or reversed flow patterns, and (2) modified habitat structures such as fish ladders, water abstraction devices, modified channels, or impoundments (Davis et al. 2012; Grossman et al. 2013). Demetras et al. (2016) developed an innovative predator detection system that documented predation by Striped Bass and unidentified fish predators on tethered Chinook Salmon, and found a positive relationship between current velocity and predation rate and an inverse relationship between water column depth and predation rate. Although this highly innovative system documented predation by Striped Bass and other predators on tethered Chinook Salmon, it is difficult to extrapolate these results to estimates of predation rates throughout the Delta.

Much of the data for my analysis came via the courtesy of researchers who provided data from their unpublished studies (e.g., Barry 2015; Brandl et al. 2015; Smith et al. 2015; Young 2015; Zeug 2015, all unreferenced, see “Notes”); and these data and the work of Baerwald et al. (2013) illustrate the advantages of DNA analysis for dietary analyses.
Analyses using DNA typing allow investigators to analyze large numbers of stomachs with a high level of precision of prey identification. Unfortunately at present, they cannot be used to quantify the intensity of prey consumption, but only whether or not a given prey species has been consumed.

The Data and Analysis

I searched the literature and contacted researchers actively working on dietary or predator–prey studies on Delta fishes and compiled a matrix of predator species and their piscine prey. Sources are listed in Table 2. Based on the frequency of prey occurrence in predator gut contents, I ranked predator consumption as occasional, moderate, or common (Table 2). I used these data to evaluate the frequency of predation on native and invasive species and their hypothesized effect on prey populations (Table 3). I also evaluated the data base of dietary and predation papers and ranked data for each species as: none, fair, good, or excellent (Table 2). In general, the data base of dietary/predation studies is fair but very weak for mammals, many carnivorous birds and some reptiles and amphibians (Table 2). In addition, the data base is biased towards highly valued game species (e.g., Striped and Largemouth Bass) and the level of prey resolution varies according to the authors’ purposes (in many studies, broad prey categories were used e.g., “fish”). I recorded 32 different predator categories, although categories such as gulls, herons and egrets, and terrestrial garter snakes contained multiple predatory species (Table 2). I recorded 36 different prey species and five multi-species categories, including unidentified fishes, unidentified salmonids, unidentified shad, Lepomis species, and Tridentiger species. The data indicated that most predators only were occasional consumers of individual prey species, although moderate and common consumption also were observed, for example,

1. Moderate: Sacramento Pikeminnow consuming Longfin Smelt, Striped Bass consuming Sacramento Splittail, Largemouth Bass consuming Prickly Sculpin; and

2. Common: Striped Bass consuming Chinook Salmon, Largemouth Bass consuming Sacramento Pikeminnow, Channel Catfish consuming Largemouth Bass (see also Table 2).

My analysis yielded few generalizations regarding predator–prey interactions for Delta fishes other than the observation that most predators were unspecialized and consumed a wide variety of both native and invasive fishes. The sole exception was Prickly Sculpin which, when it consumed fish, preyed upon mostly natives (Table 2; Merz 2002b). Both Striped and Largemouth Bass exhibited broad dietary breadths, with Striped and Largemouth Bass preying upon 32 and 28 categories of prey, respectively (Table 2). These predators preyed upon a wide variety of both native and invasive fishes, and, as important game fish, had the broadest temporal and spatial coverage in dietary data (Table 2). The Sacramento Pikeminnow, a native predator, also displayed broad dietary breadth, with 14 different categories of fish prey eaten. These data reinforce the general opinion of the literature that most fish predators are generalists that consume many different prey types (Moyle and Cech 2004).

Most predators fed primarily on invasive species, which are the most abundant fishes in the Delta (Brown and Michniuk 2007). Given the generalist nature of vertebrate predators, this likely represents consumption of prey in proportion to their abundance. In addition, it is likely that some predators, such as snakes, fed primarily on invasive species because both predator and prey are found in modified habitats where invasives dominate (B. Halstead, pers. comm., to me, 2015, unreferenced, see “Notes”). The effects of avian, reptilian, amphibian, and mammalian predation on fishes in the Delta are not well known, although Caspian terns can consume high numbers of stocked Chinook Salmon (Evans et al. 2011; Andrean et al. 2012; Hostetter et al. 2015).

Several additional impediments limited my ability to reach strong conclusions about the effects of predators on the abundance of prey species. First, dietary data lacked resolution to the family or species level because predation on fish prey frequently was listed merely as “unidentified fishes.” Second, there was a lack of data on direct predation as well as a lack of dietary data that included prey availability so that prey selection could be determined (also see Grossman et al. 2013). Third, although we know fish were consumed, we do not know if they were healthy, stressed, or even dead.
Table 2  Predator–prey interactions among Delta animals. Prey consumption by predators is ranked as O–occasional <5% by frequency of occurrence, M–moderate >5% to 49%, C–Common >50%. A question mark indicates that the predator is known to be piscivorous but no dietary data are available for the Delta.

|          | SACRAMENTO SMOLT  | SACRAMENTO SPIRIT  | STEELHEAD  | CHERNOCK SALMON | PIKLY SCULPIN | STRIPED BASS | LARGEMOUTH BASS | SMALLMOUTH BASS | WARMOUTH | BLUEGILL | WHITE CRAPPIE | WHITE CATFISH | CHUMMED CATFISH | BROWN BULLHEADS | BLACK BULLHEADS | MISSISSIPPI SILVERDOES | COMMON CARP | YELLOWFIN GOLDFISH | SHIMOFURI GOLDFISH | CACAPAN TENN | CALIFORNIA LANTERN FISH | DOUBLE-CREATURED CONSUMERS | HERONS & EGRETS | GOLDS | GIANT GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC GARTER SNAKE | AQUATIC S
Table 2 (Continued) Predator-prey interactions among Delta animals. Prey consumption by predators is ranked as O–occasional <5% by frequency of occurrence, M–moderate >5% to 49%, C–Common >50%. A question mark indicates that the predator is known to be piscivorous but no dietary data are available for the Delta.

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Table 3 Hypothesized effects of predators on Delta fish populations based on Table 2. If a citation is not given, the entry is based on the author's personal observation. A "U" in a category means that only unidentified fishes were listed as fish prey. Scientific names listed only when not previously mentioned.

<table>
<thead>
<tr>
<th>Species</th>
<th>Abundance (since 1980)</th>
<th>% Native fish in diet (frequency of occurrence)</th>
<th>% Invasive fish in diet (frequency of occurrence)</th>
<th>Hypothesized predator effect on prey population</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Native predators</strong></td>
<td></td>
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<tr>
<td>Sacramento Pikeminnow</td>
<td>Likely decreasing(^a)</td>
<td>30</td>
<td>70</td>
<td>Unlikely, except in predation hot spots</td>
</tr>
<tr>
<td>Thicktail Chub, <em>Gila crassicauda</em></td>
<td>Extinct</td>
<td>None</td>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>Sacramento Perch</td>
<td>Functionally extinct</td>
<td>None</td>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>Sacramento Splittail</td>
<td>Common</td>
<td>U</td>
<td>U</td>
<td>Unlikely</td>
</tr>
<tr>
<td>Steelhead</td>
<td>Declining</td>
<td>U</td>
<td>U</td>
<td>Unlikely</td>
</tr>
<tr>
<td>Chinook Salmon</td>
<td>Declining</td>
<td>100</td>
<td>0</td>
<td>Unlikely</td>
</tr>
<tr>
<td>Prickly Sculpin</td>
<td>Common</td>
<td>75</td>
<td>25</td>
<td>Unlikely, low % of fish in diet</td>
</tr>
<tr>
<td>Caspian tern</td>
<td>Common(^b)</td>
<td>100</td>
<td>Unknown</td>
<td>Possibly on stocked salmonids</td>
</tr>
<tr>
<td>River otter</td>
<td>Increasing(^d)</td>
<td>Piscivorous</td>
<td>Piscivorous</td>
<td>Possibly in isolated habitats or hot spots</td>
</tr>
<tr>
<td>Giant garter snake, <em>Thamnophis gigas</em></td>
<td>Threatened</td>
<td>Largely piscivorous</td>
<td>Largely piscivorous</td>
<td>Unlikely to be significant</td>
</tr>
<tr>
<td>Herons and egrets, <em>Ardea and Nycticorax spp.</em></td>
<td>Common</td>
<td>Largely piscivorous</td>
<td>Largely piscivorous</td>
<td>Unlikely to be significant</td>
</tr>
<tr>
<td>Mergansers, <em>Mergus spp.</em></td>
<td>Uncommon</td>
<td>U</td>
<td>U</td>
<td>Possibly in isolated habitats or hot spots</td>
</tr>
<tr>
<td>Double-crested cormorant, <em>Phalacrocorax auritus</em></td>
<td>Common and increasing(^e)</td>
<td>Largely piscivorous</td>
<td>Largely piscivorous</td>
<td>Possibly in isolated habitats or hot spots</td>
</tr>
<tr>
<td>Western grebe, <em>Aechmophorus occidentalis</em></td>
<td>Common</td>
<td>Largely piscivorous</td>
<td>Largely piscivorous</td>
<td>Possibly in isolated habitats or hot spots</td>
</tr>
<tr>
<td>Western pond turtles, <em>Actinemys marmorata</em></td>
<td>Common</td>
<td>U</td>
<td>U</td>
<td>?</td>
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<tr>
<td>California least tern, <em>Sternula antillarum browni</em></td>
<td>Increasing</td>
<td>100</td>
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<tr>
<td>Gulls</td>
<td>?</td>
<td>U</td>
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<tr>
<td>Aquatic garter snake, <em>Thamnophis atratus</em></td>
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<tr>
<td>Common garter snake, <em>Thamnophis sirtalis</em></td>
<td>?</td>
<td>0</td>
<td>100</td>
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<tr>
<td><strong>Invasive predators</strong></td>
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<tr>
<td>Largemouth Bass</td>
<td>Increasing(^f)</td>
<td>33</td>
<td>67</td>
<td>Possibly</td>
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<tr>
<td>Striped Bass</td>
<td>((^<em>)age-0 in pelagic surveys, or stable for sub-adults, (^</em>) for adults)(^g)</td>
<td>47</td>
<td>53</td>
<td>Possibly</td>
</tr>
<tr>
<td>Smallmouth Bass</td>
<td>?</td>
<td>6</td>
<td>33</td>
<td>?</td>
</tr>
<tr>
<td>Mississippi Silverside</td>
<td>Abundant</td>
<td>50</td>
<td>50</td>
<td>Possibly for eggs, larvae, juveniles</td>
</tr>
<tr>
<td>White Catfish</td>
<td>Common</td>
<td>40</td>
<td>60</td>
<td>?</td>
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<tr>
<td>Channel Catfish</td>
<td>Common</td>
<td>50</td>
<td>50</td>
<td>?</td>
</tr>
<tr>
<td>Warmouth</td>
<td>Common(^h)</td>
<td>0</td>
<td>100</td>
<td>?</td>
</tr>
<tr>
<td>Black crappie</td>
<td>Common(^h)</td>
<td>40</td>
<td>60</td>
<td>?</td>
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<tr>
<td>Bluegill</td>
<td>Common</td>
<td>U</td>
<td>U</td>
<td>?</td>
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<tr>
<td>Brown bullheads</td>
<td>Common</td>
<td>U</td>
<td>U</td>
<td>?</td>
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<tr>
<td>Black Bullheads</td>
<td>Common</td>
<td>U</td>
<td>U</td>
<td>?</td>
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<tr>
<td>Common Carp</td>
<td>Common</td>
<td>U</td>
<td>U</td>
<td>?</td>
</tr>
<tr>
<td>Shimofuri Goby</td>
<td>Common</td>
<td>0</td>
<td>100</td>
<td>?</td>
</tr>
<tr>
<td>Bullfrog <em>Lithobates catesbiana</em></td>
<td>Common</td>
<td>Piscivorous</td>
<td>Piscivorous</td>
<td>Possibly in isolated habitats or hot spots</td>
</tr>
</tbody>
</table>

\(^a\) FISHBIO 2013; Nobriga and Feyrer 2007.  
\(^b\) Evans et al. 2011; Andrean et al 2012.  
\(^c\) Only dietary data from recovered tags of stocked salmonids.  
\(^d\) P. Moyle, pers. comm., 2015, un referenced, see “Notes.”  
\(^e\) Adkins et al. 2014.  
\(^h\) Turner and Kelly 1966.
The high numbers of Chinook Salmon and the occasional Steelhead eaten by Channel Catfish suggest that these prey may have been stressed or dead when consumed, especially because genetic analysis rather than visual inspection determined their frequency of consumption (Smith et al. 2015). Fourth, my conclusions were limited by the fact that frequency-of-occurrence data are not necessarily correlated with predation intensity. For example, a frequency of occurrence of 100% is obtained when each individual of a predatory species consumes just one individual of a given prey species. However, the same value is obtained if each predator consumes 25 individuals of a given species. Consequently, Tables 2 and 3 are rudimentary estimates of predator–prey interactions for Delta fishes. Fifth, many of the dietary studies were limited in their spatial and temporal coverage, which may result in over-estimation of the intensity of predator–prey interactions if data were collected during a period when the predator fed intensively on a prey that only was temporarily abundant. Conversely, the data may under-estimate these relationships if samples were lacking for multiple days within a season, or seasons within a single year, or multiple years. Sixth, it is well known that even non-predatory fishes will consume both fish eggs and larvae, yet these resources typically are only available for weeks or a month, and are frequently missed in dietary studies. It is possible that some species such as Mississippi Silversides may prey significantly on eggs or larvae of species such as Delta Smelt (Baerwald et al. 2012). Finally, recent DNA analyses of stomach contents of Delta fish predators may allow a species’ diet to be quantified accurately and quickly (Baerwald et al. 2012; Brandl et al. 2015, unreferenced, see “Notes”; Smith et al. 2015), but cannot quantify the energetic importance of a given prey to the predator, and, concomitantly, the number of prey eaten by an individual predator.

Although it is beyond the scope of this study, Grossman et al. (2013, unreferenced, see “Notes”) provide useful examples of modeling approaches to predator–prey dynamics, Loboschensky et al. (2013) of bioenergetic modeling, and Cavallo et al. (2012) and Ferrari et al. (2014) of direct experimentation. Not surprisingly, there is a trade-off between logistical ease and realism, with indirect approaches such as population or bioenergetic modeling producing results that may be consistent with predator limitation, but typically cannot exclude alternative agents of mortality. Hence, their conclusions must be considered correlative rather than causal. Conversely, predator removal experiments are logistically difficult and may have limited realism (e.g. exclusions of pelagic roving predators), but may directly address whether predators depress prey populations on a local scale (Cavallo et al. 2012; Grossman et al. 2013). It will take a combination of methods that focus on direct experimentation, population modeling, and dietary and behavioral studies to yield further insights into the effects of predation on Delta fishes. Given the difficulty of separating ultimate from proximate causes of mortality, and the widespread distribution of bioactive contaminants on fishes throughout the Delta (Fong et al., submitted), it will be very difficult to untangle the ultimate from proximate causes of predation mortality for prey fish populations.

CONCLUSIONS

An understanding of any important ecological system requires extensive study of the abiotic and biotic relationships driving the system. The Sacramento–San Joaquin Delta supports a biological stew of native and invasive predators and prey whose ingredients are likely to change over time as new invasions occur and climate change progresses (Dettinger et al., submitted; Wiens et al., this volume). Unfortunately this means the biological conclusions drawn from the Delta of today may not be relevant a decade from now (e.g., trophic web shift caused by overbite clams), especially as invasions continue. Nonetheless, the invasive and the few remaining native fish predators in the Delta are generalists that likely consume whichever fish prey are locally available. Given that invasives are the most abundant fish species in the Delta, predators appear to prey most heavily on these species—as expected of generalist predators. However, the predator–fish prey database
is not strong, and I would urge future efforts to undertake long-term dietary studies combined with attempts to establish predator preferences, rather than just consumption. This will be particularly important given the highly fluctuating environmental conditions typical of the Delta and the likely changes that global climate change will produce, including: (1) increases in sea-level; (2) shifts in the variability, timing, and amount of precipitation; and (3) increased temperatures, which may facilitate future species invasions (Moyle et al. 2013; Dettinger et al., submitted). If the Delta experiences more drought and increased water exports, fishes may be concentrated in small patches of suitable habitat, which could alter predator-prey relationships and render prey species more susceptible to predation not only by fishes but also by birds, reptiles, and mammals. Although the data base is not extensive, it is clear that predation on fishes in the Delta is a common phenomenon that warrants further investigation. Hopefully future investigations will elucidate the effects of this phenomenon on both native and invasive fishes in the Delta.

Finally, it has recently been proposed that Striped Bass populations be significantly reduced to facilitate recovery of endangered Central Valley Chinook Salmon (http://www.youtube.com/watch?v=eEB3dnDmsBc). Although from a scientific standpoint it is always interesting to see how removal of an abundant species affects community structure and ecosystem processes, unless a removal study is conducted under rigorous experimental conditions (controls, monitoring of other predators, demonstration of direct predation and predator per-capita effects), it is unlikely to yield conclusive information on the relationship between predation and Chinook Salmon survival. As mentioned previously, the most likely outcome of Striped Bass removal is that a competing predator will increase in abundance and there will be little reduction in predation mortality for Chinook Salmon. It is likely that the most productive management strategy for decreasing predation on Chinook Salmon and other Delta fishes is to restore natural habitat and flows, especially in predation hot spots.

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NOTES


Moyle PB. Peter Moyle, University of California, Davis. In-person conversation with G. Grossman on 2015 May 14 about otter populations in the Delta.


Young M. 2015. Matthew Young, University of California, Davis. Unpublished data sent to G. Grossman via email on 2015 June 08 about predation on Delta fishes.

ABSTRACT

What happens at one place in a landscape influences and is influenced by what happens in other places. Consequently, management and restoration that focus on individual places may fail to recognize and incorporate interactions across entire landscapes. The science of landscape ecology, which emphasizes the interplay of landscape structure, function, and change at multiple scales, offers a perspective that can integrate the spatial relationships of ecological processes and the functional interconnections of land and water in the Delta.

Although the Delta is one of the most studied estuaries in the world, applications of landscape science have been limited. We describe why it is important to incorporate landscape science into management and restoration, emphasizing how Delta landscapes have changed over the past centuries. The land–water linkages of the past have been broken, waterways have been over-connected, and hard boundaries have replaced the gradual and dynamic transitions among landscape patches.

The contemporary landscape also has new, novel assemblages of species and stressors that were not there in the past. This historical perspective indicates how knowledge of past landscape functions can contribute to the restoration and management of contemporary landscapes. We illustrate these points with case studies of inundation dynamics and riparian woodlands, and use a third example to describe a landscape approach to restoration.

We propose that science that encompasses the multiple, interacting components of functional landscapes in the Delta will foster resilient and enduring restoration and management outcomes that benefit both people and wildlife. We suggest several ways of moving landscape science to the forefront of management and restoration in the Delta.

KEY WORDS

McCormack–Williamson Tract, historical ecology, inundation, landscape, restoration, riparian woodland, scale, waterscape

INTRODUCTION

The Delta tells many stories. There are stories told by indigenous people for thousands of years; stories of families that have farmed the Delta for generations; stories of the shifting balance between native and introduced species; stories of floods and droughts; and stories of astounding engineering
accomplishments. Here we consider stories of Delta landscapes: what they are, how they have changed, and how the science of landscapes can enrich management and restoration of the Delta and its resources.

Scientists who study landscapes—landscape ecologists—are concerned with how spatial heterogeneity affects ecological systems. The composition, configuration, and arrangement of patches in a landscape mosaic influence the movement and distribution of water, sediment, nutrients, organisms, people, and much else over the landscape as a whole (Turner 1989; Wiens et al. 1993; Fahrig 2007). Landscape heterogeneity affects the persistence of populations (Fraterrigo et al. 2009), species interactions (Polis et al. 2004), and ecosystem function (Lovett et al. 2005)—in short, just about everything that happens in the environment.

Viewing the Delta through the lens of landscapes is important for several reasons. First, landscapes are where people and the Delta intersect, where people raise crops and families, and experience the environment and nature. Thinking about “the Delta as an evolving place” (Delta Reform Act of 2009; California Water Code §85054) requires thinking about where people are in a landscape and what they are doing there.

Second, landscapes in the Delta include the water, the transitions across wetlands and levees, and the multiple uses of the lands behind the levees. A landscape perspective expands the focus from the fish, flows, and water that dominate science and drive debates to consider how the lands and waters interact in the Delta.

Third, landscapes provide a feasible middle ground, somewhere between the individual places where restoration or management are undertaken and the overwhelming complexity of the entire Delta. The focus of planning and management, for example, can expand from protecting or restoring individual habitat areas for native species to a more comprehensive approach that encompasses multiple species and functioning ecosystems (Opdam et al. 2002; NAS 2016).

Fourth, by considering entire landscapes rather than individual places, the broad-scale patterns and processes that determine how Delta ecosystems function emerge. An integrated landscape perspective can provide the foundation, for example, for managing or restoring ecological connectivity, habitat diversity, landscape adaptability, and resilience to change—all critically important factors in a Delta faced with climate change and sea-level rise (Cloern et al. 2011; Dettinger et al., submitted).

These points speak to the value of going beyond site-specific plans and actions to manage larger functional and interconnected landscapes, as called for in the Delta Plan (DSC 2013). Here, we explore how current thinking in landscape ecology can foster the development of a stronger, more cohesive approach to restoration and management in the Delta. We emphasize studies that have documented past landscapes in the Delta and how they have changed (Whipple et al 2012; SFEI–ASC 2014), not to establish historical targets for landscape management or restoration (see Wiens and Hobbs 2015), but because these studies show how changes to the landscape have led to the loss of desired functions and suggest opportunities to regain some of that functionality.

**PERSPECTIVES ON LANDSCAPES**

Landscapes can be viewed in multiple ways (Figure 1). A soil scientist or geographer may see the landscape as a mosaic of soil types or topographies. An engineer may see the same landscape as an interwoven network of roadways and levees. To a land-use planner or farmer, the landscape is a patchwork of ownerships, land uses, and agricultural crops. A wildlife biologist may focus on the distribution of habitats for a species of interest. A conservationist may view the landscape in terms of protected areas or occurrences of invasive species.

To a landscape ecologist, these perspectives are all characterized by their spatial heterogeneity—they are mosaics of relatively discrete “patches” such as the agricultural fields or riparian woodlands of Figure 1. Patches are delimited by boundaries of varying width and resistance to movement (permeability) and may be linked together in various ways (connectivity). This patch-boundary-connectivity-mosaic conceptualization of landscapes fits nicely with tools such as remote sensing, geographic information...
Figure 1  The landscape surrounding the McCormack–Williamson Tract, as viewed from a variety of perspectives. The details of each map are unimportant; the important point is that different perspectives lead to quite different perceptions of how the same landscape is structured.
systems, geospatial modeling, spatial statistics, percolation theory, and even fractal geometry. It has spawned a variety of landscape metrics and has nurtured much of the growth of landscape ecology as a science (Forman 1995; Wiens 1999).

Landscapes defined by spatial patterns in hydrology, land uses, vegetation, human infrastructure, or geology express how humans using different criteria perceive landscapes. The species that are often the focus of management, however, respond to the spatial heterogeneity of an area in different ways, based on different landscape features (Wiens et al. 1993; Mac Nally 2005; Fahrig et al. 2011). A barrier to a frog or salamander may be a corridor to a small mammal; a strip of riparian woodland that is too narrow to support a hawk may be just right for a songbird. When the objectives of management deal with species, how the species and we humans view the landscape must both be considered.

STRUCTURE, FUNCTION, AND CHANGE

These perspectives emphasize the spatial structure of landscapes. Landscape-level processes, such as movements of materials, nutrients, or organisms or the spread of disturbances are shaped by this landscape structure. For example, fish movements are influenced by the configuration of water channels, the spatial arrangement of feeding and spawning habitats, or the occurrence of predators (Bennett and Burau 2015). Movements of terrestrial species among habitats can be blocked by infrastructure or land uses that sever migration corridors or fragment habitat. Concentrations of contaminants such as selenium depend on where in a landscape selenium is deposited or released and how it is moved from place to place, affecting its uptake and accumulation in food webs (Luoma et al. 2008).

Landscape structure and processes are spatial properties of an area. But there is also a temporal dimension: landscapes change. The contemporary landscape of leveed islands, farms, and urban communities bears little resemblance to the historical mosaic of wetlands and riparian woodlands (Whipple et al. 2012). Biological communities and ecosystems have been dramatically altered, particularly by the spread of invasive, non-native species (Moyle 2013). In addition, land uses have changed in response to regional and global economics. As Delta ecosystems and landscapes have been engineered away from their natural states, their inherent resilience has been reduced, increasing their vulnerability to large disturbances (as suggested more generally by Walker and Salt 2006). This vulnerability will increase as the effects of accelerated sea-level rise and climate change emerge with full force in the Delta, bringing additional changes in the occurrences of species, agricultural practices, and the structure and composition of landscapes.

These three dimensions of landscapes—structure, function, and change—intersect (Figure 2). The spatial arrangement of landscape elements and the composition of a mosaic—its structure—affect how organisms are distributed and disperse. For example, the species composition of fish assemblages in the Delta may vary among sampling locations and seasons in ways related to the structural characteristics of the sampling sites (Nobriga et al. 2005). Landscape structure also determines how water, nutrients, and disturbances move through an

Figure 2 A Venn diagram showing the intersection of structure, function, and change that are the key elements of a scientific approach to the study and management of landscapes. The relationships among these elements change with changes in spatial or temporal scale.
area (Turner et al. 1989; Reiners and Driese 2004). These functions also affect structure. Movements of organisms alter patch composition, movements of nutrients affect water quality, floods reconfigure river channels, and droughts alter the vegetation or land uses in a landscape. This mélange of structure, function, and change in landscapes is why the Delta is such a complex and dynamic place, rife with “wicked problems” that challenge management or restoration (Luoma et al. 2015).

**SCALE**

The complexity of Delta ecosystems is exacerbated by scale. Components of the system operate at different scales in space and time. Delta Smelt (*Hypomesus transpacificus*) and Sacramento Splittail (*Pogonichthys macrolepidotus*) occupy only small parts of the Delta, whereas Chinook Salmon (*Oncorhynchus tshawytscha*) and sandhill cranes (*Grus canadensis*) cover large areas in migratory movements that extend well beyond the Delta. Each of these species responds to spatial and temporal variation in different features of the environment at different scales of resolution (Nobriga et al. 2005).

People do this, too. The composition of crops in an agricultural landscape is determined by farmers at a local scale, but their decisions are influenced by policies and economics at national and international scales. Management of natural resources is usually implemented at scales of hectares to a few square kilometers—scales that humans find familiar and manageable. Problems arise, however, when the scale(s) of management actions do not match the scale(s) of the species or processes they are intended to benefit.

Scale also affects the design of habitat restoration or monitoring. It might seem obvious, for example, that restoring even small areas of wetland should be worthwhile, since so much of the historic wetland has been lost and the cumulative effects of restoring many small areas may appear to be large. But if the scale of the restored patches is insufficient to meet the needs of the target species, or if the restored hydrology does not provide appropriate habitat or connectivity for fish feeding and movement, the restoration project may not have the desired results. Monitoring designs should capture the scales of the overall distribution of monitoring targets and the scale(s) on which they respond to environmental factors. These scales differ among species, making it difficult to design monitoring protocols that will be effective for multiple targets. A formal analysis of sampling design that takes landscape structure and scale into account could allow managers to determine the optimal density, distribution, and placement of sampling sites within the landscape mosaic. Unfortunately, such analyses are seldom undertaken, greatly compromising the efficiency and effectiveness of monitoring.

The upshot is that the appropriate scale (or scales) for measurement and analysis depends on what is being managed or restored, and for whom. Scale can be thought of as an umbrella that extends over landscape structure, function, and change (Figure 2). As scale changes, how people perceive landscapes and how the landscapes should be managed also change.

**LANDSCAPE LINKAGES**

A core principle of landscape ecology is that places in a landscape are linked together—what happens in one place affects what happens elsewhere, and the nature of the effects depends on the spatial configuration and composition of the landscape mosaic (Bennett 1999). These interconnections are determined in part by the permeability of boundaries—how they differentially hinder or facilitate movement of energy, materials, organisms, or disturbances among the patches in a landscape mosaic (Wiens et al. 1985; Hansen and di Castri 1992). Maintaining such movements in the face of ongoing fragmentation of habitats (which creates hard boundaries that impede movements) is the rationale for enhancing landscape connectivity, which has become a major focus of terrestrial landscape management and conservation (Crooks and Sanjayan 2006; Hilty et al. 2006; Lindenmayer and Fischer 2006). There is a plethora of methods (e.g., graph theory, network theory, least-cost path analysis; Pinto and Keitt 2002; Cushman and Huettmann 2010) to describe and

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1 The California Essential Habitat Connectivity Project (available at [https://www.wildlife.ca.gov/Conservation/Planning/Connectivity/CEHC](https://www.wildlife.ca.gov/Conservation/Planning/Connectivity/CEHC)) develops a strategy to enhance the connectivity of habitats across the state; the framework was used as the basis for evaluating habitat connectivity in the Delta as part of the Bay Delta Conservation Plan ([http://baydeltaconservationplan.com/](http://baydeltaconservationplan.com/)).
quantify landscape connectivity. Most of these relate to landscape structure (e.g., the physical networks in a GIS image; Figure 1), but landscape ecologists also talk of functional connectivity, how organisms, materials, etc. actually move from place to place (Tischendorf and Fahrig 2000).

Land and water are largely disconnected in the contemporary Delta landscape. A highly engineered network of levees directs flows and (most of the time) prevents the water from intruding into places where it is not wanted (Figure 1). At the same time, however, the engineered waterways of the rivers and Delta transport materials more quickly than would have occurred in the dendritic network of the past (Figure 3). This excessive connectivity can compromise navigational cues used by migrating fish, facilitate the spread of non-native species, and cause phytoplankton to be moved through the system before their populations can become large enough to fuel the food web. The increased connectivity of aquatic systems in the Delta contrasts with the reduced connectivity among terrestrial habitats and between terrestrial and aquatic habitats that has resulted from landscape alteration and fragmentation.

The ecological partitioning of water and land in the Delta has been accompanied by a bureaucratic

**Figure 3** The McCormack–Williamson Tract in the early 1800s (left) and contemporary aerial photography (right). Modifications to the Delta over time have altered hydrological flows and the way land and water interact. Source: Beagle et al. (2013).
partitioning of management and regulatory responsibilities among various state and federal agencies. Some, such as the U.S. Bureau of Reclamation, the California Department of Water Resources, or the U.S. Army Corps of Engineers, deal chiefly with the water (and the levees). Others, such as the Delta Protection Commission and the U.S. Department of Agriculture, focus on the lands behind the levees. Agencies charged with managing wildlife (the U.S. Fish and Wildlife Service, the National Marine Fisheries Service, the California Department of Fish and Wildlife) consider either water or land (or both), depending on the species of interest. Overall, more than 230 agencies, institutions, and stakeholders are involved in water and environmental management in the Delta (Luoma et al. 2015), many dealing with different parts of the same landscapes.

The message of landscape ecology is that, while landscape structure can be easily partitioned into discrete classes, ecological processes such as nutrient flows, food webs, or the movements of many species are not so restricted. Because landscapes are linked and interconnected across the boundaries between water and land, management practices and policies must also be. Agency mandates need to facilitate the important ecological exchange processes, and decision-makers need to understand that decisions about one element in a landscape have implications for adjacent (or even distant) elements.

**TAKING LANDSCAPE ECOLOGY INTO THE WATER**

The previous sections make clear not only that Delta landscapes encompass both land and water, but also that the conceptual framework of landscape ecology—structure, function, change, and scale—applies to aquatic ecosystems—waterscapes—as well as to the land (Poff 1997; Wiens 2002; Leuven et al. 2002). There are obvious differences, however, Most of the functions and dynamics of terrestrial landscapes are determined by the structure of the land, which is relatively fixed over short time periods. Variations in the medium above the substrate (the air) are important, but they do not dominate the ecological systems. In water it is reversed. The substrate beneath the water may have many of the same structural features as terrestrial landscapes—patches, corridors, boundaries, etc. The important processes, however, take place in the water.

As in most riverine ecosystems (Postel and Richter 2003; Auerbach et al. 2012; Webb et al. 2015), hydrological flows dictate much of the ecology of the Delta. Delta Smelt and their predators respond differently to patches of high-turbidity water, migrating salmon seek out streams of cool water that provide connectivity, and the shifting boundary of the low-salinity zone (X2) determines the distribution of pelagic organisms that are the foundation for aquatic food webs. Where channels intersect, complex and tidally changing flow dynamics affect how both actively migrating and passively drifting organisms distribute among the channels. Small changes in the physical structure of a channel can create fine-scale turbulence and eddies that provide feeding and resting opportunities for fish (Bennett and Burau 2015). At a broader scale, tidal surges, releases of cold water from upstream reservoirs, floods, droughts, and changing diversion patterns create highly variable temporal dynamics. These dynamics are what make management of the water and its occupants so difficult. Terrestrial landscape managers can change the structure of a mosaic and the changes will stick, at least for a while. Similar kinds of changes are much more difficult in the water. Managing landscapes in water is a slippery business. Modern fluid-dynamics models (e.g., Fong et al. 2009; Monismith et al. 2014) may allow one to build models of Delta channels and channel junctions and test how modifying the physical configuration would affect the waterscape. Provided one knew what kinds of fluid-dynamic features one wanted, the Delta could be restructured to provide those features. Moyle et al.’s (2010) proposal to reconfigure Delta channels to be more structurally complex so as to create spatial variation in water-residence time and local productivity reflects this kind of landscape thinking.

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2 X2 is the distance, measured upstream from the Golden Gate Bridge, where salinity at the river’s bottom is about 2 parts per thousand (ppt).
SETTING THE STAGE:
HISTORICAL LANDSCAPE CHANGE

Although we are constrained to manage landscapes (and waterscapes) in the Delta as they are now, history leaves its imprint on the present. Understanding past landscapes can provide a useful perspective on the interplay among structure, function, change, and scale.

According to the detailed historical ecology investigations carried out by the San Francisco Estuary Institute (Whipple et al. 2012; SFEI–ASC 2014), landscapes in the Delta before the major modifications of the 19th and 20th centuries had a finely reticulated structure of water, wetlands, and riparian and terrestrial vegetation, all functionally linked together and all subject to the time and space dynamics of tides and river flows. The aquatic ecosystem was a complex, fine-scale mosaic of patches that differed in depth, velocity, turbidity, salinity, and water-residence time. Tidal and freshwater wetlands and riparian woodlands linked the water to the land, promoting exchanges of nutrients and organic material that were the foundation of the aquatic food chain. Seasonal flooding reworked and redistributed sediments, activating foodwebs in enormous off-channel flood basins and creating and maintaining natural levees and complex river habitats. The seamless linkages among habitat types allowed unobstructed movement of organisms and materials, so there was a high degree of connectivity between adjacent habitats—organisms that depended on more than one habitat type had easy access to virtually any Delta habitat.

That is how the Delta used to be. But even then the landscapes and the ways they functioned were not the same everywhere in the Delta. At a broad scale, landscapes in different regions of the Delta were structured and functioned in different ways that reflected differences in the interplay among river hydrology, topography, and tides (see Figure 4; Whipple et al. 2012). The same was true of Suisun Marsh, where distinct subregions were defined by different geomorphologic and hydrologic processes (Manfree 2014).

These broad sub-regions of the Delta and Suisun Marsh were, in turn, mosaics of habitat patches. The modern Delta is also a mosaic of habitat patches, but the extensive marshes with their associated tidal channels have been almost entirely replaced by diked agricultural lands and the branching network of sloughs and channels has been straightened and simplified by levees lined with riprap rather than by wetland or riparian vegetation (SFEI–ASC 2014; Figure 1). Much of Suisun Marsh is similarly diked and is managed for waterfowl production by duck-hunting clubs (Moyle et al. 2014). Instead of the extensive and highly productive wetland transition zones between upland and water of historical landscapes, terrestrial and aquatic systems are now functionally partitioned. Instead of dendritic channels with variable water-residence times and complex productivity, channels are now mainly uniform and trapezoidal—they are designed to convey water efficiently, not to provide habitat for aquatic species. Instead of a terrestrial mosaic that provides habitat for many species, large areas are now dedicated to specific human uses. These dramatic changes in the structure and function of the Delta landscape place severe constraints on the Delta’s ability to support many native species, which poses a formidable challenge to managers.

CASE STUDIES OF LANDSCAPE CHANGE OVER TIME IN THE DELTA

A premise of landscape ecology is that that what happens in one place is contingent on structure, function, and change in the broader landscape—and everything is affected by scale. The only constant seems to be that “it all depends,” which may explain why the incorporation of a landscape perspective into management, conservation, and restoration in the Delta has been slow. To illustrate how a landscape perspective can help to provide a foundation for management in the Delta, we now consider three case studies. The first two draw from the historical landscape studies conducted by the San Francisco Estuary Institute (Whipple et al. 2012, SFEI–ASC 2014); the third provides a landscape perspective on an ongoing restoration effort (Beagle et al. 2013).

Inundation

Historically, inundation shaped the Delta landscapes. At the regional scale, the Delta was a meeting area of two great rivers coming from the north and south,
Figure 4  The three primary landscapes of the historical Delta. The map indicates the general extent of the north Delta (a landscape of flood basins, shown in green), central Delta (a landscape of tidal islands, shown in blue), and south Delta (a landscape of distributary rivers, shown in brown). Conceptual diagrams illustrating the variation in habitat mosaics among these landscapes are shown to the right. Source: Whipple et al. (2012).
with smaller tributaries entering from the east, and the tidal waters coming from Suisun Bay to the west. The intersection of these physical forces with varying climatic settings led to the development of distinctly different landscapes in the north, central, and south Delta (Whipple et al. 2012; Figure 4). In the central Delta, millennia of marshland peat formation created a flat topography where tidal inundation dominated and immense islands of tules grew between large tidal channels. These islands were wetted twice daily and flooded on spring tides. A dendritic network of tidal channels distributed nutrients, organic materials, and organisms into and out of the freshwater marshes. In the north Delta, strong seasonal flooding from abundant rain runoff and snowmelt in the northern Sierra watershed inundated extensive stands of non-tidal marshes for months at a time every winter and spring. Floods created broad, elevated natural levees that supported well-developed riparian woodlands. In the southern Delta, the snowmelt-driven flooding from the higher-elevation southern Sierra watershed peaked later in the year, providing spring-to-summer inundation that created a complex mosaic of oxbow lakes, off-channel ponds and slow-moving water. Natural levees were not as high as in the north Delta, and the topographic gradient was more gradual, creating a broad transition zone between fluvial and tidal influences. Around the edge of much of the Delta, wet meadows and small tributaries flooded for just a few days or weeks many times over the wet season, creating a soft transition from the vast Delta wetland to the adjacent uplands. On the western edge, sand mounds (remnants of Pleistocene dunes) rose above the marsh, providing high-tide refuges for terrestrial species in an otherwise wet landscape.

Across the Delta, these different types of floodplain habitat constantly shifted from wetter to drier and back again on different time-scales and at different times of year. These spatial and temporal dynamics supported a complex set of ecological functions. As an example, we can surmise how food webs might have been affected by inundation, using our current knowledge of wetland ecology to fill in the blanks where historical records are lacking. During the driest and warmest times of the year, in late summer and early fall, the central Delta marsh would have been flooded with the tides, while lakes in the north Delta would remain wet. Then, as rains came in the winter, the extensive northern flood basins would start to fill, activating food webs based on algae, wetland plants, and fluvial inputs that supported abundant invertebrates. Fish from the lakes, rivers, and tidal channels would spread out across the flooded basins to eat, followed quickly by wading birds, diving birds, mammalian predators, and any other consumers that could arrive to the feast (including indigenous people). This productivity throughout the spring would be punctuated by the shorter-duration floods at the edge of the Delta, which would activate food webs for cranes, amphibians, reptiles, and smaller terrestrial carnivores. As the north Delta inundation began to wane, the south Delta would flood from the San Joaquin River, and the complex distributary river habitats would feed invertebrates, fish (especially salmon; Sommer et al. 2001; Jeffres et al. 2008), and a variety of terrestrial consumers that could access the mosaic of shallow habitats. Thus, landscapes in different parts of the Delta would support food webs in different ways year-round, always characterized by extensive land–water connectivity driven by inundation.

The functional importance of this floodplain connectivity is illustrated by the consequences of recent breaching of levees at several locations within the Cosumnes River Preserve (Swenson et al. 2012), which allowed seasonal flooding of the land behind the levees. Juveniles of numerous fish species rapidly moved into the floodplain to feed in the seasonally available habitat. Juvenile salmon grew better in the seasonally flooded habitat than in either permanent ponds on the floodplain or in the adjacent river channel (Jeffres et al. 2008). Native and alien species also responded differently to inundation, native species being most abundant on the floodplain during larger scale flooding in spring when water temperature was cool (Crain et al. 2004).

Overall, the changes in inundation patterns of Delta landscapes have been profound (Figure 5). The wetland components of the landscape are mostly gone. The scales of floodplain inundation over space and time are now orders of magnitude less than they once were. Delta landscapes have changed from a dynamic, ceaselessly shifting wetland and aquatic mosaic of intricate complexity to a terrestrial landscape with stable patchiness (Figure 1), in which the aquatic
Figure 5 Approximate maximum extent and type of inundation in the historical (left) and modern (right) Delta. While the extent of perennial open-water features has increased over time, areas that experience tidal inundation, seasonal short-term flooding, and seasonal long-duration flooding have all decreased in extent. Source: SFEI-ASC (2014).
components are spatially well defined, temporally much less dynamic, and generally closely regulated. The largest floodplain of the Sacramento River, the Yolo Bypass, rarely floods enough to achieve significant food-web activation (SFEP 2015). In the rare years when the Yolo Bypass is fully inundated, the flood lasts for just a few days and covers only a small fraction of the extent of the historical north Delta flood basins (Figure 5). The San Joaquin River has little active floodplain (although some areas have been re-established on the Cosumnes River).

Given the magnitude of these changes, the goal of management and restoration should not be to replicate the historical hydrodynamics of the Delta, but instead to use this historical information to help determine where in the landscape re-establishment of more natural hydrodynamic processes might most effectively support ecosystem functions such as food-web dynamics.

Riparian Woodlands

Although the loss of riparian woodlands since the early 1800s may seem moderate in comparison with loss of marshes, the changes in the landscape structure of riparian patches are arguably more profound (Figure 6). Historically, the woody riparian areas of the Delta were connected, dendritic networks that were built and maintained by sediment from flooding rivers. They were like long, spidery fingers that spanned enormous distances along fluvial channels. With its massive floods and high sediment load, the Sacramento River created by far the largest of these riparian networks—gallery forests of oak and sycamore with multi-layered understories that provided structurally complex habitat for wildlife. The San Joaquin River maintained a smaller network of narrower woody riparian areas dominated by willows and other shrubs. The Cosumnes River had a still smaller woody riparian network.

The structure of riparian habitats influenced ecological functions at several scales. At the landscape scale, woody riparian habitats were highways for terrestrial animals to move deep into the marsh and floodplains—a way to get to the wetland food resources on offer. Larger species like mule deer and coyotes could travel down the riparian corridors and then work along the marsh edge to forage. Smaller species like insectivorous bats and birds would have used the riparian structure for cover while hunting along the edge of the marsh or the channel. Thus, woody riparian tendrils greatly increased the amount of wetland edge accessible to terrestrial wildlife. At the local scale, the structural complexity of woody riparian areas offered cover from predators, shelter from the elements, and reproductive sites for a diverse suite of animals, from spiders to birds to medium-sized carnivores.

Woody riparian areas were shaped by regular flooding regimes that varied in intensity according to the water year. While the main channels within the tidal zone were probably fairly stable, flooding could scour and rebuild physical substrate, resetting plant-community succession. The floods would also have shifted the entire landscape to a temporary semi-aquatic state, with rapidly increasing productivity, and a host of animals arriving to consume it.

Although woody riparian areas still occur widely in the Delta, their structure is greatly altered, especially at the landscape scale. Large natural levees are now occupied by orchards and towns along the Sacramento River, and are used for agriculture along the San Joaquin. Where two large, contiguous woody riparian networks once flanked the great rivers in the north and south Delta, thousands of tiny riparian fragments are now scattered widely across the Delta’s artificial levees, mostly in places where woody riparian plants did not historically occur (SFEI-ASC 2014; Figure 6). These fragments are much narrower than the historical gallery forests, and the adjacent habitats have largely changed from marsh to agriculture. At a finer scale, the riparian structure has shifted to younger and smaller trees, often of non-native species. Present-day woody riparian areas still support some riparian wildlife, but the ecological functions have diminished as the structure and flooding regimes have been altered. Riparian corridors no longer provide uninterrupted pathways to productive areas for terrestrial animals to feed. Animals living in most woody riparian fragments are subjected to the effects of diminished patch size, severed connections, and increased threats from the surrounding landscape.

The most obvious management and conservation solution to these problems is to restore more riparian
Figure 6  Historical riparian habitat was predominately continuous forest (left), while today woody riparian habitat is scattered throughout the Delta in small isolated patches (right). The longest stretch of contiguous riparian forest historically spanned more than 55 km, providing a migration corridor across much of the Delta. The longest current stretch of woody riparian habitat extends 16 km. Source: SFEI-ASC (2014).
woodland habitat in contiguous networks adjacent to marsh. But there are also obvious constraints. Most levees in the Delta are, at best, suitable only for narrow strips of riparian vegetation, and there is continuing controversy about whether woody vegetation on levees is a good or a bad idea (i.e., providing wildlife habitat versus weakening levees by rooting or harboring burrowing mammals). The opportunities to expand riparian woodlands may be greatest where levee set-backs are part of a restoration plan. However, re-establishment of riparian woodlands should consider the adjoining hydrology (e.g., seasonal flooding, sediment deposition or erosion) and the composition of the surrounding terrestrial or wetland landscape—whether it will support predators that prey on animals in the riparian zone or provide additional food sources. Placing riparian management and restoration in a landscape context requires data and careful thought (and spatial modeling wouldn’t be a bad idea), but it can help to determine the potential effectiveness of actions.

Changes in a Delta Island: The McCormack–Williamson Tract

A landscape perspective is particularly relevant to habitat restoration. Restoration is generally undertaken on small parcels because the availability of suitable places is limited and the per-acre costs of restoration can be large. However, the parcels are parts of broader landscapes, which can alter the effectiveness of the restoration.

One of the largest restoration efforts underway in the Delta is the McCormack–Williamson Tract (MWT), a 1,645-acre (6.6-km²) property along the Mokelumne River in the northeast Delta (Figure 1). Historically, the MWT lay at the transition between tidal and fluvial influences (Figure 3) (Beagle et al. 2013). Shallow but frequent tidal flows entered the island on its western side. Deeper and longer-duration riverine flood flows would spread across the MWT from the north as the Sacramento flood basin filled with snowmelt and runoff. The eastern side of the MWT was defined by the natural levee of the Mokelumne River, which supported a broad riparian forest that contrasted with the tules and freshwater wetland vegetation of the rest of the island. Because of this natural levee, hydrological connectivity to the Mokelumne was probably restricted to high-flow events that would overtop the levee and flow through the forests into the MWT. This landscape structure of varying elevations and modes of hydrologic connection supported a dynamic and complex array of floodplain and riparian functions.

The MWT is now hemmed in on all sides by artificial levees. Except for unusually extreme flood events, the levees block hydrologic connectivity, creating a terrestrial environment suitable for agriculture. The restoration efforts aim to reintroduce tidal and fluvial flood flows to the island by opening strategic breaches in the levee system. Re-establishing hydrologic connectivity will allow these flows to restructure the landscape into spatial patterns organized by a dendritic tidal-channel network with varying water depth and chemistry that change with tidal and seasonal cycles. Hopefully, this new structure will support aquatic species and food webs that have long been missing.

Of course, the MWT is also part of a larger landscape. This limits how far restoration of connections can proceed, at least over the short term. The perimeter levee must be maintained to ensure that adjacent properties are protected from flooding. This precludes the potential for restoring habitat connectivity to the mature Delta Meadows marshland immediately to the west, which would create a larger wetland complex with greater diversity. It also precludes the re-establishment of a self-maintaining riparian forest on the natural levee of the Mokelumne River, one of the few locations in the Delta suitable for such restoration at a significant scale. These current barriers to landscape linkages also prevent connecting the marsh plain and potential riparian forest to adjacent landscape features immediately upslope, which would enable tidal and low-elevation species and habitats to move upstream as sea level rose.

Although the MWT is large as restoration projects go, it is small compared to the expanse of marshes in the historical Delta, where 98% of the marsh was in areas of 2500 acres (1000 ha) or larger. Today, there is not a single marsh patch of that size. This fragmentation limits the ability of Delta marshes to support physical and ecological processes (SFEI–ASC 2014). If the MWT could be connected to neighboring existing (and future) marshes, it could create a large...
marsh complex that supported landscape processes at a scale currently not represented in the Delta.

Thus, while some of the natural structure, function, and dynamics of Delta ecosystems can be restored within the landscape of the MWT, the restoration can be only a piece of a broader landscape subject to multiple uses (Figure 1). Expanding the scale in space and time indicates several actions that could be undertaken within or beyond the MWT to enhance the restoration effectiveness (Figure 7). For example, when the lease on a radio tower located within the MWT expires, the parcel could become available for expanding wetlands or other landscape elements. At a broader spatial scale, acquiring properties that would link the MWT with other restored or protected lands, such as the Cosumnes Preserve, could help to restore overall landscape functionality. In a similar fashion, the Regional Restoration Strategies recommended in the Delta Plan (DSC 2013) and California EcoRestore3 can be part of a larger vision to restore more functional and resilient landscapes in the Delta.

3 http://resources.ca.gov/ecorestore/

CONCLUDING COMMENTS

Landscape ecology has a well-established foundation as a basic and applied science. Yet its application to resource management and restoration has come slowly to the Delta. The State of Bay–Delta Science, 2008 (Healey et al. 2008) acknowledged the holistic nature of the Delta, the importance of land–water linkages, the importance of connectivity, and the reality of landscape change. Luoma et al. (2015) emphasize similar points. But the specifics of just how differences in particular features of landscape structure can influence specific functional properties of ecosystems; how the dynamics of change differ in different places in a landscape; how linkages or boundaries among landscape elements affect movements of contaminants or invasive species; how different scales of management or analysis may affect the outcomes of actions—these were, and still are, largely missing from Delta science. Even though the Delta is one of the most intensively studied ecosystems in the world, applications of landscape science, concepts, and thinking have lagged behind.

Figure 7 Several options for expanding the time and/or space scales of restoration of the McCormack–Williamson Tract (MWT). Adapted from Beagle et al. (2013).
There are many reasons, not the least of which is the combination of the complexity of Delta ecosystems, the decades of controversy over water allocations and management, and the byzantine web of agencies, jurisdictions, and agendas that determine management practices. All of these reinforce a tendency to focus on individual sites or projects, divorced from their broader landscape context. The overwhelming emphasis on aquatic systems and the associated focus on getting hydrological flows “right” (however defined) has also fostered a neglect of landscapes that integrate land, water, and the places in between. A synthetic, landscape approach is needed in Delta science.

**RECOMMENDATIONS**

We suggest that several actions might help us reach this goal. These echo the recommendations offered by Luoma et al. (2015), but with an emphasis on landscape and spatial data and analysis.

- Develop a holistic landscape vision across broad areas or regions of the Delta, to integrate aquatic habitats with terrestrial and wetland habitats, emphasize functional interconnections, and capture synergies among individual projects;
- Foster inter-agency collaboration for landscape structure and functions rather than (or in addition to) traditional agency domains and agendas. Agencies cannot afford to collaborate everywhere, so orienting their shared activities about shared landscapes makes sense;
- Develop a spatial information management system. Common libraries of digitized, spatially explicit information on multiple aspects of landscapes (e.g., Figure 1) can help to show how actions on some elements of a landscape in some places will affect and be affected by the structural configuration of other elements or places in a landscape;
- Analyze spatial data on multiple physical, chemical, and ecological factors to identify the spatial relationships of opportunities and constraints, and show where the return on investment in habitat restoration may be greatest;
- Enhance scientific capacity in landscape modeling and quantitative analysis. Spatial modeling is often the quickest and most efficient way to integrate disparate kinds of information into a common landscape setting, and to explore alternative restoration or management options;
- Use spatial analyses and landscape maps to overlay projects, agency responsibilities, key resources, land uses, ownership, species distributions, or ecological functions to show where opportunities to integrate projects and actions might exist, or which habitat patches and physical processes need to be in place in which areas of the Delta for the landscape to support key life stages of a given species;
- Finally, consider the scale(s) of management or restoration actions and the anticipated and actual responses, and use landscape analyses to ensure the compatibility of the scale(s) of actions and desired outcomes.

Currently, landscape science in the Delta might be characterized as a state of general awareness of landscapes and multiple perspectives about what is important about them. Building on this awareness requires a greater depth of understanding of the details of landscape structure, function, change, and scale and how they apply to the Delta. The concepts and tools of landscape ecology are well developed; they now need to be applied and integrated to determine how a landscape approach can enhance the effectiveness of management and restoration in the Delta.

**ACKNOWLEDGMENTS**

Thanks to Martina Koller and Ruth Askevold for preparing the figures.
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Jeffres, CA, Opperman JJ, Moyle PB. 2008. Ephemeral floodplain habitats provide best growth conditions for juvenile Chinook Salmon in a California river. Env Biol Fish 83(4)449–458. doi: http://dx.doi.org/10.1007/s10641-008-9367-1


ABSTRACT

The State of Bay–Delta Science 2008 highlighted seven emerging perspectives on science and management of the Delta. These perspectives had important effects on policy and legislation concerning management of the Delta ecosystem and water exports. From the collection of papers that make up the State of Bay–Delta Science 2016, we derive another seven perspectives that augment those published in 2008. The new perspectives address nutrient and contaminant concentrations in Delta waters, the failure of the Delta food web to support native species, the role of multiple stressors in driving species toward extinction, and the emerging importance of extreme events in driving change in the ecosystem and the water supply.

The scientific advances that underpin these new perspectives were made possible by new measurement and analytic tools. We briefly discuss some of these, including miniaturized acoustic fish tags, sensors for monitoring of water quality, analytic techniques for disaggregating complex contaminant mixtures, remote sensing to assess levee vulnerability, and multidimensional hydrodynamic modeling.

Despite these new tools and scientific insights, species conservation objectives for the Delta are not being met. We believe that this lack of progress stems in part from the fact that science and policy do not incorporate sufficiently long-term perspectives. Looking forward half a century was central to the Delta Visioning process, but science and policy have not embraced this conceptual breadth. We are also concerned that protection and enhancement of the unique cultural, recreational, natural resource, and agricultural values of the Delta as an evolving place, as required by the Delta Reform Act, has received no critical study and analysis. Adopting wider and longer science and policy perspectives immediately encourages recognition of the need for evaluation, analysis, and public discourse on novel conservation approaches. These longer and wider perspectives also encourage more attention to the opportunities provided by heavily invaded ecosystems. It is past time to turn scientific and policy attention to these issues.

KEY WORDS

Sacramento–San Joaquin Delta, scientific advances, new scientific perspectives, new scientific tools, policy perspectives, novel conservation methods.
CONNECTING THE PAST WITH THE PRESENT

The State of Bay–Delta Science 2008 (Healey et al. 2008a, hereafter SBDS 2008) synthesized the scientific literature on the Sacramento–San Joaquin Delta that addressed the issues considered most important at that time. SBDS 2008 was written mainly for managers and policy makers and was successful in its objectives. The papers included in the special issues of San Francisco Estuary and Watershed Science that make up the State of Bay–Delta Science 2016 (SBDS 2016)\(^1\), on the other hand, are written for a more technical audience, although we hope they will still be of value to policy-makers. As was the case in 2008, it was not possible to include in SBDS 2016 a paper on every scientific issue that affects water supply and the Delta ecosystem. To narrow the list, we canvassed senior managers, policy-makers and scientists familiar with the Delta and eventually selected 12 topics that led to the list of papers published in San Francisco Estuary and Watershed Science (Table 1). Some of the topics reprise issues examined in 2008 (e.g., water supply, levees, food webs) but take a different perspective and bring in new information. Others address issues that have come more to the fore since 2008 (e.g., nutrients, contaminants, predation). The chapters still address only a partial list of relevant topics, however, and we anticipate that additional papers will be commissioned as new concerns arise, keeping SBDS an up-to-date, growing source of synthesized information for Delta scientists and managers.

SBDS 2008 began with seven new perspectives on how the Delta functions that emerged from research conducted over the preceding decade (Table 2). In this paper, we draw from SBDS 2016 and other sources to examine how the perspectives put forward in 2008 have influenced policy and management; how well those perspectives have held up under a decade of intense research on the Delta; how they have evolved as a result of new information and changing conditions in the Delta; what new insights and perspectives have emerged since 2008; and how those new perspectives are contributing to the coequal goals of a reliable water supply and a healthy ecosystem in the context of protecting the Delta as a unique place.

It is a measure of how thinking about the Delta has evolved that the 2008 perspectives now appear self-evident whereas in 2008 they were quite novel. New Perspective 1 (Table 2) emphasized that the Bay–Delta is not a static environment and never has been. Human occupation of the Delta, particularly after 1850, brought accelerated change as well as entirely new directions of change. The ecosystems of the Delta are now structurally and biologically very different from those that existed before the Gold Rush (Whipple et al. 2012); they are novel ecosystems (Hobbs et al. 2006). As such, it is not possible to conserve desired native species by recreating the historic conditions in which they evolved. The new ecosystems, along with the changes impinging from beyond the Delta, will impede and likely undermine attempts to restore those past conditions. With climate change, sea level rise, and the possibility that water may be diverted around the Delta as part of California WaterFix\(^2\), Delta ecosystems will continue to morph and adapt to new conditions. The Delta presents a moving target to which managers must adjust in their pursuit of the coequal goals of a reliable water supply and a healthy ecosystem. Perspectives 2 and 3 in 2008 were corollaries of Perspective 1, which emphasized the uncertainty and complexity that go with continual change (Luoma et al. 2015) but also the necessity of encouraging variability to support desired species and ecosystem services. Managing novel ecosystems calls for novel and flexible approaches (Seastedt et al. 2008). As described in Healey (2008) and further elaborated upon in Luoma et al. (2015), the Delta presents managers with a “wicked” problem (Rittel and Weber 1973) that cannot be solved but only managed.

Perspectives 4 and 6 were also interlinked, emphasizing that water and environmental management are not separate problems and that creative, integrated science is needed to help resolve the water supply/healthy environment nexus. The perspectives also acknowledged that, while providing insights into one set of problems, science may at the same time identify other, unsuspected problems. The nature of wicked problems is that even as you take action to resolve one issue, new issues arise.

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Table 1  List of topics and papers published in *San Francisco Estuary and Watershed Science* as the SBDS 2016. Asterisked topics were also addressed in SBDS 2008 (Healey et al. 2008a).

<table>
<thead>
<tr>
<th>Topic</th>
<th>Published Paper Title</th>
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<tbody>
<tr>
<td>Managing the Delta</td>
<td>Luoma SN, et al.  <em>Challenges Facing the Sacramento–San Joaquin Delta: Complex, Chaotic, or Simply Cantankerous?</em> doi: <a href="http://dx.doi.org/10.15447/sfews.2015v13iss3art7">http://dx.doi.org/10.15447/sfews.2015v13iss3art7</a></td>
</tr>
<tr>
<td>Water Supply*</td>
<td>Lund JR.  <em>California’s Agricultural and Urban Water Supply Reliability and the Sacramento–San Joaquin Delta.</em> doi: <a href="http://dx.doi.org/10.15447/sfews.2016v14iss3art6">http://dx.doi.org/10.15447/sfews.2016v14iss3art6</a></td>
</tr>
<tr>
<td>Climate Change</td>
<td>Dettinger M, et al.  <em>Climate Change and the Delta.</em> doi: <a href="http://dx.doi.org/10.15447/sfews.2016v14iss3art5">http://dx.doi.org/10.15447/sfews.2016v14iss3art5</a></td>
</tr>
</tbody>
</table>

Table 2  New perspectives on the Delta as presented in SBDS 2008 (Healey et al. 2008b)

<table>
<thead>
<tr>
<th>Number</th>
<th>Perspective</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>The Delta is a continually changing ecosystem. Uncontrolled drivers of change (e.g., population growth, changing climate, land subsidence, seismicity) mean that the Delta of the future will be very different from the Delta of today.</td>
<td></td>
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<tr>
<td>2</td>
<td>Because the Delta is continually changing, we cannot predict all the important consequences of management solutions. The best solutions will be robust but provisional, and will need to be responsive and adaptive to future changes.</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>It is neither possible nor desirable to freeze the structure of the Delta in its present, or any other form. Strengthening of levees is only one element of a sustainable solution and is not applicable everywhere.</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>The problems of water and environmental management are interlinked. Piecemeal solutions will not work. Science, knowledge and management methods all need to be strongly integrated.</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>The capacity of the Sacramento-San Joaquin water system to deliver human, economic and environmental services is likely at its limit. To fulfill more of one water-using service we must accept less of another.</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>Good science provides a reliable knowledge base for decision-making, but for complex environmental problems, even as we learn from science, new areas of uncertainty arise.</td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>Accelerated climate change means that species conservation is becoming more than a local habitat problem. Conservation approaches need to include a broad range of choices other than habitat protection.</td>
<td></td>
</tr>
</tbody>
</table>
In the Delta Reform Act of 2009, the legislature implicitly acknowledged the multifaceted and complex nature of the Delta management problem in paragraph 85302, which includes a long list of characteristics of a healthy ecosystem and reliable water supply that were to be addressed by the Delta Plan (Table 3). They further acknowledged the importance of science to successful management by mandating creation of the Delta Science Program with the mission: “...to provide the best possible unbiased scientific information to inform water and environmental decision-making in the Delta...through funding research, synthesizing and communicating scientific information to policy makers and decision-makers, promoting independent scientific peer review, and coordinating with Delta agencies to promote science-based adaptive management.” SBDS 2016 is, in fact, one of the Science Program’s initiatives to synthesize and communicate scientific information to policy-makers and decision-makers. The Delta Reform Act also established the Delta Stewardship Council (the Council) as a new governance entity to help guide the overall management and development of the Delta. The Council is to coordinate the actions of state and, through leadership, federal, agencies responsible for water and environmental management, thereby helping to avoid the fragmented and sometimes conflicting actions that occur when agencies act independently. The Council also tracks progress using a suite of performance measures and facilitates avoidance of damaging actions. These charges to the Council reflect the ongoing incorporation of adaptive management into

### Table 3: Some of the requirements for the Delta Plan as listed in paragraph 85302 of the Delta Reform Act of 2009

<table>
<thead>
<tr>
<th>(c) The Delta Plan shall include measures that promote all of the following characteristics of a healthy Delta ecosystem:</th>
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<tbody>
<tr>
<td>1 Viable populations of native resident and migratory species.</td>
</tr>
<tr>
<td>2 Functional corridors for migratory species.</td>
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<tr>
<td>3 Diverse and biologically appropriate habitats and ecosystem processes.</td>
</tr>
<tr>
<td>4 Reduced threats and stresses on the Delta ecosystem.</td>
</tr>
<tr>
<td>5 Conditions conducive to meeting or exceeding the goals in existing species recovery plans and state and federal goals with respect to doubling salmon populations.</td>
</tr>
</tbody>
</table>

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<tr>
<th>(d) The Delta Plan shall include measures to promote a more reliable water supply that address all of the following:</th>
</tr>
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<tbody>
<tr>
<td>1 Meeting the needs for reasonable and beneficial uses of water.</td>
</tr>
<tr>
<td>2 Sustaining the economic vitality of the state.</td>
</tr>
<tr>
<td>3 Improving water quality to protect human health and the environment.</td>
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</tbody>
</table>

<table>
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<tr>
<th>(e) The following sub-goals and strategies for restoring a healthy ecosystem shall be included in the Delta Plan:</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Restore large areas of interconnected habitats within the Delta and its watershed by 2100.</td>
</tr>
<tr>
<td>2 Establish migratory corridors for fish, birds, and other animals along selected Delta river channels.</td>
</tr>
<tr>
<td>3 Promote self-sustaining, diverse populations of native and valued species by reducing the risk of take and harm from invasive species.</td>
</tr>
<tr>
<td>4 Restore Delta flows and channels to support a healthy estuary and other ecosystems.</td>
</tr>
<tr>
<td>5 Improve water quality to meet drinking water, agriculture, and ecosystem long-term goals.</td>
</tr>
<tr>
<td>6 Restore habitat necessary to avoid a net loss of migratory bird habitats and, where feasible, increase migratory bird habitat to promote viable populations of migratory birds.</td>
</tr>
</tbody>
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4 The already existing CALFED Science Program was simply transferred to the Delta Stewardship Council, maintaining a continuity of the science coordination begun under CALFED.
the operating procedures of state and federal agencies that began with CALFED (2000).

Perspective 5 pointed to the obvious but often-ignored fact that California has limited water resources, so hard decisions have to be made about water allocation. Indeed, much of the water supply/healthy environment conflict in the Delta has been about how water is allocated. Although the Delta Reform Act does not specifically mention limitations to water supply, it does establish that the policy of California is to reduce the state’s reliance on the Delta for water. This is an implicit admission that the Delta, indeed California as a whole, is at the limit of its surface and groundwater supplies and that the future lies in making smarter use of the state’s finite supply of water (DSC 2013; Luoma et al. 2015).

The Delta Reform Act of 2009, the Delta Plan (DSC 2013) and the Delta Science Plan (DSP 2013) are all consistent with the seven perspectives presented in 2008. Both the Delta Plan and the Delta Science Plan are in early stages of implementation; however, there are already some notable successes, such as the establishment of the Delta Regional Monitoring Program, which reflects the improved monitoring proposed in SBDS 2008’s Perspective 4. As required under the act, the Delta Plan lists a number of performance measures by which to gage progress. This is another new and positive feature of Delta management that was proposed in the 2008 perspectives. Unfortunately, many of the measures are satisfied by “improvement” as determined by status and trends monitoring. The problem is that conditions can be “improving” at the same time as species are going extinct (Moyle et al. 2016). The Delta Plan is to be updated on a regular basis. It is possible that within a few years advances in multidimensional fluid dynamics modeling coupled with physical, chemical, and ecological process modeling (MacWilliams et al. 2016; Perry et al. 2016; Schoellhamer et al. 2016) will allow specific endpoints to be identified that will constitute positive environmental and water supply outcomes. At the very least, development and testing of such models can become a priority in the Science Action Agenda (DSP 2013).

We can, therefore, make a case that the perspectives identified in SBDS 2008 have influenced the evolution of policy and management for a reliable water supply and a healthy environment in the Delta. Furthermore, we argue that the new perspectives identified in SBDS 2008 remain relevant today, even as the evolving science of the Delta is bringing to light additional perspectives that will affect how the Delta is managed.

NEW PERSPECTIVES, 2016

In the years since SBDS 2008, Delta science has provided new insights into the functioning of the Delta ecosystem and into water supply and management. Here we offer seven new perspectives that emerge from the science.

1. Nutrients are important.

Whereas in the past we considered nutrients to be relatively unimportant in Delta productivity, increasing water clarity means that both the absolute and relative concentrations of different nutrients in the Delta can now be drivers of Delta productivity, including inhibition of phytoplankton growth by ammonium and promotion of the expansion of invasive Microcystis, Egeria, and Eichhornia.

Over several decades of research, a view of production at the base of the Delta food web (primary production) became established; nutrient concentrations were high, nutrient limitation was rare, and factors other than nutrients (e.g., grazing by Potamocorbula amurensis, water residence time, high turbidity) regulated rates of primary production (Jassby et al. 2002). That is to say, there seemed to be little need to worry about nutrient levels in the Delta. As turbidity is declining in the Delta, however, this view is giving way to a more complex narrative that considers the various forms of nitrogen as well as nutrient ratios, fluid dynamics, and the role of invasive species in productivity at the base of the food web (Dahm et al. 2016; Schoellhamer et al. 2016). Of particular interest now are the nutrient conditions that promote development of blooms of cyanobacteria, especially Microcystis, and expansion of the waterweeds Egeria densa and Eichhornia crassipes. High nutrient loads coupled with long
water residence times may contribute to the growth of these invasive toxic and nuisance species.

In the past, high turbidity conferred a measure of resilience against high nutrient loading in the Delta, and high ammonium levels may also have inhibited phytoplankton growth in some parts of the Delta, preventing the development of blooms. Now, however, increasing water clarity (Schoellhamer et al. 2016) may allow phytoplankton to grow fast enough to escape the inhibitory effect of ammonium, and to develop blooms fueled by the abundant nitrate. But phytoplankton in the Delta will now be competing with *Microcystis*, *Egeria*, and *Eichhornia*, which can absorb ammonium preferentially and outcompete phytoplankton for the nutrient pool (Dahm et al. 2016). How these invading species will affect the food web is, as yet, uncertain.

The situation is complicated further because much of the ammonium that enters the Delta comes from wastewater treatment plants (Jassby 2008), which discharge a myriad of other substances (some toxic) that can also inhibit plankton growth (Dahm et al. 2016; Fong et al. 2016). Wastewater treatment plant upgrades to eliminate ammonium may not remove other pollutants, so that the upgraded discharges may still inhibit phytoplankton growth. The reality is that numerous environmental variables (abiotic and biotic) can affect the growth of phytoplankton and other aquatic plants, and different factors may play a role in different areas of the Delta. To develop effective nutrient and bloom management strategies, a holistic understanding is needed about the factors that affect phytoplankton growth in the estuary. The planned upgrades to the Sacramento Regional Wastewater Treatment Plant (SRWTP) offer an opportune experiment in nutrient reduction that could provide further insights into the relationship among nutrient loading, plant growth, and the functioning of the Delta ecosystem (Dahm et al. 2016).

2. Delta waters are contaminated.

The complex cocktail of contaminants that enters the Delta from agriculture, urban, and industrial discharges can cause serious damage to the ecosystem and human health.

The CALFED Ecosystem Restoration Program strategic plan (CALFED 2000) identified the effects of contaminants that enter the Delta as an area of critical uncertainty in need of focused research. SBDS 2008 echoed this need, noting that poor environmental water quality was a potential human health risk and a factor that contributed to ecological problems (Luoma et al. 2008). Despite the well-documented need for a comprehensive program of contaminant monitoring and assessment in the Delta, little has been accomplished. Recently, the Central Valley Regional Water Quality Control Board established the Delta Regional Monitoring Program. Once it is fully operational, this program will sample mercury, pesticides, nutrients, and pathogens at a number of Delta locations, both routinely and in response to events such as seasonal flush, storms, and dry periods. The program ignores pharmaceuticals and personal care products among other contaminants, but, if implemented as described, it will be an important start toward developing an understanding of the status and trends of selected problem constituents in Delta water.

Targeted research in the Delta and adjacent waters has shown that Delta water is often acutely or sublethally toxic to a range of aquatic organisms, particularly near sources of urban or agricultural discharge (Brooks et al. 2012; Biales et al. 2015). Runoff from agricultural lands is a source of both nutrients and pesticides, and the pesticides in use change regularly, often in response to concerns about toxicity. For example, use of organophosphate and carbamate pesticides has declined because of insect tolerance, the toxicity of organophosphates to humans and the persistence of carbamates in the environment. These pesticides were largely replaced by pyrethroids and neonicotinoids, both of which are highly toxic to invertebrates, notably to the very bees on which agriculture relies (Sanchez–Bayo et al. 2014). Pyrethroids were believed to be less persistent in the environment than carbamates. However, pyrethroids adsorb to sediments and are persistent when adsorbed. In this form, pyrethroids can be carried with river borne sediments to deposition sites where they may accumulate to concentrations acutely toxic to test organisms (Fong et al. 2016).
Neonicotinoids are soluble and persist in the environment for a long time (Fong et al. 2016). Acute toxicity of these pesticides in the ecosystem is rare but sublethal concentrations can contribute to population declines by, for example, making organisms more vulnerable to predation or interfering with reproduction.

Contaminants likely played a role in the pelagic organism decline (POD) that began in the early 2000s (Sommer et al. 2007). For example, Fong et al. (2016) used multi-factorial models to examine the relationship between species abundance indices in fall midwater trawls and a range of flow and water-quality variables. Pyrethroid use and flow explained the most variability in species abundance, except for Threadfin Shad (*Dorosoma petenense*), for which only pyrethroids were significant determinants. Among all species, pyrethroid use explained 21% to 73% of the variability of species abundance indices in fall midwater trawl samples. Although multiple factors likely contributed to the POD, this analysis provides strong evidence that contaminants were a contributing factor.

Once regular monitoring of contaminants is underway, analyses will be possible to improve our understanding of interacting and sublethal effects of contaminants in the Delta. As monitoring data accumulate, new multi-dimensional models of hydrodynamics and distribution of water constituents should become feasible (MacWilliams et al. 2016; Schoellhamer et al. 2016), allowing dispersal and concentrations of contaminants moving through the Delta to be predicted. Coupled with data on lethal and sublethal toxicities, species distributions, and food webs (Brown et al. 2016; Fong et al. 2016; Perry et al. 2016), these tools should facilitate assessments of the species-specific and ecosystem consequences of contaminant loadings in the Delta. Such analyses would have been nearly impossible a decade ago.

3. **Aquatic food webs no longer sustain native species.**

*Food webs in the Delta now bear little resemblance to those that supported communities of native organisms before European colonization. Driven by physical and chemical changes in the Delta and invasions by alien species, the aquatic ecosystem has gone through an ecological regime shift that probably cannot be reversed. The present food web appears stable but is much less able to support native fishes than in the past.*

The Delta food web today bears little resemblance to that which existed before 1850. Key among the causes of this change were the physical transformation of the Delta and its invasion by exotic species. When the Delta was transformed from a landscape of marshes and flood basins with a complex drainage system into a landscape of leveed islands separated by wide linear channels, the base of the food web shifted from high quality organic detritus from the marshes and flood basins to phytoplankton produced in the open waters of the channels. Much more recently, the invasive overbite clam, *Potamocorbula amurensis*, co-opted the majority of phytoplankton production, resulting in a Delta in which native fishes are severely food limited (Brown et al. 2016). The low-salinity zone, once the most productive region of the Delta, now produces little food, and its foragers are sustained by imports from upstream and downstream.

Although changes in Delta geometry and the invasion of the overbite clam can be identified as critical events, the transformation of the Delta food web unquestionably has multiple causes (e.g., water operations affecting Delta hydrology [MacWilliams et al. 2016], contaminant discharges [Fong et al. 2016], additional species invasions [Brown et al. 2016], sediment load [Schoellhamer et al. 2016]). The food web appears to have been relatively stable for the past 15 years or so (Brown et al. 2016) but new stressors (e.g., climate change [Dettinger et al. 2016], contaminant loads [Fong et al. 2016], further changes in hydrology [MacWilliams et al. 2016], and new invasions [Brown et al. 2016]) can push the ecosystem in undesirable directions. Any actions to improve conditions for native species will have to be undertaken in the light of this new food web structure. There is no returning to the pre-1970s structure. Delta Smelt (*Hypomesus transpacificus*), currently on the brink of extinction (Moyle et al. 2016), will likely be the first native fish species to succumb to the new regime. Others have only tenuous holds on survival (e.g., Katz et al. 2013). Improving conditions for listed species in the Delta remains a major challenge. And, as climate change begins to exert a stronger influence, it may become...
necessary to consider more radical approaches to species conservation.

4. Species declines are a result of multiple stressors acting together.

There are few instances in which a single stressor can be identified as the primary cause of any species’ decline. Effective conservation of aquatic species requires a holistic approach to improve habitat quantity and quality.

Many species in the Delta are listed as threatened or endangered. The traditional approach to recovering viable populations of these species has been to search for the most important stressor and try to reduce its effect on the species. In the Delta, conservation actions have focused on improving hydrology and restoring habitat. It is now apparent, however, that no single stressor can be singled out as the “cause” of Delta species’ declines. Rather, numerous stressors acting together are increasing the vulnerability of each species to the point that a viable population can no longer be sustained. The Delta Smelt illustrates this complexity very well. When this species first began to decline, attention focused on water exports and the changes in Delta hydrology that resulted in smelt being lost at the export pumps (Moyle et al. 1992). A precipitous decline in Delta Smelt and three other fish species beginning about 2002 (the POD) stimulated a focused analysis of available information, which showed that water export was only one cause of species declines in the Delta (Sommer et al. 2007). Research has continued so that Delta smelt, now virtually extinct, is one of the most thoroughly studied fish species in the world. However, the causes of its demise remain elusive. Food limitation (Brown et al. 2016), exposure to toxic chemicals (Fong et al. 2016; Moyle et al. 2016), interaction with exotic predators (Grossman 2016), and shrinking areas of suitable habitat (Moyle et al. 2016) are all implicated, in addition to water operations and water exports from the Delta. To this list of specific stressors must be added the artificial geometry of the Delta, which represents an alien habitat for most native species.

Native species conservation in the Delta is, therefore, a multi-factorial problem. It is unclear which combinations of stress reductions would lift enough of the burden from native species to allow their persistence. Any viable solution must, however, address multiple stressors. To improve conditions for native species, Moyle et al. (2010) proposed restoring channel complexity and variable water residence time to mimic conditions before 1850, along with improving water quality and increasing marsh and brackish open-water habitat. Current emphasis is on reestablishing marsh and flood basin habitats that will be inundated tidally and seasonally in the northwestern part of the Delta, and reestablishing a more “normal” hydrograph. A somewhat broader but similar suite of management actions are proposed in the Delta Smelt Resiliency Strategy7. These are worthwhile experiments but could be sabotaged by the multitude of exotic species and contaminants that now populate the Delta as well as by our incomplete understanding of Delta Smelt.

5. Future water management will be driven more by extreme events (of all types) than by long-term averages, even as those averages change.

As California’s climate changes because of global greenhouse gas emissions, more frequent and more extreme storms and droughts will occur. Management will have to restructure to respond to these changes.

The seventh “new” perspective in SBDS 2008 noted that climate change increasingly meant that species conservation is more than a local habitat problem (Healey et al. 2008b). However, Healey et al. (2008b) said little about the forms that climate change was likely to take. For many years, the focus of most assessments of climate-change vulnerabilities in California was on the likely impacts of the projected gradual change in average climatic and hydrologic conditions (Cayan et al. 2008; CDWR 2009; Cloern et al. 2011). Since publication of SBDS 2008, it has become increasingly clear in the Delta as well as globally that the most immediate and difficult problems that climate change will bring will be increased climatic and hydrologic extremes (e.g., Dettinger 2011; Dettinger et al. 2016; IPCC8). Average precipitation may not change much, but more will fall as rain rather than snow. Precipitation

events will be more intense, and the intervening dry periods dryer. Wet episodes will be wetter and dry periods longer, so that the threat of both flood and drought will increase (Dettinger et al. 2016). Reservoir managers will be faced with conflicting challenges: maintain storage space in reservoirs as a hedge against higher flood risk; or store more winter streamflow for summer irrigation and stream-temperature management. Sea level will rise 60 centimeters or more by the end of the century, challenging managers’ ability to control salt intrusion into the Delta from freshwater outflows, particularly during extended droughts. Rising temperatures and declining water supplies will make it difficult or impossible to maintain water temperatures tolerable to native species, particularly during heat waves and droughts. All of these changes will challenge the capacity of the water and environmental management systems to sustain ecologically desirable conditions in the Delta while providing a reliable water supply to millions of Californians. In addition, rising sea level and higher flood flows will increase the risk of levee failure (Deverel et al. 2016), with attendant dramatic effects on Delta agriculture and communities, on water exports from the Delta, and on the ecosystem.

Although climate change places extreme events into clear and urgent focus, the critical role of extremes in the Delta extends beyond climatic extremes supercharged by climate change. With various species on the precipice, with nutrients and foodwebs modified far beyond their natural ranges, with increasing risk of levee failure, with lethal and sublethal cocktails of contaminants in the Delta’s waters, and with water resources pushed to their limits, the capacity of the Delta to absorb extremes of all kinds is declining. Thus, even before weather extremes supercharged by climate change force changes in management, other extremes in the Delta system will command the attention of management much more than they have in the past.

Fortunately, there is time for the water-management system to evolve so as to partially mitigate these effects, and managers now have a number of tools that give them greater flexibility to achieve multiple objectives. Replenishing the depleted groundwater basins in the Central Valley to provide an additional reservoir for use during extended dry periods will be particularly important (Lund 2016). However, both the water management system and water users will have to adjust to reduced and more variable inflows to the Delta and to overall less-predictable sources of water supply. Sustaining a Delta ecosystem hospitable to native species will be much more difficult—perhaps impossible. Under these conditions, it may become necessary to manage for novel plant and animal communities that provide desired ecosystem services (Hobbs et al. 2006) rather than focus almost entirely on attempts to restore native species.

6. Delta habitats work together as a landscape-scale mosaic.

The success of local restoration depends on what happens in adjacent habitats and vice versa. Any habitat restoration, therefore, has cascading effects that propagate far beyond the restored habitat. Landscape ecology provides a set of tools and concepts to identify and take into account these cascading effects.

SBDS 2008 recognized that the Delta’s problems are not entirely local, but emphasized the external forces (such as climate change and invasive species) that restoration activities need to acknowledge and accommodate. In the past decade, the interconnectedness of the internal parts of the Delta has increasingly been recognized as imposing its own level of “non-locality” on Delta decision-making. The Delta is a mosaic of landscape patches that are interconnected geographically, hydrologically, and ecologically (SFEI-ASC 2016; Wiens et al. 2016). As a consequence, what is done at the restoration site itself only partially determines the success or failure of any habitat restoration. Where a particular patch of restored habitat is located in relation to other habitat patches in the landscape—and how it interacts with other patches, both nearby and distant—are also important to the success of a restoration project. Bond and Lake (2003) describe five reasons that habitat restoration may fail in a landscape context:

- If the restored habitat is within a matrix of unsuitable habitat, target species may not be able to find and colonize it.
• The restored habitat may be mainly suitable for one life stage, which may not be the limiting stage for the target species.

• Alien species may be the first to colonize new habitat, and their presence may preclude successful colonization by target species (Brown et al. 2016; Dahm et al. 2016; Moyle et al. 2016).

• The legacy of past as well as ongoing large scale disturbances (e.g., regional land-use changes, hydrologic changes) and large scale temporal change (e.g., climate change) can compromise the success of local restoration (Dettinger et al. 2016; Fong et al. 2016; Luoma et al. 2015).

• The scale of a restoration may be inappropriate for the target species (Wiens et al. 2016).

All of these factors are important in decisions about habitat restoration in the Delta. The science of landscape ecology provides concepts and tools for designing habitat restorations that are robust to these and other factors. As yet, however, the tools of landscape design have not been fully integrated into Delta restoration planning (SFEI–ASC 2016).

7. The situation for native species is dire.

The ecological regime shift coupled with the emerging effects of climate change in the Delta are creating conditions that will likely accelerate the current downward spiral of native species. This situation makes it urgent that the scientific foundations for new management responses be developed.

Despite management actions that in some instances appear heroic, fish species continue to decline in abundance in the Delta (Moyle et al. 2011; MacNalley et al. 2010). Moyle et al. (2016) describe three scenarios for the imperiled Delta Smelt that may also apply to other declining species: (1) extinction; (2) establishment of intensively managed remnant populations in circumscribed habitats, such as flooded islands or upstream reservoirs; and (3) development of a semi-natural, although area-restricted, refuge for Delta Smelt by creating an arc of suitable habitat from Yolo Bypass, through the Cache–Lindsay Slough complex and the lower Sacramento River and into Suisun Bay and Marsh. Option 3 is generally consistent with habitat restoration or enhancement proposals in the Delta Plan and is likely to be undertaken. Option 2 constitutes assisted relocation—a hotly debated conservation option for species on the brink (Hewitt et al. 2011). In SBDS 2008, we argued that it was time to begin a serious debate about whether assisted relocation should be considered as a conservation measure (Healey et al. 2008b). We revisit this issue in the last section of this paper, and explore some policy issues about assisted relocation and other novel conservation ideas.

NEW TOOLS THAT ARE ADVANCING DELTA SCIENCE

Rapid advances in science often follow the introduction of new measurement tools or analytical techniques. The advance in scientific understanding of the Delta that has occurred over the past decade is in large measure a consequence of the development and deployment of such new tools. Here we discuss a few of the tools that we believe have contributed in major ways to the advancement of science in the Delta.

1. Miniaturized acoustic tags have allowed the migratory pathways of salmon and steelhead through the Delta to be measured in detail, including rates of travel and rates of mortality in different Delta channels.

It has been long known that migrating salmon smolts passing through the central Delta survive more poorly than those that remain in the main channel of the Sacramento River. However, traditional tagging could not determine the conditions that brought them into the central Delta, or where the additional mortality occurred. One of the most important new tools for fisheries research in recent years, therefore, has been the miniaturized acoustic tag (McMichael et al. 2010; Pincock et al. 2010). These tags and associated software to analyze data from receiver arrays are now widely used to understand fish migration and survival in the Delta and elsewhere (Welch et al. 2009; McMichael et al. 2010; Perry et al. 2016). The tags are small enough to be implanted in larger salmon smolts, and provide detailed information about individual fish as they migrate through the Delta. By situating telemetry monitoring stations at important junctions and locations, researchers can
estimate migration speed, choice of migration route and survival between monitoring stations. When coupled with real-time hydrodynamic measurements and modeling, the behavior of individual fish can be correlated with tidal mixing and fine-scale features of water movement at junctions. Use of acoustic telemetry techniques has provided detailed information on when and where juvenile salmon travel in the Delta and location-specific survival rates. Managers now have a more thorough understanding of how Delta hydrology affects migrating smolts and the options available to protect them.

2. Deployment of sensors that provide almost continuous measurements of nutrients, carbon and other water-quality variables allows new insights into water quality.

Historically, water-quality monitoring has involved collecting water samples weekly or monthly. The resulting infrequent observations impeded timely responses to unusual discharges of contaminants or nutrients, and limited managers’ ability to identify specific causes (Pellerin and Bergamaschi 2014). Data sets with poor spatial and temporal resolution limited the ability of scientists to model and interpret water-quality processes. The interpretation of these data was also largely based on assumptions of linear system behavior, even though it is well established that hydro-ecological systems are strongly non-linear (Krause et al. 2015). Sampling at fixed intervals also allowed pulsed events, such as discharge of nutrients or contaminants during a storm, to be missed, resulting in serious underestimation of the mean and range of environmental conditions.

Automated, near-continuous recording of water-quality parameters using a variety of fixed and mobile sampling platforms, as well as remote sensing, is increasingly possible. This technology can provide much better information on sources, concentrations, and distributions of water-quality variables. However, the technology can generate huge amounts of data, and the sensors and associated equipment are very expensive. Designing monitoring programs to use this technology cost effectively remains a challenge (Pellerin and Bergamaschi 2014; Pellerin et al. 2016). As a result, automated, high-frequency sampling of water quality has only recently begun in the

Delta (Dahm et al. 2016). Eight nutrient-monitoring stations currently operate in the Delta. Nitrate is the only nutrient continuously recorded, but phosphate and ammonium are recorded on an event basis. The data are being used to identify nutrient sources and nutrient dynamics in relation to phytoplankton uptake, and to provide insight into how the SRWTP affects downstream nutrient concentrations and food web dynamics. A commitment to adaptive high-frequency monitoring in the Delta (Krause et al. 2015) would provide insight into how ecosystem processes are affected by short-term (storms) and long-term (droughts) events, and might permit impending regime shifts to be forecasted (Scheffer et al. 2012).

3. Advances in 2- and 3-dimensional hydrodynamic modeling have allowed much more detailed understanding of water and suspended particle movements in the Delta.

One dimensional (1-D) hydrodynamic models (i.e., models that resolve only average flow in the channel), such as DSM²⁹, have been the mainstay of hydrodynamic modeling in the Delta for many years, and have been coupled with particle tracking modules to simulate the movement of plankton or fish (Culberson et al. 2004; Kimmerer and Nobriga 2008). Although it was well known that 1-D models had important limitations, the computational requirements for models of higher dimension often precluded their development and application. Advances in modeling technology have made it much more practical to develop and apply two-dimensional (2-D, able to resolve flow along the channel and variations in flow either across channel or with depth) and three-dimensional (3-D, able to resolve flow variation along the channel, across the channel and with depth) models. It is now possible to apply multidimensional modeling that spans geography from the Gulf of the Farallones through San Francisco Bay, the Delta, and upstream in the Sacramento and San Joaquin rivers (MacWilliams et al. 2016). More narrowly focused models allow detailed resolution of complex flow and transport patterns at channel junctions over tidal cycles. These models make it possible to address coupled physical/ecological questions that could not be

but also establish critical ecological corridors and habitats, all while ensuring movement of high-quality water from northern to southern California for drinking water and irrigation. Levees are, thus, critical to the human service function of the Delta, and any major disruption of the levee system would have serious consequences for human use of the Delta and its waters as well as dramatic but uncertain effects on the Delta ecosystem. Threats to levee stability include earthquakes, seepage, sea level rise, floods, and land subsidence inside the levees. Because the levees were constructed over a long period of time using a variety of materials, their internal structure and, therefore, their vulnerability are poorly known.

In SBDS 2008, Moore and Schlemon (2008) summarized the history of levee construction in the Delta, the history of levee failure, and what was known at the time about the vulnerability of levees to the various threats listed above. They recommended greater use of geophysical and remote sensing techniques to evaluate the condition and vulnerability of the levee system. In 2007, to determine interior structure of levees, CDWR began conducting topographic surveys of levee condition using combinations of airborne and in situ LiDAR (Light Detection and Ranging) surveys, as well as geomorphic analysis using cone penetrometers, boreholes and electromagnetic surveys.10 These surveys indicated that 84% of the length of levees surveyed conformed to the minimal Hazard Mitigation Plan standard, whereas only 61% of the surveyed length conformed to the stricter PL84-99 standard. Most non-compliant levees were located in the central Delta (Deverel et al. 2016). However, because of the variable material composition of the levees, borehole surveys are insufficient to fully characterize internal levee structure. Over the past decade, remote sensing techniques have been more widely used to assess levee structure. Two techniques that have provided important information about levee deformation are synthetic aperture radar (SAR) and LiDAR (Deverel et al. 2016).

This does not mean that 1-D models can be consigned to the scrap heap. For many questions, a 1-D model is the right tool. Furthermore, comparison of high-resolution 3-D models with lower resolution models shows that in some instances (e.g., mixing at the Georgiana Slough/Mokelumne junction) lower-resolution models can give very similar results. Where a 1-D model is inadequate, the improved representation of a 2-D model may be sufficient, and the extra computational effort of 3-D models may not provide much improvement (MacWilliams et al. 2016). However, physical processes that are inherently 3-D, such as stratified flows associated with salinity gradients in Suisun Bay and the western Delta, generally require 3-D modeling.

We anticipate that modeling of physical processes at various levels of resolution in space and time, coupled with improved population and ecosystem modeling, will become a critical tool for habitat restoration and species recovery in the Delta. We also anticipate that these complex coupled hydro-ecological models will provide insight into the circumstances under which species recovery can actually be accomplished.

4. Remote sensing tools used in conjunction with in situ measurements of levee internal structure from boreholes are providing more reliable mapping of levee structure and vulnerability.

The hundreds of kilometers of levees in the Delta not only protect human life, farmland, and a network of roads, railways, and water, gas and power lines, 

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measurements of surface deformations of the levees. InSAR measurements are taken in image swaths that can be up to hundreds of kilometers wide, while detecting surface deformations as small as 5 to 10 mm. Significantly higher accuracy of 1 to 2 mm deformation per year can be achieved from repeated measurements (Deverel et al. 2016). InSAR also has the advantage that it can see through clouds, smoke, and haze, and can image surfaces without solar illumination. These techniques have provided useful information on changes in levee shape and subsidence that can signal weakness and potential for failure, and offer a basis for much more confident monitoring of levee deformations and breaches.

To predict levee failures, however, the internal condition of the levees also needs to be determined and tracked. Boreholes continue to be the primary source of information on internal condition; however, two promising methods for remote subsurface assessment are Electromagnetic Induction (EM) and Capacitively Coupled Resistivity (CCR), both of which measure variations in conductivity with depth. When correlated with borehole data, conductivity maps can be used to identify subsurface materials between boreholes. A combination of EM and CCR proved useful to assess foundation conditions and potential under-seepage areas, and compared favorably with borehole data along a section of levee on the Feather River. As experience grows with these remote sensing techniques it should become possible to produce reliable maps of levee composition and failure risk.

5. Improvements in—and greater availability of—regional downscaling of global climate predictions are allowing better-informed prediction of local climate changes and their effects.

Each new generation of climate change assessments has included more climate projections from a greater diversity of climate models, thus providing more detailed and varied sets of climate variables. This greater “depth” of the projections has enabled climate projections and their uncertainties—and thereby the climate we will soon be facing—to be better understood (Dettinger et al. 2016). One factor that limits the exploration of these projections has been the fact that global climate models (GCMs) still yield climate projections on coarse spatial grids of 100 to 200 km. To translate these coarse spatial projections of climate change into the finer scales of local climate variability that drive the watersheds and rivers of California, a process called “down-scaling” is performed. Two common approaches to down-scaling are dynamical down-scaling and statistical down-scaling (CCTAG 2015). Dynamical down-scaling involves simulating local-to-regional weather responses to coarse GCM outputs. These models represent the physics of weather and climate as well as we understand them at high resolution and, thus, provide a full suite of climate variables. However, they suffer from significant uncertainties that arise from our incomplete understanding of the factors and forces that determine local weather. These models are also burdensome in terms of computation and data storage. Statistical down-scaling, by comparison, involves using statistical methods to interpolate coarse scale GCM outputs onto much finer scale historical weather observations. Statistical down-scaling’s advantage is that it is less computationally burdensome than dynamic down-scaling and, therefore, can be applied to the many climate-change projections that are now available for analysis. A disadvantage is that statistical down-scaling assumes that historical large- to fine-scale relationships will apply in the future.

At present, statistical down-scaling is the most widely used method and has been a basis for much greater insight into the detailed character of climate-change influences that are most likely to challenge the Delta. One major advance in the past decade has been the development of new classes of down-scaling methods that represent climatic extremes more completely than was possible in the past (Dettinger et al 2016). The resulting high resolution and higher-fidelity climate scenarios have made important contributions to our understanding of the importance of changing climate extremes on the Delta’s future. In years to come, new statistical tools—hybrids of statistical and dynamical modeling, and, eventually, fully dynamical down-scaling—will be needed to address the climate change issues most likely to threaten California water and the Delta.
6. New analytic tools, such as liquid chromatography and high-resolution mass spectrometry, allow water samples to be screened for a very broad spectrum of potential contaminants.

In the past, contaminant studies have typically targeted a few chemicals from a particular contaminant group (e.g., pesticides or metals or pharmaceuticals) because analysis for more than a few possible contaminants was time consuming and costly. In some cases, analytic techniques did not exist for certain contaminants. New products (e.g., pesticides, pharmaceuticals, personal care products, and flame retardants) continually come onto the market, and scientists have struggled to expand their analyses to the latest chemicals of interest. Furthermore, most toxicity values refer to the concentration of a single chemical that causes mortality of one or a few test organisms in a specified period of time. Because of this, relatively little information exists on toxicity of mixtures, on toxicity to different life stages or species not routinely used in laboratory tests, on toxicity of chemical breakdown products, or on a wide range of sublethal effects. Because of these constraints, it was virtually impossible to fully assess the toxicity of Delta water at any particular time and place, or to monitor effectively for chemical contamination. However, toxicity tests are now available that integrate the effects of multiple contaminants, even when they are below the levels detectable by chemical analyses (Fong et al. 2016). To get meaningful results, however, it is crucial that appropriate test species and endpoints are chosen. Toxicity tests with multiple species—preferably across multiple trophic levels, and including in vitro tests for specific sublethal effects—broaden the range of risks that can be assessed but significantly increase the time and resources needed to complete an assessment.

Water samples taken from the Delta contain an unknown cocktail of contaminants at different concentrations. These samples can be tested for toxicity, but if an effect is found it has been very difficult to determine the chemical species responsible. New analytical techniques, such as liquid chromatography-high-resolution mass spectrometry, allow simultaneous analysis for more than 200 compounds and more than 100 transformation products (Moschet et al. 2014). Non-targeted analyses using the new techniques do not require scientists to hypothesize what chemicals they expect to find; instead, chemical spectra can be compared to whole libraries of chemicals for identification of the particular ones present in a sample. Using this new technique, Moschet et al. (2014) determined that, by relying on traditional analytic methods, Swiss authorities were underestimating mixture toxicity by a factor of 2, and up to 10 in extreme cases. Recent evaluations using non-targeted chemical analyses on water samples from the Cache Slough complex detected over 100 chemicals after storm runoff events (Fong et al. 2016). In addition, the toxicity risk associated with chemical mixtures, as well as chemical and species co-occurrence, can be assessed with interactive mathematical models such as the Co-Occurrence Pesticide Species Tool (Fong et al. 2016). These new analytical assessments and models could greatly advance our understanding of contaminant effects because current studies typically analyze fewer than half of the pesticides applied in the Bay–Delta (Kuivila and Hladik 2008) and rarely analyze for pharmaceuticals or other contaminants.

An important adjunct to a better understanding of the complex mix of contaminants in the Delta and their toxicity risk would be the development of bio-markers. Bio-markers are measurable biological responses that serve as indicators of organismal health, allowing sublethal stressor effects to be evaluated. They can be used to predict responses at higher levels of biological organization. Bio-markers are particularly useful as part of a weight-of-evidence approach, and are informative for understanding mechanisms behind sublethal organismal responses, such as behavioral alterations. Bio-markers have been used by generations of epidemiologists, physicians, and scientists to study human disease and have been employed in the Delta to detect physiological response to endocrine disruptors (e.g., Brander et al. 2013; Biales et al. 2015). However, a suite of bio-markers has not yet been identified to address ecosystem health questions in the Delta.

POLICY PERSPECTIVES

The current state of Bay–Delta science is technically advanced and progressing rapidly, but it is also
too narrowly focused in several regards. As a consequence, the papers included in the special issues that comprise the State of Bay–Delta Science 2016 tend to underplay or ignore some major issues. As editors of the State of Bay–Delta Science reports, we identified some missing perspectives and policy implications in composing this summary and felt it was appropriate that we discuss them. We emphasize that, although we may link our discussion to individual papers, the policy perspectives we present are our own and not those of the authors referenced. Some of our discussion elaborates on policy perspectives we previously identified in SBDS 2008 (Healey et al. 2008a). Others highlight the need to evaluate new ways of approaching long-standing problems in the Delta.

**Policy Perspective 1**

The “state” of Delta science is held back by, and needs to push beyond, its tendency to focus on near-term issues and crises. Taking a longer, 50-year viewpoint was part of the Delta Vision exercise. That kind of long-range thinking needs to be incorporated into the whole Delta science and management endeavor. Progress towards that goal is lagging.

In SBDS 2008, we emphasized that the Delta is a continually changing ecosystem. In both SBDS 2008 and SBDS 2016, as well as more generally, much has been made of the changes to landforms and hydrology imposed by human occupation of the Delta and its catchment area, and the multiplicity of alien species introduced either deliberately or inadvertently. However, from its beginnings, the Delta has been changing physically and ecologically. Our concept of the Delta is anchored by what science has revealed about its structure and functioning over the past half century, and by what we can piece together from historic documents and paleoecology (Whipple et al. 2012) about what it was like 200 years or more ago. In the distant past, changes in the Delta were driven by natural processes: sea level rise as the Wisconsinian ice age retreated; tectonic rebound as the ice disappeared; post glacial climate warming with changing weather patterns; and species invasion as land, ocean, and freshwaters warmed in the post-glacial climate regime and as present-day ocean circulation patterns developed (Culberson et al. 2008). These were all dramatic events, but occurred over a long time and before we were present to witness them. Some continue into the present day, but, today, the most important drivers of change are human caused, and occur on a much compressed time-scale.

In light of this continual change, two policy-relevant facts are evident. First, many aspects of the Delta ecosystem probably came into existence in the not-too-distant past, and may have been destined to fade away even if there had been no human-caused transformation of the Delta. Second, the Delta has always been a changing place and, looking to the future, further dramatic change is inevitable. The pace of change is accelerating so that Delta science needs to begin answering the questions, “What kind of a Delta ecosystem it is feasible for us to have 50 or 100 years from now?” and “What actions would allow the Delta to evolve in desirable directions?” To respond to the current and approaching challenges will require more of a scientific (and policy) focus on long range thinking, and acknowledging and determining in a realistic way what we can accomplish in the future, with less focus on trying to re-create what we “had” in the past.

Taking a forward-looking, 50-year viewpoint was part of the Delta Vision exercise, but it is not clear that most scientists and policy-makers are actively continuing that perspective. An appreciation of the changes that are coming, particularly those associated with global climate change (Dettinger et al. 2016), needs to more strongly inform our planning. Even if current efforts to reduce global greenhouse-gas emissions are successful, climate will still change dramatically over the coming century or two. California will become hotter and dryer, and precipitation and water supply will become more variable and less predictable. At the same time, changing land use, agriculture, economic changes both local and global, and other factors will impose their own stresses on the Delta. In the short to medium term, skillful water and environmental management may mitigate some of these effects and allow the Delta to remain more or less as it is (Luoma et al. 2015; Dettinger et al. 2016; Lund 2016). Ultimately, however, rising sea levels and temperature, enhanced floods and long droughts, and changing land use and economic conditions will
transform the Delta as we know it, likely before the end of this century. Even before that time, most of the listed native species will likely be gone from the Delta. Scientific, policy and management frameworks need to be developed now to minimize the negative and maximize the positive consequences of these inevitabilities (Luoma et al. 2015). Delaying action until the crisis is upon us will greatly increase the risk and cost of failure.

A key to developing forward-looking policy is well organized and active science. The Delta has an exceptional science community that has vigorously engaged with the complex problems of water and environmental management. New tools—particularly remote sensing, new analytic approaches and modeling opportunities, and new sensors for real-time measurement of water quality—offer Delta science more powerful machinery for looking forward. Linked climatological-hydrological-ecological models are now possible and would allow scientists to explore a wide range of potential futures and to provide policy makers with plausible scenarios within which to assess policy outcomes. Indeed, advances in modeling technology and computing power now offer the opportunity to develop fully integrated models of the wider Delta ecosystem that extends from the Sierras to the sea (MacWilliams et al. 2016). The USGS-led CASCaDE12 project (Computational Assessments of Scenarios of Change for the Delta Ecosystem) was an initial attempt to develop such a set of models. Work on the components of CASCaDE continues, and this strategy could be elaborated further through inclusion of even more detailed hydrodynamic and water-quality monitoring, and by building in the essential food web dynamics in a landscape ecology context (Brown et al. 2016; Dahm et al. 2016; Fong et al. 2016; Grossman 2016; MacWilliams et al. 2016; Schoellhamer et al. 2016; Weins et al. 2016).

Like global climate models, the kind of integrated ecosystem models we have in mind should be designed to look ahead to develop plausible future scenarios that can inform policy. Like the early global climate models, such models would, at first, include many uncertainties. However, simply constructing such models would reveal a great deal about how the Delta system functions as a whole, about the limits of our current knowledge, and about new directions for both science and policy. Redirecting even 15% of current science budgets to this kind of long-range forecasting would have an important effect.

Policy Perspective 2

Major obstacles to integrating forward-looking science and policy are the common constraints on science within government agencies that tend to focus narrowly on immediate policy and management issues with little freedom to investigate more broadly and to look far into the future. Conflicting agency mandates can also sometimes put agency-supported science at cross purposes.

Science within government agencies, particularly state agencies, tends to be narrowly focused on immediate issues and to be constrained by each agency’s mandate. If, as we believe, science is the key to developing forward-looking policy, then we need more forward-looking science. As the speed and uncertainty of change in the Delta increases—driven by climate change, socio-economic globalization and local developments—closer integration of basic and applied research and greater investment in forward looking research are needed. Compared with the billions of dollars spent on Delta infrastructure, levees, roads, pipelines, and water conveyance, investment in science is tiny. Governments at all levels will need to invest more in exploratory science not linked to any current policy if science is to stand ready to support policy-makers when the inevitable surprises emerge in the Delta (Doak et al. 2008). The Delta Science Plan provides a collaborative framework to bring research and policy to the table, but the science community also needs to be willing to shift some of its emphasis from the immediate to the longer-term future. Short-term science cannot be expected to lay the groundwork to address long-term change. We realize this is not an easy transition for agencies struggling to keep on top of multiple immediate problems, but in times of rapid uncertain change, it is essential.

Policy Perspective 3

We need to begin laying the scientific (and societal) foundations for alternatives to conservation in place, including such approaches as assisted relocation, the

creation of refuge populations, assisted evolution, and genomic banking. At present, scientific understanding is not sufficient for these tools to be engaged with confidence as conservation options. Targeted science within an adaptive management framework is needed to demonstrate the potential of these techniques.

The most vulnerable orphans in this era of rapid change are likely to be the Delta’s endangered native species. Given the scope of the changes the Delta faces, it is increasingly insufficient to focus entirely on a policy of in situ conservation through habitat protection and restoration. Many, if not all, native species are likely doomed in the Delta. This is not a signal that they should simply be abandoned. They are not going to disappear from the Delta tomorrow and endangered species laws require that all reasonable effort be made to provide for them. The increasing likelihood of extinctions is, instead, a strong signal that we should begin exploring alternatives to conservation in place, including approaches such as assisted relocation, the creation of refuge populations, assisted evolution, and genomic banking.

Assisted relocation, which goes by various names in the conservation literature, is hotly debated and divisive (Minteer and Collins 2010; Seddon 2010; Hewitt et al. 2011). Issues run the gamut from ethical to technical. Assisted relocation is presented as either an abandonment of ecological integrity or the last hope for threatened species—as the logical and necessary application of ecological understanding to achieve a desirable goal—or as a flawed and unproven over-reach. Assisted relocation has been attempted many times but often has proved unsuccessful (Dodd and Seigel 1991; Fischer and Lindenmayer 2000). Nonetheless, some deliberate and many accidental relocations have been successful. The Delta, in particular, is a highly invaded ecosystem because of deliberate and accidental introductions of exotic species (Cohen and Carlton 1998; Healey et al. 2008a). This checkered history points to the need for targeted science to determine the viability of this strategy. A good place to start would be Moyle et al.’s (2016) suggestion that a refuge population of Delta Smelt might be established to help ensure the species’ survival. Delta Smelt are already successfully cultured in a hatchery at Byron in the south Delta, providing a source of potential colonists as well as a captive refuge population. However, the technique of successful assisted relocation is likely to differ from species to species, so it will be necessary to begin exploring approaches for other species as well. It is telling that policy options like assisted relocation do not receive even a comment in strategic documents such as the Delta Smelt Resiliency Strategy recently published by the California Resources Agency.

The proper choices of recipient ecosystems for species relocation will require scientific research and public debate. If climate change is the primary driver of extinction, then recipient ecosystems need to be chosen that will remain within species tolerances for decades or longer, even if humanity achieves its goal of preventing average global temperatures from increasing more than 2.0 °C (Kinley 2016). In tandem with scientific uncertainties, legal and ethical issues will also arise. Although not prohibited by the Endangered Species Act or by California’s fish and wildlife laws, such relocations for conservation purposes are problematic under U.S. Fish and Wildlife Service policy (Shirey and Lamberti 2010). Among the ethical issues is the question of which species are worthy of conservation through relocation. As distasteful as it is, rules for species triage need to be developed (Bottrill et al. 2008; Hagerman et al. 2010). Decisions to pursue these options will presumably benefit from a strong scientific foundation that does not yet exist, and will also presumably require wide-ranging discussions by scientists, policy-makers and the general public. Even though extinctions of Delta species from climate change are mostly some time in the future, it is not too early to begin such discussions and, especially, to begin preparing for them. If we wait for the crisis to be upon us, it will be too late.

Some endangered Delta species—such as the Delta Smelt, Lange’s metalmark butterfly (Apodemia mormo langei), and salt marsh harvest mouse (Reithrodontomys raviventris)—are endemic and are restricted to small areas of habitat in the Delta. When these species go extinct in the Delta, they will be gone forever. Such species might be prime candidates for assisted relocation, although endemism should not be the only consideration in such choices. Some listed Delta species are widely distributed (e.g., Chinook Salmon, Steelhead Trout, Green Sturgeon) and their
disappearance from California would not constitute extinction. These species are threatened by climate change from British Columbia to California, and could be extirpated from most of the southern part of their range within a few decades (Healey 2011). At the same time, as Arctic ice recedes, new habitat suitable for salmonids (and sturgeon) is opening up on some Canadian Arctic islands and in mainland Arctic rivers. It might be prudent for California to begin discussions with Canada and Alaska to ensure that suitable habitats for colonization in the Arctic are not compromised by other forms of development. Assisted relocation to speed the colonization of these habitats would help ensure their protection.

Relocation strategies will also raise scientific and policy issues regarding management of the receiving ecosystems. Relocated species can disrupt the receiving ecosystem. It may also be desirable or necessary to modify receiving ecosystems to make them more hospitable to the relocated species. The complexities and the conflict that will attend any proposed relocation will be huge and only partially ameliorated by reliable science, so that adaptive-management experiments will generally be required. Perhaps even more unsettling for agencies and communities in the Delta is the possibility that the Delta might become a receiving ecosystem for threatened species from the south, intentionally or otherwise. Though we may prefer to ignore such difficult questions, they will almost certainly be presented to us. Sound policies to undertake or respond to these options will likely depend on stronger scientific understanding of their prospects and consequences—before they arise.

Closely related to assisted relocation is the development of refuge populations to ensure preservation of critical genomes. Refuge populations can be captive or free living in constructed or appropriated habitats. California and the National Marine Fisheries Service (NMFS) have already done this with a captive winter-run Chinook Salmon broodstock program at Bodega Bay (discontinued in 2004) and for Delta Smelt in a hatchery at Byron in the south Delta. In addition, Moyle et al. (2016) suggested that a refuge population of Delta Smelt could be established in a pond near the Delta. Similar approaches could be taken with other species. Local refuge populations might not be viable for long, as climate and water temperatures continue warming. However, lessons learned in the process might allow the species to be relocated again later, or might buy time for other options to be developed.

A further alternative to conserving species in situ is assisted evolution—helping the species to evolve tolerance to changing local conditions, usually thermal tolerance in the case of climate change (Jones and Monaco 2009). This approach is being attempted with corals near Hawaii, where researchers are selecting for individuals with high temperature tolerance as a way to protect the reefs from bleaching as a result of rising ocean temperature (van Oppena et al. 2015). Artificial selection for desirable traits has been a common practice in agriculture for centuries but has only recently been proposed as a conservation measure. Presumably, modern techniques of genetic modification could be used to hasten the process of establishing higher thermal tolerances in selected species. However, assisted evolution raises important questions about biodiversity conservation. The United Nations Convention on Biodiversity highlights conservation not just of species and ecosystems but also of genetic diversity. The extent to which modification of genomes to adapt species to the effects of climate change can be reconciled with this principle of biodiversity conservation is not clear. A critical question in this context is whether a genetically modified species is the equivalent of the unmodified species.

Another approach with a genetic focus is genome conservation through cryopreservation (Kaviani 2011; Comizzoli and Holt 2014). Like genetic modification, this technology has been relatively little used in conservation, although more so than assisted evolution. The technology is fairly well developed for plant species (Engelmann 2011) and has been promoted to ensure the preservation of charismatic wildlife species such as lions and tigers (Kumar 2012). The appropriate tissue and method depends on the physiology and genetics of reproduction, which is unknown for many species (Fickel et al. 2007), posing important research questions in the Delta context.
Policy Perspective 4

Ecosystems worldwide are lightly to heavily invaded by alien species. To date, the conservation focus, enshrined in the UN Convention on Biodiversity, has been to prevent, eradicate, or contain species invasion. The Delta ecosystem is now dominated by alien species. A growing contingent of conservation ecologists worldwide is calling for active management of invasive species for human benefit.

One of the important stressors on threatened native species in the Delta is the large number of alien species found there. Whatever decisions are made about in situ conservation of native species, large parts of the Delta are likely to remain dominated by alien species. This poses the question, “How should these parts of the Delta be managed?” Too little attention has been given to how best to use the habitats that are no longer suitable (and that cannot be made suitable) for native species. Without downplaying the substantial economic and ecological effect of invasive species, these species are here to stay (Pejchar and Mooney 2009). Important questions for both science and policy are what ecological, recreational, and economic value these species could provide, and how management could realize those values. Fishbio13 ranked the south Delta Largemouth Bass fishery—a species that invaded in the early 20th century—9th out of the 100 best bass fishing waters in the US. Other Centrarchids are also present, contributing to a diverse, warm-water sports fishery. Under global warming, these species will likely fare better in the Delta than native fish. We are not suggesting that native species should be written off (see Policy Perspective 3), but we are suggesting that ecosystems dominated by invasive species can also have value that we should strive to understand and nurture.

Policy Perspective 5

Current Delta science has focused almost entirely on the co-equal goals of water supply reliability and ecosystem conservation, but a legislated constraint on management to attain the coequal goals is largely being neglected. Achievement of the co-equal goals must happen in a manner that protects and enhances the “Delta as Place.” The current state of science in the Delta is lacking in terms of the definition and strictures of this mandate.

Delta scientists have actively pursued research projects to address questions related to the “co-equal goals” of water supply reliability and ecosystem conservation. The Delta science community, however, has missed an opportunity to examine another important legislative condition. The Delta Reform Act of 2009 requires that the coequal goals of water supply reliability and ecosystem conservation “be achieved in a manner that protects and enhances the unique cultural, recreational, natural resource, and agricultural values of the Delta as an evolving place” (CA Water Code §85054). The Delta Plan prepared by the Delta Stewardship Council elaborates on the Council’s understanding of “Delta as place.”14

Existing Delta science uncovers information related to “Delta as place,” and acting on existing science to achieve the co-equal goals can affect “Delta as place,” but these effects are incidental to the research design, are rarely mentioned in the interpretation of results, and are not being effectively communicated through policy channels. There are likely multiple interacting reasons for this deficiency including:

- “Delta as place” is not the mandate of any agency or academic research unit with a significant science budget;
- A key word in the definition of “Delta as place” is “values,” which scientists consider not to be in their domain. However, the goals of water reliability and ecosystem conservation are only pursued because they have values, so applying this argument to “Delta as place” is no more binding than it is for the co-equal goals;
- “Delta as place” is not well specified, making it difficult for scientists to deal with the multiple issues and questions included in the term.

The Delta Protection Commission has sponsored the “Delta Narratives Project” that brought together historians, Delta librarians, and Delta museum archivists to prepare new portrayals of the history of the Delta and to organize a Delta cultural

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bibliography, among other products. The project provides a start on the cultural values of “Delta as place” but it does not provide clear pointers for natural science aspects of “Delta as place.”

In the process of reviewing Delta science in the context of the Delta Plan, the Delta Independent Science Board has been grappling with the questions around “Delta as place.” It intends soon to identify the issues more thoroughly and provide some guidance. However, it is our view that Delta science, across the agencies, should begin to include the “Delta as place” in its planning and analyses.

CONCLUSION
Science has informed policy-making in the Delta for decades, and we expect it will continue to do so into the foreseeable future. The papers included in the State of Bay Delta Science 2016 illustrate major advances that science has provided over the past decade. Now, as we continue through a period of great change and great uncertainty, science is all the more important as a source of reliable, verifiable information on which to base policy. We believe this is a time to invest more heavily in science, particularly forward looking, integrative and exploratory science. As was pointed out in SBDS 2008 (Healey 2008) and in Luoma et al. (2015), the Delta presents managers with a “wicked” problem, a problem that cannot be solved in the traditional sense but only managed. Broadly based, reliable, forward looking, and integrated information is the key to managing such problems. The problems of the Delta are destined to become ever more wicked as climate change unfolds. An investment now in forward-looking science—particularly in integrative models that represent the labyrinth of interacting processes at work in the Delta and that allow the most likely outcomes from a diversity of future scenarios under various management policies to be forecasted—will pay dividends in identifying and testing novel management options, particularly when the inevitable surprises arise (Doak et al. 2008).

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